National Park Service
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Natural Resource Stewardship and Science

Bandelier National Monument

Natural Resource Condition Assessment

Natural Resource Report NPS/BAND/NRR—2015/1000

ON THIS PAGE
View of Upper Alamo Canyon, 2009
Photography by: National Park Service

ON THE COVER
View across Burnt Mesa, Bandelier National Monument
Photography by: Dale Coker

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The National Park Service, Natural Resource Stewardship and Science office in Fort Collins, Colorado, publishes a range of reports that address natural resource topics. These reports are of interest and applicability to a broad audience in the National Park Service and others in natural resource management, including scientists, conservation and environmental constituencies, and the public.

The Natural Resource Report Series is used to disseminate comprehensive information and analysis about natural resources and related topics concerning lands managed by the National Park Service. The series supports the advancement of science, informed decision-making, and the achievement of the National Park Service mission. The series also provides a forum for presenting more lengthy results that may not be accepted by publications with page limitations.

All manuscripts in the series receive the appropriate level of peer review to ensure that the information is scientifically credible, technically accurate, appropriately written for the intended audience, and designed and published in a professional manner.

This report received informal peer review by subject-matter experts who were not directly involved in the collection, analysis, or reporting of the data.

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Executive Summary

The Natural Resource Condition Assessment (NRCA) Program, administered by National Park Service (NPS) Water Resources Division, aims to document current conditions of important park natural resources through a spatially explicit, multi-disciplinary synthesis of existing scientific data and knowledge. The NRCA for Bandelier National Monument (NM) began in 2010 but was interrupted in June of 2011 by the Las Conchas Fire, which burned over 60% of the monument.

This assessment includes a description of selected resources and, when possible, reference conditions, current condition and trends, and factors that affect current conditions or are likely to affect future conditions. It addresses biological integrity at the landscape scale, as well as for communities and species of management concern. Recent drought and concomitant tree mortality, perhaps foreshadowing a predicted future warmer, drier climate, and the modern history of large severe fires are a focus of the report. Air quality, water quality and hydrology, and natural sounds are also addressed.

Bandelier NM protects over 33,000 acres of rugged but beautiful canyon and mesa country, as well as evidence of a human presence going back over 11,000 years. Petroglyphs, cavate dwellings carved into the soft rock cliffs, and standing masonry walls throughout the landscape are a record of a culture that survives in the surrounding communities.

The landscape of Bandelier NM has changed dramatically over the last century. Heavy grazing, prior to and during early federal management of the monument, effectively excluded fire in a system that had previously included frequent, low-severity surface fire. Subsequent national fire suppression policy and the timing of favorable wet climate windows in the American Southwest allowed land managers to manage ecosystem processes and created a semblance of stability across much of the landscape of the monument for most of the 20th century. Fire exclusion promoted dramatic increases in upland forest density and fuel loading during the early and mid-20th century. Subsequent large and severe crown fires burned over large areas of the monument, converting many forested areas into savannas, shrublands, or grasslands.

In the summer of 2010, scientists from the staff of Bandelier NM and the Southern Colorado Plateau Network (SCPN) of NPS convened a scoping workshop to begin work on the Bandelier NRCA. Collectively, the park natural resources staff and collaborating U.S Geological Survey (USGS) scientists brought over one hundred years of experience and knowledge about the park to this effort (see Chapter 3). At the end of two days of lengthy discussions, prioritization exercises, and decision-making, a list of focal resources emerged which provided direction for setting indicators and measures of resource condition. The final list included twenty-three resources grouped under five main categories: air and climate; geology and soils; water; and biological integrity (see Table 4-1). During the initial scoping, the group also decided that the assessment would develop a quantified and spatially explicit description of recent vegetation change within the monument using the wealth of vegetation data that had been collected by NPS, USGS, and collaborating scientists. Given recent drought-induced tree mortality and repeated catastrophic wildfire impacts over the last several decades, the time was right to synthesize current knowledge of broad-scale vegetation patterns in Bandelier NM.

The resulting vegetation analysis indicated drastic vegetation change within Bandelier NM between 1981 and 2004. Woodlands and savanna with juniper as the canopy dominant have expanded. Immediately upslope, the most striking change was the complete loss of mature piñon from piñon-juniper woodlands. Further upslope a continuing contraction and fragmentation of ponderosa pine stands was evident, with increased shrub dominance in the understory. Ponderosa pine-mixed conifer stands have been replaced by ponderosa pine-shrub communities.

The full impact of the Las Conchas Fire on ponderosa pine and mixed conifer forests has not yet been assessed. Nor are the long-term effects or recovery times known for wildlife communities that depend on forested habitats, such as breeding bird communities. The Las Conchas Fire had substantial physical impacts on both Capulin Creek and the Rito de los Frijoles, greatly altering stream morphology, destroying riparian vegetation, and decimating aquatic macroinvertebrate communities. Given the scale and severity of the Las Conchas Fire, the recovery time for the monument's aquatic systems is uncertain.

Many of the rare plant species occurring in Bandelier NM are associated with mesic forested or riparian habitats that have since been severely altered by the Las Conchas Fire. Overall, their current condition within the monument is poor. The last breeding Mexican spotted owls (MSO) within Bandelier NM were observed in 2002. The Las Conchas Fire destroyed most potential MSO breeding habitat in the monument. Likewise, although less is known about their population biology and distribution, Jemez Mountain salamanders—typically found in mixed conifer forests—are likely declining within the monument.

Since the 1990s natural resource management at Bandelier NM has focused on documenting soil erosion rates in piñon-juniper woodlands and developing restoration treatments to foster the recovery of herbaceous understory vegetation. These efforts have not only improved the ecological integrity of piñon-juniper communities, but have also helped to stabilize archeological sites. More than 75% of the known pre-contact archeological sites within Bandelier NM occur within piñon-juniper communities. Most of these sites have experienced adverse effects related to soil erosion and lack of post-fire vegetation recovery. Erosion management and long-term monitoring of restoration response and erosion rates should continue. Research has begun to assess the role of biological soil crusts in degraded and restored piñon-juniper woodlands, as a next step in the multi-decade campaign to stabilize erosion with the monument.

Bandelier Wilderness is designated as a Class 1 area under the Clean Air Act. However, in the most recent assessments, many of the criteria for air quality fell within the "moderate concern" category, including visibility, ozone, and nitrogen deposition. The Clean Air Act bestows an "affirmative responsibility" on federal land managers to protect Class 1 areas from the adverse effects of air pollution. In section 169A, "Congress hereby declares as a national goal the prevention of any future, and the remedying of any existing, impairment of visibility in mandatory Class 1 Federal areas in which impairment results from manmade air pollution."

Bandelier NM serves as a harbinger of the types of changes projected for the American Southwest as a result of anthropogenic climate warming. The recent decadal drought, fires, and insect outbreaks have resulted in a landscape much different from one that would be familiar to the park's namesake. This assessment will help guide future goals and direction as the monument resource management focus transitions to preserving key ecological processes and basic ecological capital, and fostering incremental landscape adaptation in the face of a hotter and drier climate.

Contributors

All natural resource staff in the Bandelier National Monument Division of Resources Management provided their expertise directly as NRCA chapter authors, Chapter 4 section authors, supporting contributors, and/or chapter and section reviewers. Craig Allen was involved with the Bandelier staff in all phases of the project; his careful scientific review and attention to scholarship in the last stage of writing were vital contributions. Collin Haffey's writing skills were put to use on various writing and editing assignments in the final months of the project.

Matt Bowker and David Smith's report, *Assessment of Vegetation Change in Bandelier National Monument, a "Barometer of Change in the National Park System"*, formed the basis of the sections on forest and woodland vegetation change in Chapter 4. The unpublished report is included in its entirety in Appendix B.

The Southern Colorado Plateau Network contributed to the report preparation in several ways. Steve Monroe and Stacy Stumpf wrote the water quality and aquatic macroinvertebrate section in Chapter 4; Steve also contributed to the riparian sections. Jean Palumbo and Sonya Daw edited the document and formatted the final publication within InDesign.

Barbara Judy, Lisa Thomas and Cathy Schwemm served as project managers for the Bandelier NRCA and contributed to its writing. Barbara Judy and Lisa Thomas also served as reviewers.

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Chapter 1: NRCA Background Information

Natural Resource Condition Assessments (NRCAs) evaluate current conditions for a subset of natural resources and resource indicators in national park units. NRCAs also report on trends in resource condition (when possible), identify critical data gaps, and characterize a general level of confidence for study findings. The resources and indicators emphasized in a given project depend on the park's resource setting, the status of resource stewardship planning, and science, the identification of high-priority indicators, the availability of data and the expertise to assess current conditions for a variety of potential study resources and indicators.

NRCAs represent a relatively new approach to assessing and reporting on park resource conditions. They are meant to complement—not replace—traditional issue- and threat-based resource assessments. All NRCAs have several characteristics in common:

1. They address a subset of natural resource issues in each park, not all resources. The resource topics are selected by a team of park staff and outside experts based on a variety of criteria, including legal status, management need, and available data.
2. They assess current condition by comparing various measures of present-day status with ecological reference measures that describe past and/or desired conditions.
3. NRCA products are intended to include a substantive spatial component, when relevant, providing data and analyses in spatial presentation and GIS data formats that can be incorporated in future planning and management efforts.
4. They are organized around ecological indicator frameworks, and conform to national guidelines and standards. Specifically, the products include descriptions of indicator resources, definitions of reference conditions, descriptions of data and analysis methods used, evaluations of level of confidence for each assessment, identification of information gaps and research needs, and references.

Although the primary objective of NRCAs is to report on current conditions compared to reference conditions and values, NRCAs also report on trends, when possible and relevant (i.e., when the underlying data and methods support such reporting). Factors that influence resource conditions, such as past activities or conditions that provide a helpful context for understanding current conditions, may also be included.

The methodology for NRCAs typically involves an informal synthesis of scientific data and information from multiple and diverse sources, with a level of rigor and statistical repeatability that varies by resource, depending on available data and expertise. The credibility of an NRCA results from the data, methods, and reference values used in the project work, which are designed to be appropriate for the stated purpose of the project, and should be adequately documented. There is a close connection with the NPS Inventory and Monitoring (I&M) program: NRCAs utilize I&M data whenever possible, and can potentially contribute to the I&M program by providing current condition estimates and establishing reference conditions and/or baseline values for vital sign indicators.

NRCAs do not establish management targets for study indicators; that process occurs within the realm of park planning and management activities. NRCA products can, however, help park managers define short-term workload priorities, frame data and study needs for important park resources, and communicate current park resource conditions to various audiences. A successful NRCA delivers science-based information that is both credible and has practical uses for a variety of park decision-making, planning, and partnership activities. The condition analyses and data sets developed for NRCAs will also be useful for park-level climate-change studies and planning efforts.

For more information on the NRCA program, visit:
http://nature.nps.gov/water/nrca/index.cfm

Chapter 2: Park Resource Setting/Resource Stewardship Context

2.1 Introduction

2.1.1 Enabling legislation/ Administrative history/Cultural significance

Located in Los Alamos County, New Mexico, the 33,997-acre Bandelier National Monument (NM) contains one of the largest concentrations of prehispanic archeological sites in the American Southwest. Within the monument are more than 3,000 sites, most associated with the Ancestral Pueblo period and dating from AD 1100 to 1550. These sites consist of large villages containing up to 400 rooms, hundreds of small farming hamlets, cliff houses, and scatters of artifacts. Major sites include Frijolito, Yapashi, Tyuonyi, Long House, San Miguel, Painted Cave, and Tsankawi. The park and the surrounding area also contain a high concentration of a unique architectural form called cavates (cliff houses that have been carved out of the soft volcanic tuff bedrock).

Adolph F. Bandelier, a pioneer in the study of Southwest history and ethnology, visited Frijoles Canyon in October 1880 and was the first person to record the existence of many of the major archeological sites in the area. He was guided by inhabitants of Cochiti Pueblo, who have direct ancestral ties to the sites in Frijoles Canyon. Bandelier's scientific and popular writings brought the area to public attention. In the late 1890s the archeological remains in the region were first proposed for protected status under the names of "Pajarito National Park or Cliff Cities National Park." This park proposal, spearheaded by Edgar L. Hewett, included a much larger tract than the current monument boundaries. To further his proposal, Hewett assisted in the development of the Antiquities Act, which became law in 1906, permitting the president to create national monuments "to preserve historic and prehistoric structures and objects of historic or scientific interest" (Antiquities Act, section 2). Bandelier NM was established by presidential proclamation on February 11, 1916, named after Adolph Bandelier, who died in 1914.

Bandelier's original 22,352 acres were administered by the US Forest Service from 1916 until the National Park Service assumed responsibility in 1932. Over time, lands have been added to the monument, and the current boundaries include approximately 33,997 acres. Congress designated 23,267 acres of the monument as wilderness in 1976.

Bandelier NM is a remarkably rich and significant cultural resource site, and its natural setting has affected these resources throughout its history. Ancestral Puebloan communities were established where natural resources were abundant, and the conditions of those natural resources are now affecting the persistence of the evidence of those communities, especially the erosion resulting from past grazing practices and the

Fishing in El Rito de los Frijoles, 1940.

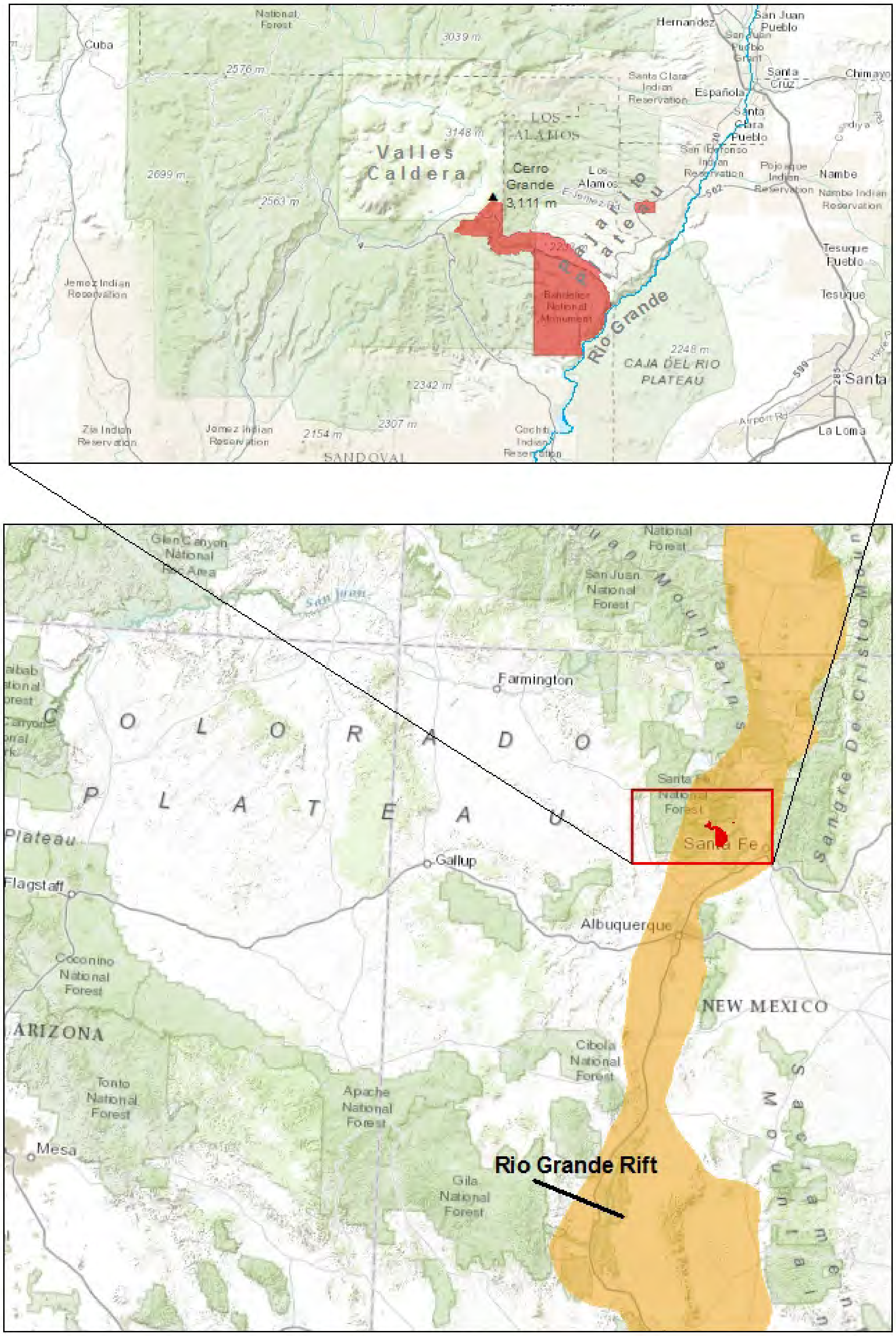


Figure 2-1. Bandelier National Monument is located on the Pajarito Plateau in north central New Mexico.

lack of post-fire vegetation recovery. Because this NRCA is necessarily focused on natural resources, we will not be addressing the cultural contribution of the monument. The reader is directed to several valuable sources for additional information on the human history of the monument, beginning with the “Bandelier National Monument Archaeological and Historic District – National Register Nomination Form” (NR66000042, updated

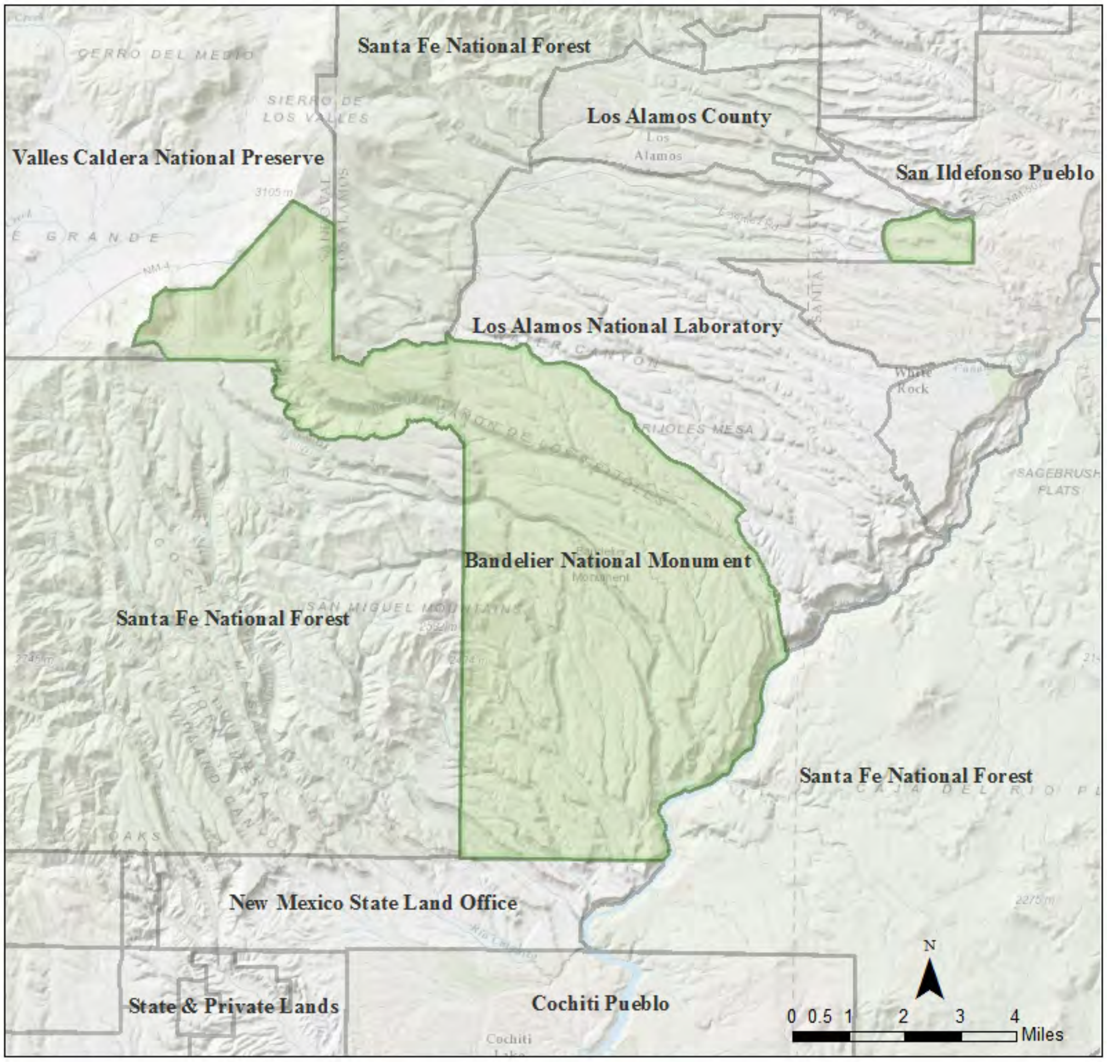


Figure 2-2. Map of public and private lands surrounding Bandelier National Monument.

2014).

2.1.2. Geographic setting

Bandelier NM is located along the southern portion of the volcanic Pajarito Plateau on the southeastern flank of the Jemez Mountains in north-central New Mexico (Figure 2.-1). The Pajarito Plateau is composed of volcanic ash and lava flows ejected from the Valles Caldera in a series of eruptions, the last of which occurred approximately 50,000 years ago (NPS 2014). Erosion has carved the plateau into a series of deep canyons that extend from the edge of the Valles Caldera to the Rio Grande. Of these, Capulin, Alamo, Frijoles, and other smaller canyons are within the monument boundaries. Bandelier NM is bounded on the north along the plateau by Los Alamos National Laboratory (LANL); to the northwest and west by the Jemez District of the Santa Fe National Forest (SFNF); in the northwest corner by Valles Caldera National Preserve (VCNP), and to the south by Cañada de Cochiti land grant and the Rio Grande in White Rock Canyon.

Within the park, there is a long elevational gradient that extends from 1600 m (5,300 ft) at the Rio Grande in the bottom of White Rock Canyon to 3,109 m (10,199 ft) at the summit of Cerro Grande in the Sierra de los Valles (which forms the natural boundary with Valles Caldera). The long elevational gradient, in combination with the geologic, topographic, and soils diversity provides the framework for a wide variety of ecosystem types and complex vegetation patterns.

2.1.3 Surrounding lands/Adjacent ownership

The monument is surrounded by public land managed by federal agencies with a range of mission directives (Figure 2-2). Northwest of the monument is the Valles Caldera National Preserve (89,000 acres/~36,000 ha), federally-owned lands that are at present managed by a public trust. The preserve was re-designated as a unit of the National Park Service by the 113th Congress in December 2014 and will transfer to the Department of Interior in 2015. The area is open to regulated public use and hunting is permitted. To the northeast is Los Alamos National Laboratory (LANL) and the Township of Los Alamos. LANL includes 26,500 acres (~10,700 ha) of open space lands and federal facilities, though use of the land is highly restricted and mostly closed to the public. The remainder of the lands surrounding the monument are primarily within Santa Fe National Forest, including 5,200 acres (~2,100 ha) of the Dome Wilderness directly adjacent to the western boundary of Bandelier Wilderness.

2.1.4 Visitation statistics

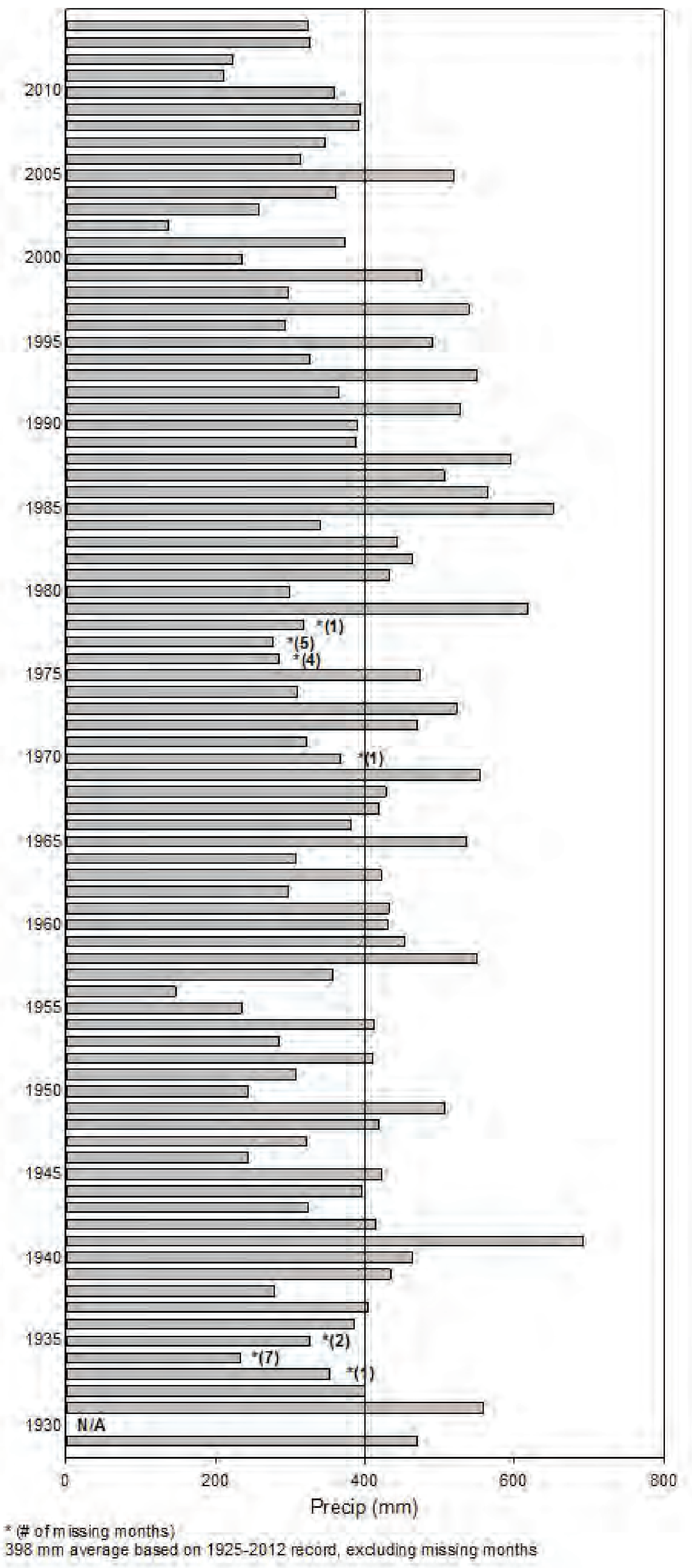
Recorded visitation at Bandelier NM has ranged in recent years from 234,896 (in 2010) to 126,682 (in 2013). During 2010, about 750 people were issued backcountry camping permits. During 2013, about 265 people were issued backcountry camping permits. Camping trips in Bandelier Wilderness generally average one to two nights. Wilderness day use is estimated to be higher than overnight camping use, but since permits are not required for day users the exact number of visitors is unknown. Since the 2011 Las Conchas Fire, visitor use in the wilderness has changed. For example, the southwest corner of the monument is less accessible to hikers due to fire and flood effects, resulting in reduced visitation to Capulin Canyon and Painted Cave. The wilderness portion of Frijoles Canyon, an always a popular destination for day hikers and campers, has received much less use in the past three years due to effects from the fire and subsequent flooding. (Reference: NPS IRMA Park Visitor Statistics and Interpretation Division Backcountry Permit records.)

2.2 Physical resources and processes

2.2.1 Climate

The climate of Bandelier NM is characterized by cool-to-cold, dry winters, and warm, wet (monsoonal) summers. Average monthly temperatures range from lows of -7° to -4°C (20 to 25°F) in December and January, to highs of 30° to 32°C (85 to 90°F) in June. The mean daily temperature extremes range between -10 °C and 6°C (14° and 43°F) in January and 13°C and 30 °C (55° and 86°F) in June, considered the coldest and warmest months of the year, respectively. The coldest recorded daily temperature at nearby Los Alamos was -27.2 °C (-17°F) and the warmest 35 °C (95°F).

Mean annual precipitation is about 427 mm, as recorded at the fire lookout tower near park headquarters (Figure 2-3) (unpublished weather records on file at Bandelier NM). Winter precipitation is delivered principally as snow by low-pressure systems that sweep from west to east across the Southwest, and coalesce with moisture from the Pacific Ocean or the Gulf of Mexico. Winter precipitation is generally followed by a seasonal dry period during the months of May and June. This dry period is defined as much by the increased potential evapotranspiration that accompanies increased day length, solar radiation, and temperatures, as by decreased precipitation. The spring dry period is usually relieved by the onset of the Mexican monsoon; this weather pattern typically delivers at least 40% of the annual precipitation during July through September, and is associated mostly with short-duration, high-intensity thunderstorms. Each summer, as high pressure becomes entrenched off the coast of Baja California, low pressure in the Southwest feeds Pacific moisture across the region, fueling the development of afternoon thunderstorms. The magnitude, frequency,



* (# of missing months)
398 mm average based on 1925-2012 record, excluding missing months

Figure 2-3. Water year precipitation for the period 1929–2014 from the Firetower weather station. The average water year total for the period 1929–2012 is 15.67. Years with missing months are indicated with *(n) with the number of missing months in parentheses.

and tracking of individual large, intense thunderstorm cells during this period can account for large year-to-year variability in annual rainfall and also variability between local areas. The subject of climate and climate change in New Mexico and Bandelier NM are addressed in detail in Enquist and Gori (2008), Fisichelli (2013), Gonzalez (2014), and in Chapter 5 and Appendix A of this report.

2.2.2 Fire

Historically, fire and climate were the primary environmental components that maintained ecosystem diversity at Bandelier NM. Frequent surface fires (6–15 year intervals) in most communities promoted relatively open woodlands of ponderosa pine and mixed conifer forest systems. Cycles of wet and dry periods allowed vegetation to build up during years with above-average precipitation, while dry periods limited recruitment and supported low-intensity lightning-caused fires that further limited recruitment of woody species (Allen 1989). Fire suppression and grazing have now largely eliminated surface fires, and tree densities have increased substantially as a result (Touchan et al. 1996).

High fuel loads, combined with drought conditions have led to a series of increasingly catastrophic fires in the Jemez Mountains over the last 30 years that have destroyed large areas of mature forest and have had numerous detrimental secondary impacts (e.g., exotic plant species expansion, post-fire floods). The 1977 La Mesa Fire burned 15,000 acres (60 km^2), and the 1996 Dome Fire burned 16,500 acres (67 km^2). The Cerro Grande Fire in 2000 originated as a controlled burn but eventually spread to 43,000 acres (194 km^2); while most recently the Las Conchas Fire in 2011 burned over 150,000 acres (600 km^2) and was the largest fire in New Mexico's history. The cumula-

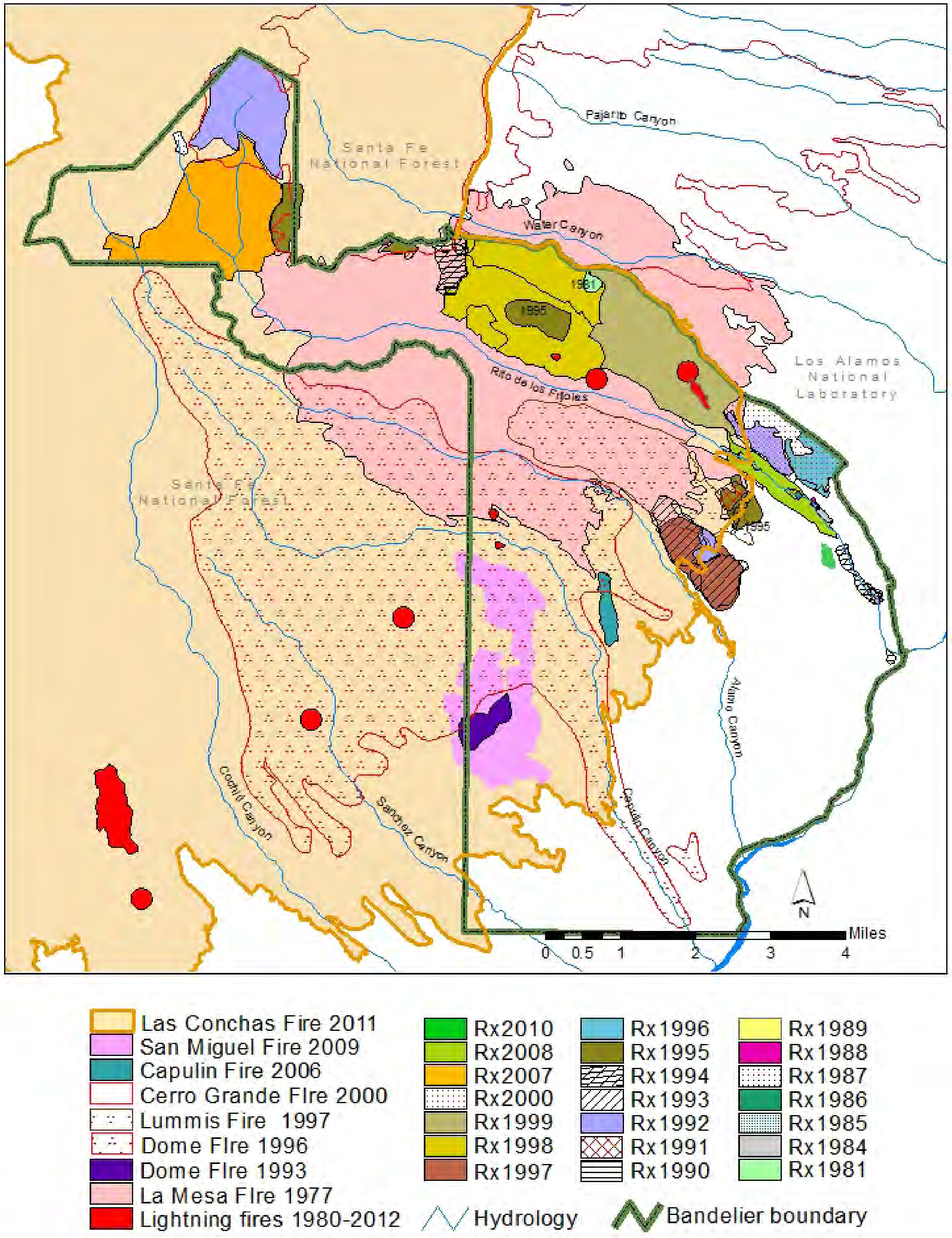


Figure 2-4. Fires greater than 10 acres within and adjacent to Bandelier National Monument for the period 1909–2011. The "Rx" indicates management-ignited fires.

tive result of these events is that there is now almost no forested area of the monument that has not burned in the last 20 years (Figure 2-4). Fire will be a recurring theme throughout this document, and conservation implications discussed in Chapter 5.

2.2.3 Geology

Geologically, Bandelier NM is located on the southeastern flank of the Jemez Mountains, which lie at the intersection of the Jemez lineament and the Rio Grande rift. The Jemez lineament is a chain of volcanic centers extending from Arizona to Colorado, and the Rio Grande rift is a crack in the earth's crust extending from the Rocky Mountains of central Colorado to Chihuahua, Mexico. This geologic interface was the site of a series of volcanic events related to tectonic movements beginning some 16 million years B.P., culminating in two massive explosions that led to the formation of the Toledo and Valles Calderas at 1.61 and 1.23 million years ago, respectively (Spell et al. 1993). The Toledo eruption ejected an estimated 396 km^3 of rock and ash, while the Valles eruption produced about 292 km^3. By comparison, Mt. Saint Helens erupted 25 km^3 and Krakatoa 18 km^3 of new material.

The Valles and Toledo eruptions cumulatively deposited the 300-meter thick Bandelier Tuff in two distinct layers that now dominate the Bandelier NM landscape as part of Pajarito Plateau (the lower Otowi and upper Tshirege members, respectively). The most recent eruption, El Cajete at 50–60,000 B.P., covered the local landscape with many meters of pumice, much of which was subsequently eroded and reworked, leaving pumice patches predominately on east-facing slopes and deep alluvial deposits on lower slopes (Wolff et al. 1996). Along the western boundary of the park lie the San Miguel Mountains, which are comprised of older Tertiary and andesitic and rhyolitic volcanics along with sedimentary sandstones of the Santa Fe Group and Galisteo Formation. The south-to-north trending Parajito Fault Zone ranges from the base of the San Miguel Mountains northward, distinctively separating the Pajarito Plateau from the San Miguel Mountains and the rim country to the west known as Sierra de los Valles and its associated mesas (Sawyer Mesa, Mesa del Rito). The Sierra de los Valles are made up of dacites associated with Cerro Grande and Sawyer Dome, along with rhyolites of Rabbit Mountain. In contrast, the Pajarito Plateau is bounded to the east by White Rock Canyon, a deep gorge containing the Rio Grande that has extensive exposures of Tertiary mafic lava beds.

2.2.3.1 Terrain and watershed characteristics

Topographically, the layered volcanic rocks of the Pajarito Plateau provide a structural control that has led to a series of deeply incised southeast-trending, steep-walled canyons (Frijoles, Lummis, Alamo, Hondo, Capulin, Medio, and Sanchez) alternating with broad mesa tablelands. The tablelands themselves are moderately incised with small drainages that create an undulating topography of small canyons and intervening "interfluves," particularly at the lower, distal ends of the mesas. Conversely, the upper portions of the mesas are flatter and are bounded by the structurally uncontrolled slopes of the San Miguel Mountains and Sierra de los Valles. Along the plateau escarpments and canyon sides there is a distinctive banding of cliffs, rock outcrops, and rubble zones that reflect the stratigraphy of the various members of Bandelier Tuff and other volcanic rocks. Conspicuous are the dramatic pink-to-orange cliffs of the Tshirege Member of Bandelier Tuff that can be over 250 m (820 ft) tall.

The canyon and valley bottoms can contain relatively broad floodplains (100–200 m; 325-650 ft) filled with deep sediments delivered by perennial, intermittent, and ephemeral streams. Frijoles Canyon and Alamo canyons together occupy 9,062 ha (22,392 acres), and Frijoles Canyon contains the only consistently perennial stream, El Rito de los Frijoles. The smaller Hondo, Capulin, Medio, and Sanchez Canyons together account for 4,087 ha (10,099 ac), and have intermittent zones of perennial waters. These drainages join the Rio Grande within White Rock Canyon, which in turn forms the eastern boundary of the monument. The Rio Grande, with its headwaters in the San Juan Mountains of southwestern Colorado, can deliver high discharges, particularly with spring snowmelt (>140 cms, 5,000 cfs), along with significant sediment deposition. While the gorge confines the river on a broad scale, there are floodplain deposits on both sides of the river that are up to 130 m (427 ft) across and support riparian and wetland vegetation. The Rio Grande is impounded 20 km south of the park boundary at Cochiti Lake reservoir. Reservoir filling began in 1972, and the pool at different times has extended up into White Rock Canyon through the monument and into the adjacent tributary canyons some 13 km (eight miles). This has created an identifiable high-water zone as high as 30 m (98 ft) up slopes above the current Rio Grande and side canyon floodplains.

2.2.4 Soils

A first-order soil survey by the Natural Resources Conservation Service (NRCS) was recently completed for Bandelier NM and made available in digital form (Hibner 2005). Thirty-nine soil map units were described and delineated at a 1:24,000 scale, and these are generally organized in broad terrain groups and along an elevation gradient. The map units are made up of various combinations of 34 soil series (and variants) from a wide variety of soil families. At the lowest elevations in White Rock Canyon and surrounding areas are a suite of units that represent the bordering escarpments, mesas and plateaus along with the bottom-land floodplains of the Rio Grande and immediate tributaries. These soils are variously derived from basalt, sedimentary bedrock, or alluvium.

Lower-elevation soils of the plateaus and mesas are derived from colluvium, slope alluvium, or eolian-deposit parent materials over rhyolitic Bandelier Tuff residuum. This group includes moderately developed soils (halplustalfs) of plateau tops (Canuela, Hackroy, and Nyjack series); weakly developed shoulder and backslope soils (ustorthents and ustipsamments) of the Palatka, Zacaton (south facing), and Abrojo (north facing) series, and rock outcrops. In contrast, this group also includes Armenta and Adornado soils that are derived from rhyolitic El Cajete pumice, which overlays the Bandelier Tuff. These soils are deeper (haplustolls), and have comparatively better developed but coarser surface horizons (sandy and gravelly loams). At similar elevations are soils of the inter-plateau canyons and valleys. These include Navajita and Piojillo soils of the lower colluvial toe slopes of canyon sides, along with alluvial soils of valley floors and floodplains (Totavi, Espiritu/Petegral, and Metate).

Mid-elevation mesa tops and slopes of the Pajarito Plateau are represented by a similarly structured group of soil units. Rotado and Tocal soils are relatively deep Paleustalfs and Haplustalfs, respectively, that are derived from rhyolytic tuff and occupy the summits of the interfluves. The shallower Urioste (Ustorthents) and Cymery (Haplustepts), along with rock outcrops, characterize shoulders and backslopes of the canyons. El Cajete pumice soils are represented again by the Adornado series on the plateau tops and the Cajete Series on the slopes. A cooler version of Metate occupies the alluvial terraces of the valley bottoms.

The highest-elevation plateau soils typically occur above the Parajito Fault Zone at greater than 2,300 m (7,500 ft). Tschicoma, Hoxoh, and Jemez are the deeper soils of the plateau summits and summit toeslopes (Argiustolls, Haplustolls, and Haplustalfs, respectively). Shallow Cymery, Urioste, and Estaban soils occupy the slopes, along with rock outcrop. Above the plateaus lie the mountain soils with Mapache and Lucito occupying the lower slopes and Casey the upper slopes and summits of the Sierra de los Valles; Wauquie and Laventana are andesitic soils found on the slopes of the San Miguel Mountains. Tschicoma and Tranquilar soils occupy small valley footslopes and floors,

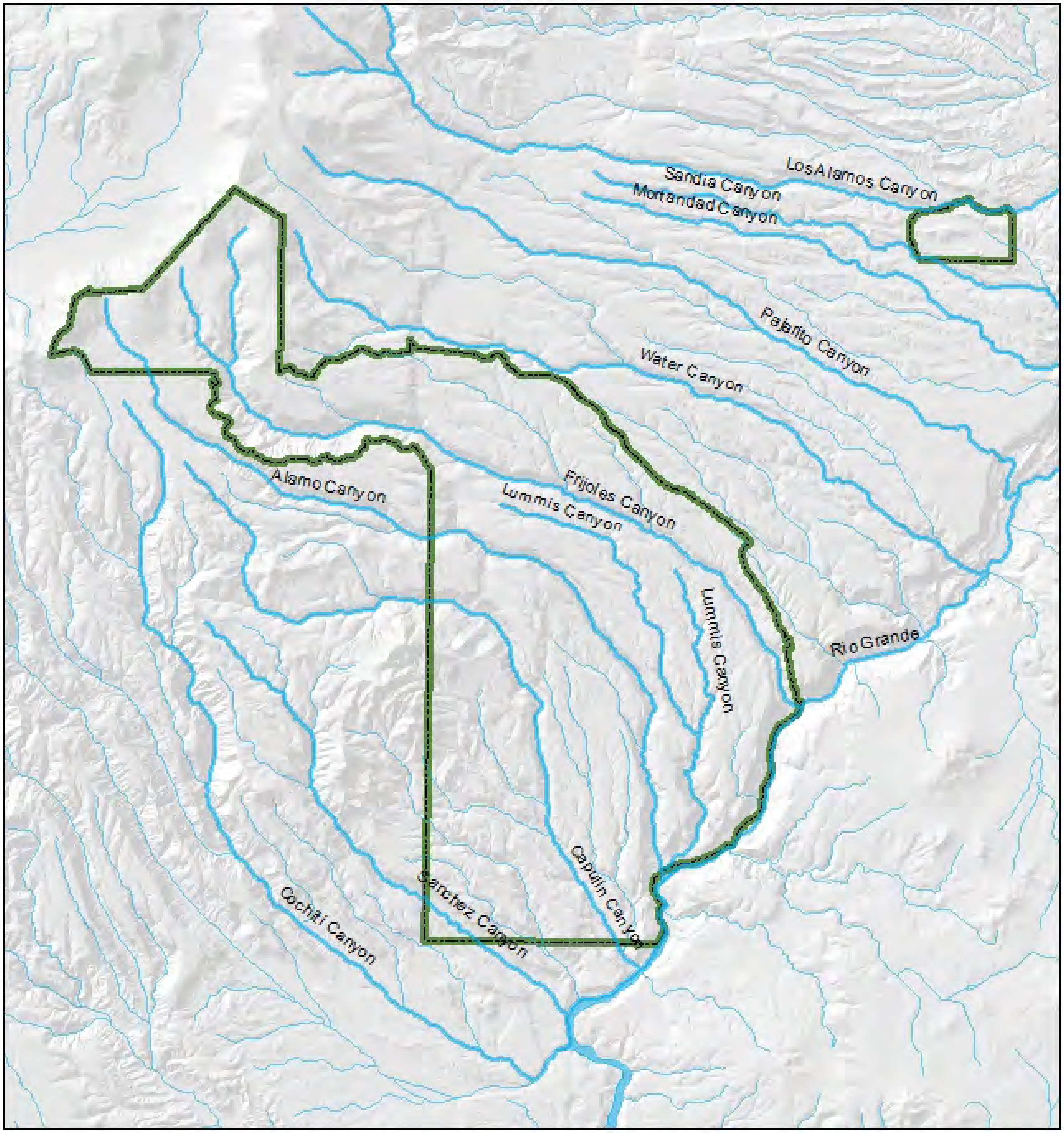


Figure 2-5. Map showing the canyons and drainages of Bandelier National Monument and adjacent lands, with the major canyons and perennial streams labeled.

respectively.

(A detailed soils description is included in Hibner 2005 and NPS 2007.) Erosion is addressed in section 4.01.

2.2.5 Hydrology and water quality

Bandelier NM is located in the Rio Grande-Santa Fe subwatershed (HUC 13020201) within the much larger Rio Grande watershed (Figure 2- 5). The monument contains significant freshwater resources, including springs, perennial and ephemeral streams, wetlands, and groundwater. Partial water sources for streams in the park are springs and seeps at relatively high elevations near the north and west borders of the park, however, most of the flowing water originates on Santa Fe National Forest and Valles Caldera National Preserve lands outside park boundaries. Of note are the two perennial streams in the monument—the Rito de los Frijoles and Capulin Creek. Both flows can become partially subsurface during very dry years. The southeastern boundary of the monument is the Rio Grande River. All water courses in the park generally flow northwest to southeast.

The quality of the freshwater resources in Bandelier NM is of concern to managers (Weeks 2007), and impacts come from two primary sources. Potential for airborne legacy waste deposition from LANL is monitored by LANL, the New Mexico Environment Department Water Quality Division, and by NPS. More recently, the landscape-scale Cerro Grande Fire and the Las Conchas Fire have led to substantial erosion that may have accelerated (temporarily) the leaching of toxics from surface sources, but which have also degraded water quality by increased sedimentation. Water quality is addressed in section 4.11.

2.3 Significant biological resources

2.3.1 Vegetation communities

The Jemez Mountains support a diversity of ecosystems sculpted by variations in elevation, soil type, topography, climate and fire history. A vegetation map for Bandelier NM was completed in 2011 (Muldavin et al. 2011; Figure 2-6), that delineates and describes the vegetation communities in detail; this chapter provides only brief descriptions of the most common plant community types. Much of the vegetation documented in the 2011 map was damaged or destroyed in the Las Conchas fire, however, and a thorough discussion of those impacts is presented for pinyon-juniper woodland, and ponderosa pine and mixed conifer forests in Chapter 4 and Appendix B.

2.3.1.1 Rio Grande corridor

The boundary at the Rio Grande includes the lowest elevations (4,000 ft/1219 m) of the monument. As a natural community, the riparian ecosystem along the Rio Grande has experienced enormous change during the last century, due largely to water impoundments and flow management, both upstream and downstream of the park, but also as a consequence of domestic grazing and exotic vegetation.

2.3.1.2 Montane grasslands

Montane grasslands typically occur in clearings between 7,500 and 10,199 ft (2286–3109 m) and can include many herbaceous species. Prior to 1900, historic high fire frequencies and vigorous grass competition maintained largely treeless grasslands. In the Jemez Mountains there is evidence that with fire suppression during the last 130 years conifers have invaded these high montane grassland sites, creating young forests and savannas rather than open grasslands and meadows.

2.3.1.3 Mixed-conifer/aspen forests

At the higher elevations in the monument (6500–10,170 ft/1980–3100 m) mixed-conifer communities can include Douglas-fir (*Pseudotsuga menziesii*), white fir (*Abies concolor*), aspen (*Populus tremuloides*), Engelmann spruce (*Picea engelmannii*), blue spruce (*Picea pungens*), and limber pine (*Pinus flexilis*) as well as ponderosa pine (*Pinus ponderosa*). Over the last 125 years, the structure and composition of mixed conifer forests have changed dramatically. Fire suppression in the 20th century allowed the development of dense sapling understories in many mixed conifer forests, with tree regeneration dominated by Douglas-fir and white fir.

2.3.1.4 Ponderosa pine forests

At somewhat lower elevations (5,800–10,000 ft/1,770–3,050 m) ponderosa pine forests occur generally below or interspersed among the mixed-conifer forests. Ponderosa pine vegetation communities can be described as forest, woodland or savanna, depending on canopy density, but generally have ponderosa as the dominant overstory species. Ponderosa pine forests across the southwestern U.S. are threatened by several factors, but primarily by changes in fire regimes that have

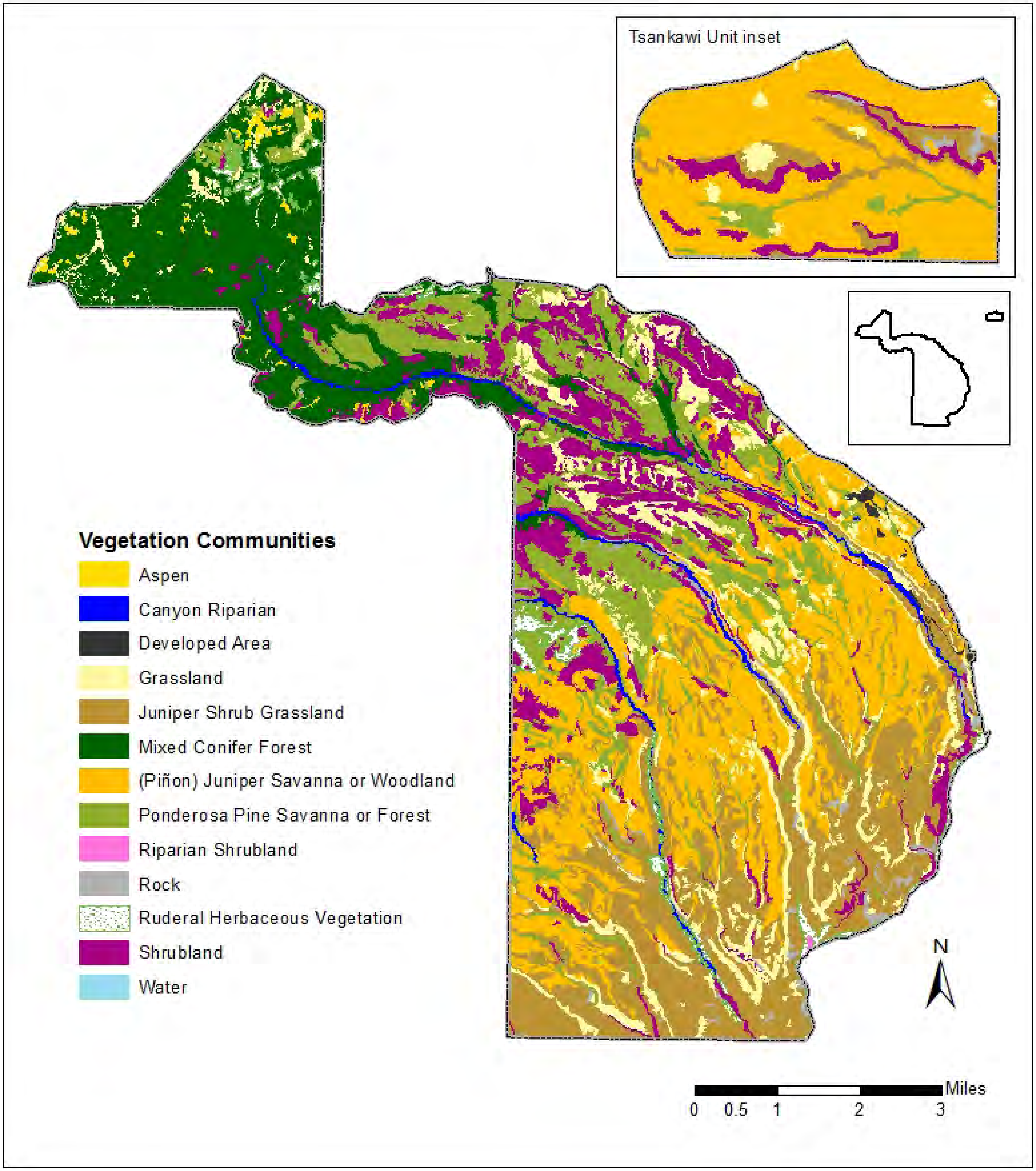


Figure 2-6. Vegetation map showing dominant vegetation types of Bandelier National Monument as of spring 2011, before the Las Conchas Fire burned approximately 60% of the park (Muldavin et al. 2011).

led to unnaturally high tree densities, and by interacting impacts of climate change and drought.

2.3.1.5 Piñon -juniper woodlands

Piñon-juniper woodlands are located in lower warm/dry areas from about 5,350–7,400 ft/1,630–2,260 m, and, at some sites, intergrade with shrub and grassland vegetation. The ecology and distribution of piñon-juniper communities have changed profoundly during the last century, and these changes are particularly apparent in the Jemez Mountains region. Under some conditions, the density of piñon pine (*Pinus edulis*) and juniper (*Juniperus* ssp.) species has increased dramatically, due to the combined effects of fire suppression and grazing. Grazing has reduced or eliminated the herbaceous understory that historically carried low-intensity fires and regulated piñon and juniper abundance; and the absence of fires has allowed trees to survive and mature in greater numbers than would have occurred with natural fire regimes.

In other areas, previous piñon-dominated woodlands recently have been decimated by the related impacts of drought and beetle infestation. Between 2002 and 2003, over 90% of the mature piñons in the monument and on the Pajarito Plateau were killed by a combination of these factors. At present the remaining mature piñons occur only at cooler upper elevations or in localized moister microsites.

The loss and degradation of the piñon–dominated woodland communities are likely permanent changes; soil loss, increasing temperatures, catastrophic fires and the absence of seeds are all factors that impede both natural and human-assisted restoration. Piñon –juniper systems are addressed in section 4.06 and Appendix B.

2.3.2 Vertebrate wildlife

2.3.2.1 Mammals

The diverse plant communities within Bandelier NM support a variety of wildlife species. Two extirpated species and 59 extant species of mammals have been documented at the park (Bogan et al. 2007). Fifteen bat species are known to occur, and of those the spotted bat (*Euderma maculatum*) and big free-tailed bat (*Nyctinomops macrotis*) are of special interest to the park and the state. Mule deer (*Odocoileus hemionus*) and elk (*Cervus canadensis*) are more abundant in the park today than prior to Euro-American arrival. Bighorn sheep (*Ovis canadensis*) had been previously extirpated from the park; New Mexico Game and Fish Department released fifty Rocky Mountain bighorn sheep into Cochiti Canyon in August 2014. Bandelier NM is part of the historic range of the gray wolf (*Canis lupus*) and the grizzly bear (*Ursus arctos*), although neither animal currently exists within the park boundaries.

2.3.2.2 Birds

170 bird species have been documented in Bandelier NM. The 2014 State of the Birds report identified western forest birds that need conservation efforts, as well as common birds in steep decline (NABCI 2014). The report identifies Yellow-Watch-List species which are birds with small populations restricted to a small range, or birds that are more widespread but with troubling declines and high threats. For Bandelier NM, these species include: band-tailed pigeon (*Patagioenas fasciata*), flammulated owl (*Otus flammeolus*), Mexican whip-poor-will (*Antrostomus arizonae*), rufous hummingbird (*Selasphorus rufus*), Lewis's woodpecker (*Melanerpes lewis*), olive-sided flycatcher,(*Contopus cooperi*), pinyon jay (*Gymnorhinus cyanocephalus*), Virginia's warbler (*Oreothlypis virginiae*), Cassin's finch (*Haemorhous cassinii*), evening grosbeak (*Coccothraustes vespertinus*). On the list of common birds in steep decline and regularly occurring in the monument are common nighthawk (*Chordeiles minor*), Wilson's warbler (*Cardellina pusilla*), and pine siskin (*Spinus pinus*).

2.3.2.3 Reptiles and amphibians

A 2002–2003 inventory of the park (Nowak and Persons 2008) observed 17 species of reptiles and amphibians, with an estimated inventory completeness of 65%. There are 44 species known or expected to occur in Bandelier NM. The Jemez Mountains salamander (*Plethodon neomexicanus)* has undergone range-wide declines in recent decades and is listed as threatened by the state of New Mexico. Further information about the Jemez Mountains salamander can be found in section 4.15

2.3.3 Threatened/Endangered species

Threatened and endangered species, such as Mexican spotted owls (*Strix occidentalis lucida*) and Jemez Mountain salamanders, can be found in the Bandelier Wilderness, as well as delisted and monitored animal species such as the peregrine falcon (*Falco peregrinus*) and bald eagle (*Haliaeetus leucocephalus*).

2.3.3.1 Mexican Spotted Owl (*Strix occidentalis lucida*)

Status: Threatened. Mexican spotted owls (MSO) are federally threatened, and until the summer of 2011 several of the major canyons within Bandelier NM were known to have suitable nesting and/or roosting habitat for the birds (Personal communication, Stephen Fettig). Though no owls have been observed during annual surveys since 2002, it was hoped that nesting birds were either going undetected or would return. The Las Conchas fire of 2011 likely destroyed any remaining habitat by removing all of the standing trees that MSO may have been using for roosting and protection. MSO are addressed in section 4.14.

2.3.3.2 Jemez Mountain Salamander (*Plethodon neomexicanus)*

Status: Endangered. The Jemez Mountain salamander is the only endemic amphibian in the Colorado Plateau region, and was listed as federally endangered throughout its range in 2013. Critical habitat was designated in USFS lands west of the monument in mixed conifer forests at elevations between ca. 7,500 to 10,500 feet, where downed trees and sufficient ground litter provide important microhabitat. The Las Conchas Fire (2011) affected much of the known salamanders in Bandelier NM; although these fossorial animals were underground and almost certainly survived the fire itself, the varying fire severities and associated alterations in vegetative habitat had unknown effects on affected populations of this terrestrial salamander. Salamanders are addressed in section 4.15.

2.3.3.3 Bald eagle (*Haliaeetus leucocephalus*)

Status: Delisted, monitored. Bald eagles winter in Bandelier NM, and winter roosting and fishing habitats are located near canyon mouths and along the Rio Grande River.

2.3.3.4 The New Mexico Meadow Jumping Mouse (*Zapus hudsonius luteus)*

Under study by the U.S. Fish and Wildlife Service (USFWS) for Endangered status. Under such designation, portions of the monument might be designated as critical habitat.

2.3.4 Species of concern

2.3.4.1 Peregrine falcon (*Falco pereginus*)

Due to impacts from DDT, peregrine falcons were one of the first animals in the U.S. to be listed under the Endangered Species Act in the late 1960s. The primary strategies for peregrine recovery were to raise birds in captivity that would be free of DDT then to release them into protected habitat after the chemical was banned (USFWS 1984). Since that time peregrine falcon populations have recovered extremely well, and the species was officially removed from the Endangered Species list in 1999 (Mesta 1999). However, USFWS requires continued monitoring until at least 2015 (unless populations decline in which case additional action will be taken; USFWS 2003). Consequently NPS still considers breeding peregrines to be a species of concern.

Suitable nesting areas for peregrine falcons occur in and immediately adjacent to the monument, and include piñon-juniper woodlands and ponderosa pine and mixed conifer forests. The 2006 annual surveys indicated the presence of an occupied nest in the park. The Peregrine Falcon Habitat Management plan identifies three management zones that surround suitable nesting ledges and describes visitor use and management that will prevent impacts, particularly to breeding falcons.

2.3.4.2 American Pika (*Ochotona princeps*)

Pikas are small lagomorphs (rabbit family) distributed across most of the high altitude regions of western North America that have been proposed for listing. Though found at high elevations, pikas do not hibernate and spend much of the relatively short, high-altitude summer foraging and collecting vegetation for winter use. Interacting, multiple effects of climate change (e.g., declining precipitation, higher temperatures, less snowfall), now appear to be significantly reducing the number of supportive sites needed to maintain functioning metapopulations of pikas. Pikas are addressed in section 4.18.

2.3.4.3 Rio Grande cutthroat trout (*Oncorhynchus clarki virginalis*)

This species may have been extirpated many decades prior to the Las Conchas Fire as a result of introducing non-native rainbow, brown and brook trout into all the major park streams, but was certainly lost following the fire in 2011 and associated extreme floods. Trout are addressed in section 4.12.

2.3.4.4 Grama-grass cactus (*Sclerocactus papyracanthus*)

The status of this species in Bandelier NM is undetermined; it was documented in the late 1980s but has been rarely seen since then. Grama-grass cactus is a small, relatively short-lived species that is often cryptic in its habitat and similar in appearance to the grama grass (*Bouteloua* spp.) with which it is often associated. Grama-grass cactus is distributed across New Mexico and adjacent portions of Arizona and Texas at elevations between about 5,000 and 7,510 ft (1,525–2,290 m). Grama-grass cactus is included in section 4.07.

2.3.4.5 Yellow lady's slipper (*Cypripedium parviflorum* var. *pubescens*)

The yellow lady's slipper orchid is extremely rare in the Jemez Mountains and is known from only a few localities, though it is widely distributed across North America. It occurs in relatively open and grassy mixed conifer forests of upper elevation, mesic canyons, on well watered benches, seeps, and bogs on the north facing sides (Personal communication, Brian Jacobs). This species tolerates shade to nearly full sun conditions in fairly open sites within riparian-associated forest communities, meadows and clearings, however, many of the populations documented in the monument prior to the Las Conchas Fire in 2011 were located near small spot fire locations from the 1977 La Mesa Fire (Personal communication, Brian Jacobs). Yellow lady's slipper is included in section 4.07.

2.3.4.6 Cerro hawthorn (*Crataegus erythropoda*)

This hawthorn is an uncommon, but locally abundant, small tree of well-watered upper canyon areas. This species is found only in the Rocky Mountain states of Colorado, Arizona, New Mexico, Wyoming and Utah, and in the Jemez Mountains is found between about 7,000–8,000 ft (2,130–2,440 m) in elevation. In Bandelier NM it occurs in upper Frijoles Canyon. The global status of this species is generally secure, though it may have been extirpated from Bernalillo and Sandoval counties in New Mexico. Prior to 2011 it was uncommon in the monument, but locally abundant in moist areas of upper canyons. Though directly impacted by the 2011 and 2013 flood events, re-sprouting by several individuals has subsequently been observed (Personal communication, Brian Jacobs). Cerro hawthorn is included in section 4.07.

2.3.5 Internal streams and aquatic resources

2.3.5.1 Capulin Creek and Rito de los Frijoles

The Capulin Creek watershed is a designated wilderness area, managed for recreational use within the boundary of Bandelier NM. The upper reaches of the watershed outside of the park boundary are managed for recreation and timber harvest by the U.S. Forest Service. In 1996 the Dome Fire burned several thousand acres in the Capulin Creek watershed. On 26 June 2011, the Las Conchas Fire ignited and ultimately burned approximately 60% of the land within Bandelier NM, including the majority of the upper portions of Capulin Creek. All canopy and understory vegetation was burned, and subsequent flooding by a large event on 21 August 2011 drastically changed the geomorphology of Capulin Creek within the park.

Rito de los Frijoles is a perennial stream flowing eastward from the Sierra de los Valles to the Rio Grande. The upper reaches of the watershed are a designated wilderness area, managed for recreational use within park boundaries. The Bandelier NM visitor center and numerous archeological sites are located near the stream in the lower portion of the watershed, resulting in high levels of visitor use. In 1977 the La Mesa Fire burned about 6070 ha (15,000 acres) in and near the Rito de los Frijoles watershed. The 2011 Las Conchas Fire burned a large portion of the upper portion of the Rito de los Frijoles watershed. A large flood event on 21 August 2011 significantly affected channel morphology and physical habitat of Rito de los Frijoles. See section 4.10 for information on the Rito de los Frijoles and Capulin Creek riparian communities

2.3.5.2 Aquatic macroinvertebrates

Aquatic macroinvertebrates serve as the primary food base for many aquatic vertebrates, function as the primary processors of energetic inputs in low order streams like those found in Bandelier NM, and are consequently important indicators of ecosystem health. Communities of aquatic macroinvertebrates in streams are commonly monitored alongside physical and chemical properties of water because of their potential to provide additional indications of water quality and overall hydrologic condition (Brasher et al. 2011). The Southern Colorado Plateau Network has been monitoring Capulin Creek and Rito de los Frijoles since 2009. (See section 4.11 for more information on aquatic macroinvertebrates.

2.3.5.3 Native fish

Historical information on native fish occurring in the creeks that are within the park is lacking. Native Rio Grande cutthroat trout (*Oncorhynchus clarki virginalis*) populations were extirpated from Capulin Creek by large flood events following the Dome Fire. The New Mexico Department of Game and Fish, Bandelier NM, and the Santa Fe National Forest collaborated to reintroduce this species to Capulin Creek in 2006. The Las Conchas Fire in 2011 and subsequent flooding extirpated all fish from both Capulin Creek and Rito de los Frijoles.

2.4 Relevant regional or landscape scale natural resource information

2.4.1 Elk

Elimination of top predators, and the intentional maintenance by the state of New Mexico of a large herd of elk (*Cervus canadensis*) for hunting purposes, have resulted in a population of elk that is imposing substantial impacts on vegetation. Elk utilize all lands on the plateau, and can have especially potent impacts on shrubs and aspen, particularly post-fire (personal communication, S. Fettig). The role of elk in monument ecosystems is discussed in section 4.17.

2.4.2 Cochiti Reservoir

The Rio Grande River forms the eastern boundary of Bandelier NM from the mouth of Frijoles Canyon on the north to a point midway between Alamo and Capulin Canyons to the south, approximately six river miles (10 km). About 12 miles (20 km) upstream of Frijoles Canyon, the Rio Grande enters White Rock Canyon (WRC), a deeply cut gorge of relatively recent geologic origin, and the river remains within this canyon system until it emerges downstream at the Cochiti Reservoir Dam some 6 miles below the park's southern boundary. Numerous ecological impacts resulted when the dam was constructed and the reservoir filled, including the extirpation of a native plant population (an orchid, *Epipactis gigantea*), and the impairment of several perennial springs that stopped flowing when they were covered with sediment (Personal communications, Brian Jacobs). Seasonal inundation and drawdown of the reservoir continues to support the establishment of weedy species along the perimeter of the lake and riparian areas upstream. Riparian systems upstream of Cochiti are addressed in section 4.10.

2.5 Threats/stressors to important park resources

A comprehensive discussion of threats and stressors to park resources can be found in Chapter 4. Excerpted below are the most critical and encompassing of these impacts.

2.5.1 Climate change

The topic of climate change and changes in the size and intensity of wildland fires is greater than can be addressed here, but is arguably the greatest ecological threat to systems and species at Bandelier NM and across the Colorado Plateau. The cumulative effects of climate change on natural resources and physical processes at the park are incorporated in nearly every topic in Chapter 4, and addressed further as a management concern in Chapter 5.

2.5.2 Fire history and ecology

Fire has played an essential role in shaping and maintaining the vegetation communities and landscapes in Bandelier NM and the Jemez Mountains. Many factors, including climatic conditions, a high occurrence of lightning strikes, availability of surface fuels and flammable vegetation, and topography make fire one of the dominant natural disturbance processes in this region. Most of the vegetation communities and wildlife that have persisted through time here have evolved under the influence of frequent fire. However, natural fire processes, to which vegetation communities and species in the region have adapted, have been altered in multiple ways through human actions, and now climate change.

2.5.3 Park-wide vegetation change

At the scale of the entire monument, there are four tightly-linked phenomena responsible for most change that has occurred in recent decades: 1) climate change, 2) fire, 3) drought/insect outbreak, and 4) past land use. Other factors that strongly affect particular ecosystems in the park, such as elk herbivory or persistent accelerated erosion, are discussed in later sections.

2.5.4 Erosion in piñon-juniper woodlands

Accelerated rates of soil erosion within large portions of the semi-arid piñon-juniper woodland zone were first identified as a management issue in the 1970s in connection with early soil mapping and efforts to control a feral burro (*Equus asinus*) population (Earth Environmental Consultants 1978, Chong 1992). However, it was not until the mid-1980s when park-wide archeological survey efforts began to systematically document erosional impacts to cultural sites.

2.5.5 Nonnative plants

In Bandelier NM, exotic species comprise approximately 15% of all the plant species found within the park, however <5% (e.g., ~40 of ~800 taxa) are considered invasive (NPS 2006). And of those, only about 10-12 species are both high risk and easy to treat. Several species are considered nuisance species in that they are common and fairly wide-ranging, usually in disturbed areas, but they are either weak competitors or treatment on a large scale is not practical. The most challenging management species are those that are so invasive that they are ubiquitous across the landscape and very difficult to constrain.

2.6 Resource stewardship

2.6.1 Management directives, planning guidance and research related to natural resources

2.6.1.1 General/Resource Plans

Master Plan (NPS 1977). The importance of "ethnographic, scientific and educational" values at Bandelier NM was defined and articulated in the 1977 *Master Plan*. The goals of protecting and interpreting the ruins and preserving the park's natural setting were identified as the two primary purposes of the monument.

Resource Management Plan and Environmental Assessment for Bandelier National Monument (NPS 1988). The plan called for landscape ecology research and actions, including inventory of monument flora;

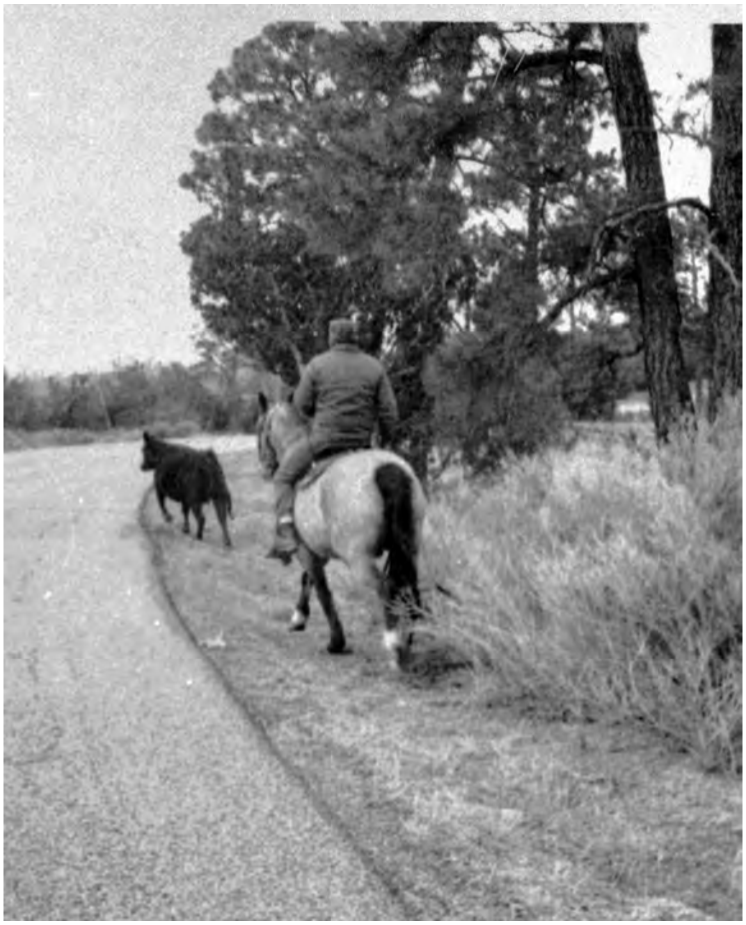
a paleo-environment study of the entire monument; management of burros, cattle and native ungulates; and focus on fire management as a key tool, among other areas of management focus.

Statement for Management (NPS 1990). This update of the park's *Master Plan* addressed the need for managing cultural and natural resources; providing for management-oriented scientific study of issues related to soils erosion and effects of fire suppression on vegetation; and documenting changes resulting from human activities.

Resource Management Plan (NPS 1995). Natural resource management goals and objectives specified in the document were to:

- preserve, protect, interpret and manage the cultural and natural resources of the park within naturally functioning ecosystems, consistent with cultural resource preservation
- provide the means and opportunity for people to study, understand, and enjoy the resources of the monument without unduly compromising the resources or ethnographic values
- restore and sustain natural ecosystem conditions and processes unimpaired from human influence to the degree practicable given landscape and cultural resource constraints
- carry out a wilderness management program which preserves and restores resource conditions and values defined by law and policy and is compatible with cultural resource management objectives
- preserve a comprehensive natural resource base for its value to promote scientific and educational interest

At the time the most critical resource issues were 1) loss of naturally-functioning ecosystems resulting in increased erosion, 2) large-scale vegetation change and loss or alteration of natural processes (predation, fire), and 3) a lack of scientific data, and increasing human activities and external development.



Moving cattle on Monument Road, Bandelier National Monument, 1973. NPS photo

Fire Management Plan (NPS 2005). The purpose of the monument's Fire Management Plan (FMP) was to describe fire and resource management goals and objectives and to provide a framework for incorporating those goals when making fire and fuels management decisions. Specifically, the use of wildland fire was described as the "practice of allowing a naturally ignited wildland fire to burn in a predefined geographic area, under specific prescription parameters, to accomplish fire and resource management goals and objectives". Ironically, as of fall 2014, there is very little area left in the monument that has not been burned in large wildfires or prescribed burns since 1996 (including the Las Conchas Fire of 2011, which alone burned over 60% of the park), and the manner in which the FMP will be applied in the future is uncertain. The FMP is currently (2014) being revised.

2.6.1.2 Specific Resource/Restoration Plans

An Environmental Assessment to Decide How to Eliminate Feral Cattle from Bandelier NM (NPS 1994b) evaluated a range of alternatives and adopted direct reduction by shooting as the proposed action. This plan has been implemented as needed since its adoption in 1994, including as recently as 2009. The plan was challenged in federal court by the New Mexico Livestock Board, who asserted a financial loss following cattle reduction in 1994. A federal magistrate denied the claim by the NM Livestock Board, clearing the way for monument staff to continue implementing feral cattle removal.

Integrated Pest Management Plan, Bandelier National Monument (Jacobs 1994). This administrative document explains the integrated pest management policy at Bandelier NM. Program components and pesticide use and responsibilities are included. The listing of pests include ants, bears, bees, feral cats, cockroaches, coyotes, feral dogs, clothing moths, poison ivy, raccoons, skunks, mice, packrats, squirrels, chipmunks, ticks, feral cattle, and feral burros. For each pest listed there is a description of its biology, impacts, action threshold, and control methods.

Bandelier National Monument Peregrine Falcon Habitat Management Plan (NPS 1994a). Peregrine falcons were delisted in 1999 under the Endangered Species Act. Bandelier NM continues to follow the Peregrine Falcon Habitat Management Plan because across New Mexico this falcon continues to show signs of low breeding success. Productivity (number of young per adult pair) has declined over the 2001–2013 period. Productivity was below 0.9 young per adult pair in 2013 with 1.1 young per adult pair being the minimum rate needed to maintain the state-wide population over the long-term.

The monument's Peregrine Falcon Habitat Management Plan sets disturbance limits for breeding habitat within the park. Disturbance limits are identified on maps as concentric polygons around the most suitable nesting habitats. The largest concentric polygons apply early in the breeding season (starting March 1 each year) when the birds are most sensitive to disturbance. Inner polygons apply later in the breeding season, when young are in the nests. Disturbance limits apply to number and duration of use of all aircraft and motorized equipment, and number of people in walking groups. A critical aspect of the plan is that all nesting habitats are managed as if occupied, unless field observations demonstrate a lack of occupancy at specific sites.

Water Resources Management Plan (Mott 1999). This administrative document presents water resources issues within the context of the monument's setting. Bandelier NM was established to preserve what remains of the area's once thriving Ancestral Puebloan culture. Springs, streams, and riparian zones allowed these ancient agrarians to flourish in an otherwise harsh landscape. The occurrence of water over a wide range of elevations and microclimates continues to support the monument's diverse assemblage of plants and animals, and provides the visitor from today's world a different manner of sustenance.

Biological Assessment Bandelier National Monument Fire Management Plan (NPS 2004). This Biological Assessment (BA) was prepared to meet the requirements of Section 7 of the Endangered Species Act, as amended (ESA) (19 USC 1536 [c], 50 CFR 402), and to assess the effects of implementation of the proposed alternative of the EA on federally listed, proposed or candidate species or their critical habitat that are known to be or could be present within Bandelier NM. This BA integrates research and documentation on federally listed species in the park and the protection measures developed as a collaborative effort of biologists and managers with the NPS and the USFWS.

Vegetation Management Plan (NPS 2002). This administrative document explains the vegetation management policy at Bandelier NM. Vegetation management is a component of many diverse activities planned or on-going at the park. A variety of routine activities, such as road and trail maintenance, hazard tree mitigation and developed landscape maintenance, can be considered vegetation management actions. Prescribed fire, exotic plant control, rare plant management, ruins stabilization and disturbed site revegetation are other management actions which can affect vegetation. Since vegetative systems are dynamic, changes occur in the absence of management actions as well; no action or, alternatively, suppression of 'natural forces', such as wildfire, can allow significant changes to occur. A vegetation management plan provides a context for these diverse activities, prescribes actions, and assigns responsibilities. It also enables park managers to assess the cumulative effects of vegetation management actions park-wide. Through vegetation management planning, managers can understand the vegetation dynamics within their park and coordinate short-term actions to achieve long-term goals.

Exotic Plant Management Plan (NPS 2006). This administrative document explains the exotic plant management policy at Bandelier NM. Written by park botanist Brian Jacobs and approved by Superintendent Darlene Koontz, the plan acknowledges that introduced species already established within the park are likely to become naturalized components of the local flora. Park management seeks to minimize the impacts of these immigrants and contain their spread, but complete eradication is in most instances impractical. At Bandelier NM, primary efforts will be focused on those 'targeted' species which can be effectively controlled or contained using mechanical and/or limited chemical methods.

Bandelier National Monument Final Ecological Restoration Plan and Environmental Impact Statement (NPS 2007). The purpose of the *Ecological Restoration Plan* is to direct efforts to re-establish healthy, sustainable vegetative conditions within the piñon-juniper woodland and to mitigate accelerated soil erosion that threatens cultural resources. This plan evaluates two options for reversing the problems identified above and includes the No Action alternative as a baseline for present management conditions. The primary goal in the Preferred Alternative is to re- establish healthy, sustainable, grass dominated plant communities within the piñon-juniper woodland to help stabilize soils and cultural resources.

Bandelier National Monument Acoustic Monitoring Report (NPS 2014). In 2011, the Natural Sounds and Night Skies Division received a request to collect baseline acoustical data at Bandelier NM. During the months of February and June, 2012, four acoustical monitoring systems were deployed for 28 and 35 days, respectively.

The goal of the technical assistance request was to complete a baseline soundscape inventory throughout the park, especially in the much-visited Frijoles Canyon portion of the park. The results of this inventory will be used in conjunction with a visitor soundscape survey to establish indicators and standards of soundscape quality that will support the park in developing a comprehensive approach to soundscape management planning. This project will develop a Soundscape Desired Condition for the park's pending Foundation for Planning and Management document, provide an impact assessment threshold for development projects within the canyon, and assist the park in assessing potential effects from ongoing and proposed air tourism and helicopter overflights related to park operations.

Pending ~2017 - Fisheries Management Plan. Monument staff have requested assistance from regional subject matter experts to develop a Fisheries Management Plan for the perennial streams in Frijoles and Capulin canyons. Topics to be addressed in the plan include fish management and restoration conditions.

2.6.1.3 Resource Stewardship Strategy

Each national park is directed to develop a Resource Stewardship Strategy (RSS) as part of the park management planning process. Indicators of resource condition, both natural and cultural, are selected by the park. After each indicator is chosen, a target value is determined and the current condition is compared to the desired condition. The completion of this NRCA is the first step in completing an RSS for the monument. Management plans will then be developed to outline actions to be taken over the next 15 to 20 years that will help achieve or maintain the desired condition(s) for each indicator. The RSS will be a multi-disciplinary effort, incorporating a variety of information from different sources.

Pending 2015 – Resource Stewardship Strategy, with Climate Change Vulnerability Assessment. Monument staff has requested assistance from regional and national program staff to carry out an RSS. As preplanning for the RSS, Bandelier NM managers have undertaken development of a Foundation Document (completion scheduled for June 2015; contact Greg Jarvis). In addition, the monument is scheduled for a State-of-the-Park Workshop in March 2015 (contact Jim Haskell). Cumulatively, these advance efforts will have identified key resources and issues and compiled the information required for the development of the RSS. The park is in an excellent position to undertake this planning effort given several decades of research and monitoring which have characterized existing conditions, trends, recent disturbances, and likely trajectories. Much of this is compiled and documented in the NRCA along with emerging issues and challenges that will provide the basis for a comprehensive RSS. Finally, since about 1995 the park has experienced a series of extreme climate events and associated ecological disturbances which, if properly addressed within the context of an RSS, could provide rationales for coherent (proactive versus reactive) management going forward.

2.6.2 Status of supporting science

2.6.2.1 History of the Natural Resource Program and the Bandelier Ecology Group

The natural resource program at Bandelier National Monument was formalized in the mid-to-late 1970s with the development of John Lissoway as the Monument's first natural resource management specialist and, with the strong support of the regional science office in Santa Fe, initiated a number of basic inventory and monitoring efforts. The La Mesa wildfire in 1977 and subsequent post-fire flooding in Frijoles watershed, along with acquisition of the Cerro Grande lands for upper watershed protection, also stimulated additional research work.

Management concerns about soil erosion and feral burro impacts in lower elevation woodland communities, and the potential for catastrophic crown fire in upland forests were major drivers for targeted research, including fire history reconstructions to support the prescribed fire program. Much of the early fire-related work was subsequently documented in the La Mesa symposium proceedings, including the beginnings of integrated natural-cultural efforts.

The resource program began to gain critical mass during the 1980's with an influx of outside researchers (including Brian Jacobs and Craig Allen), and particularly in 1990 with the arrival of a strongly supportive superintendent, Roy Weaver, who then hired a dynamic chief of resource management, Charisse Sydoriak, in 1991 when John Lissoway became the first full-time fire management officer at Bandelier. As part of this critical mass, the park hired Craig Allen as an ecologist in 1989, Kay Beeley as a biologist/GIS specialist in 1992, Brian Jacobs as park botanist/vegetation specialist in 1992, Stephen Fettig as wildlife biologist in 1994, and Laura Trader in 1995 as fire ecologist with the Bandelier fire monitoring program begun in 1992.

By the early 1990s a core staff was in place at the park, and Weaver and Sydoriak continued to develop an integrated natural/cultural/fire resource management program throughout the 1990s, until the Cerro Grande fire brought a close to that chapter of park history. Brian served as the Bandelier NM Botanist until his retirement in December 2014 (see the inset for a description of

Brian Jacobs, botanist, began working in Bandelier National Monument in 1986. He retired in December 2014.

Brian Jacobs retires

Throughout his career at Bandelier National Monument, Brian Jacobs has engaged in both landscape scale research and management of the natural resources in effective ways that have fundamentally improved the current and future condition of the park's ecosystems. Beginning in the late 1980s, with the first comprehensive flora of Bandelier, Brian established a key foundation for all subsequent vegetation studies and management activities. He managed the field collections from the flora in an on-site herbarium, which served as a resource well outside the boundaries of the monument and is now on permanent loan in the regional herbarium collection of the Museum of Southwest Biology at the University of New Mexico and available online from the University of Wyoming.

Brian Jacobs will likely be remembered for his key role in the extraordinary project to ecologically restore large portions of the park's piñon-juniper woodlands. He both contributed vital research on piñon-juniper woodlands and was instrumental in overseeing implementation of the project across the woodland portions of Bandelier. He lined up and oversaw related research activities of graduate students (Richard Gatewood and Brian Hastings), and sizable grants to analyze, treat and monitor the affected woodland landscape between 2002-2012.

Brian's research on piñon-juniper woodlands helped distinguish the age of piñon-juniper stands based on growth characteristics such as size of individuals, and stand and canopy structure. This research contributed to a management framework for different "types" of piñon-juniper and has helped conserve the unique ecosystem across much of the Colorado Plateau. In Bandelier, through years of testing potential approaches, Brian determined that herbaceous ground cover could be markedly improved and soil erosion rates greatly reduced, simply by thinning relatively young piñon-juniper trees (and lopping and scattering their stems as a coarse woody mulch), thereby promoting a more resilient woodland closer to historic pattern and process conditions.

Overall, Brian's contributions to Bandelier supported natural patterns and processes in the ecosystems of Bandelier and adjoining landscapes. He worked on his own and with the NPS Exotic Plant Management Team to eradicate backcountry exotic plants (particularly Tree of Heaven (*Ailanthus altissima*), toadflax (*Linaria vulgaris*), and Russian olive *(Elaeagnus angustifolia*)); he designed and implemented the highly successful rehabilitation of several miles of and old logging and fire road on Burnt Mesa, now an attractive and popular hiking trail; he made many contributions to the park's fire management program, ranging from major inputs into various fire management plans to supervising the multi-park NPS fire monitoring program for a number of years; he was the main architect and driver of Bandelier's current vegetation management plan; and for over 20 years he was the main park liaison for all things hydrological, from gaging stations to water quality studies, including development of a well-regarded water resource management plan for Bandelier.

Brian also focused his energy on studying landscape changes across Bandelier, including the effects of severe drought since ca. 2000, numerous fires, and the ecological restoration project. He partnered with colleagues to acquire diverse data on vegetation and geomorphic changes across the monument through multiple approaches, ranging from plot-level monitoring to remote sensing. For example, in 2014 he initiated a study with researchers from the USGS Canyonlands Field station on the ecological role of microbiotic crusts in semi-arid piñon-juniper woodlands, and their potential to promote or impede vascular plant community recovery after historic landuse.

Brian's contributions to Bandelier National Monument typically were undertaken quietly but substantively in his thoughtful and understated manner, characterized with extraordinary competence and effectiveness. His durable legacy, including valuable research studies implemented, development of long-term natural resource monitoring programs and associated data collected and archived, creation of high-quality management documents, and careful implementation of ecologically-based land management actions, will remain hugely important far into Bandelier's future.

his contributions), while Craig, Kay, Steve, and Laura are still working at Bandelier.

In 1993 the ecologist position was transferred to the new National Biological Survey (NBS), but Craig Allen remained based at Bandelier as a research ecologist, continuing to work directly together with Kay Beeley as the "Bandelier Ecology Group". Dr. Allen was a core partner in building the 1990s resource program. Informed by the solid foundation of his 1989 dissertation work on ecological changes in the Jemez Mountains landscape he identified critical natural resource issues (e.g. soil erosion in piñon juniper woodlands and impacts on cultural resources, stand structure and fire hazards in upland forests, alteration of natural flow regimes on Rio Grande by the U.S. Army Corps of Engineers, elk overpopulation and historic grazing impacts, negative impacts on park resources from activities on adjacent lands) and worked with park management to garner additional support for the program and recruit/build a credible staff.

With the addition of vegetation, GIS, fire, and wildlife expertise to the park's natural resource program, Bandelier's capacity and efforts to address a number of outstanding issues increased markedly during the 1990s, while continuing to build the information base and attract outside researchers. By 1995 the NBS became part of the U.S. Geological Survey, and since then Craig has led the USGS Jemez Mountain Field Station and with Kay the Ecology Group at Bandelier, providing the park with a stable ecology program and expertise which has supported core monitoring efforts while fostering an active research program with numerous external collaborators. USGS ecologist Collin Haffey joined the team near the end of the NRCA project. The long-term synergy between the USGS field station and NPS resource management staff allowed for great strides throughout the past two decades, despite the programmatic setbacks incurred in the aftermath of the Cerro Grande wildfire event. Collectively, this group holds over 125 years of resource management, monitoring and research experience in the Bandelier landscape.

2.6.2.2 NPS Inventory and Monitoring Program

With a mission to improve overall park management through expanded use of scientific knowledge, the Inventory and Monitoring (I&M) Program was established to collect, organize and provide natural resource data, as well as information derived from data through analysis, synthesis and modeling (NPS 2011).

The primary goals of the I&M Program are to:

- Inventory the natural resources under NPS stewardship to determine their nature and status
- Monitor park ecosystems to better understand their dynamic nature and condition and to provide reference points for comparisons with other altered environments
- Establish natural resource inventory and monitoring as a standard practice throughout the National Park System that transcends traditional program, activity, and funding boundaries
- Integrate natural resource inventory and monitoring information into NPS planning, management, and decision making
- Share NPS accomplishments and information with other natural resource organizations and form partnerships for attaining common goals and objectives (NPS 2011)

To realize these goals, 270 parks with significant natural resources were organized into 32 regional networks. Bandelier NM is part of the Southern Colorado Plateau Network, which also serves eighteen additional parks. Through a rigorous multi-year, interdisciplinary scoping process, each network selected a number of important physical, chemical, and/or biological elements and processes for long-term monitoring. These ecosystem elements and processes are referred to as 'vital signs', and their respective monitoring programs are intended to provide high-quality, long-term information on the status and trends of those resources.

SCPN conducts long-term monitoring of a number of vital signs in Bandelier NM (Thomas et al. 2006). Network staff monitors aquatic macroinvertebrates and water quality in Capulin Creek and in El Rito de los Frijoles. They monitor upland vegetation and soils in the pinyon-juniper woodlands and work in conjunction with the Bandelier Fire Effects Program to monitor the mixed conifer forest. Through a cooperative agreement with Northern Arizona University, they also monitor upland bird communities in the mixed-conifer forest. The network's Land Surface Phenology project uses Moderate Resolution Imaging Spectroradiometer (MODIS) satellite data to monitor the phenology and condition of vegetation and snow cover in all SCPN parks, including Bandelier NM.

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Chapter 3: Study Approach

The Bandelier National Monument (NM) Natural Resource Condition Assessment (NRCA) project was coordinated by the Southern Colorado Plateau Network (SCPN). In January of 2010 the program managers of the Southern and Northern Colorado Plateau Networks hired Cathy Schwemm as a term GS-11 Ecologist to assist both networks with coordination of NRCA projects. The NRCA Ecologist was later funded through a cooperative agreement but continued to perform similar functions throughout the project. In addition, the Bandelier NM staff provided substantial input to the project, including project definition and direction, data summaries and analysis, GIS support, writing, and review. An outside cooperator was funded to focus on the topic of highest priority, and this work also was reviewed by the SCPN program manager, the Bandelier NM staff, and Craig Allen, Jemez Mountain Field Station, U.S. Geological Survey (JMFS-USGS).

3.1 Preliminary scoping

The preliminary scoping workshop was held in July 2010 at Bandelier NM Headquarters in Los Alamos NM. The group that was convened for the initial scoping continued their substantive involvement over the course of the NRCA project. The NRCA team included Bandelier staff Kay Beeley (GIS Specialist), Brian Jacobs (Botanist), Barbara Judy (Resources Management Chief), Laura Trader (Fire Ecologist), and Stephen Fettig (Wildlife Biologist), as well as Craig Allen (JMFS-USGS Research Ecologist). Lisa Thomas (SCPN Program Manager) and Cathy Schwemm (NRCA Ecologist) provided project coordination with able assistance from Barbara Judy. The scoping workshop was held over two days and began with an introduction by SCPN on the general goals and process for completing the NRCA. The remainder of the time was devoted to developing a preliminary list of focal resource topics. To achieve this, the group first constructed a complete list of all possible natural resource topics, but did not attempt to prioritize the topics or identify data sources. The group then discussed how appropriate each topic was for the NRCA, whether the resource was of high concern for management and/or legal reasons, what types of data were currently available and how complete and usable the relevant datasets were, and finally what the relative priority level might be for each topic.

Prioritizing topics was difficult, but necessary for several reasons. First, NRCA guidelines state very clearly that the NRCAs will focus on, '…a subset of important natural resources in national parks.' Given the funding levels for NRCAs and the complexities of national park ecosystems, it is not possible to include all natural resources at the level of assessment required (nor is it probably useful). The group discussed at length how each potential topic would be addressed within the guidelines of the NRCA and whether it was appropriate to include a given topic, perhaps at the expense of something else. It was sometimes difficult to accept that some topics would be considered of moderate or low priority, even though such a designation did not probably indicate that the resource itself was of low priority, only that it might not benefit as greatly from a higher level of attention as might something else.

By the end of the two-day meeting the team had completed the first draft list of resource topics (Table 3-1) and priority levels, and incorporated this list into the NPS Ecological Framework (described below). For various reasons, this list changed somewhat over the course of the project. For example NRCA guidelines generally discourage including visitor and human use issues, so the group agreed to eliminate those topics. The final list (i.e. the Table of Contents for Chapter 4) is presented within the Ecological Framework format (Table 3-2).

NRCA funding for the Bandelier project was sufficient for the park staff to work cooperatively with an outside investigator to address one or two priority topics at a higher level of analysis than was possible without this fund-

Table 3-1. First draft list (July 2010) of all potential topics for the Bandelier National Monument Natural Resource Condition Assessment, herein sorted by priority. Items in bold were identified as being currently monitored or researched at some level.

Element	Management priority	Project priority	Data availability	Level of data summarization	Comments/ data availability
Elk impacts on woody vegetation (aspen)	high	high	high	medium	Multiple data sources are individually summarized; political implications; establishing reference conditions an important and non-trivial task; put BAND data in order first.
Historic spatial patterns of fire	high	high	high	medium	Complete record for 1932 thru 1996 exists, so data since 1996 need to be digitized and updated; spatial patterning of fire and insect outbreaks of interest; fire effects data could be incorporated.
Rio Grande corridor ecosystem	high	high	medium	low	Data are from multiple sources outside NPS, including USFWS and Corps; water holding episodes have resulted in sediment accretion, creating habitat for riparian vegetation, SW willow flycatchers, etc.
Large-scale ecosystem change	high	high	high	medium	Would qualitatively summarize large-scale ecosystem changes over past several decades, and assess changes in relation to natural vs. human-caused drivers, particularly climate; numerous independent lines of anecdotal or semi-quantitative information; find a cooperator to synthesize BAND and Jemez data (possibility to include Forest Service and Valles Caldera NP).
Erosion in pinon-juniper communities	high	medium	high	high	Data are only from a few sites; representative?
Invasive plants (cheatgrass)	high	medium	low	low	Not clear where data would come from: fire effects? I&M? Veg. map?
LANL contaminant impacts on park resources	medium	medium	high	medium	It will likely be difficult to get access to most of the data that would be needed to assess potential impacts to park.
Peregrine falcons	medium	medium	high	high	Summarize BAND population demographics and assess how they relate to statewide decline in productivity; lowest PEFA reproduction in 15 years.
Vegetation change	high	medium	high	high	This is currently being done in PJ; expand to other communities?
Water quality (DDT, cattle, NPS)	medium	medium	high	Low	Water quality impacts come from LANL, DDT site, cattle and NPS; this is important but may not be appropriate for this project.
Jemez Mountain salamander	medium	low	medium	medium	The species is a priority, but much work is being done elsewhere.
High elevation mammals (pikas, bushy-tailed woodrats, red-backed vole)	low	low	low	low	Ongoing research, data availability will increase in coming years.
Bighorn habitat	medium	low			Many outstanding questions regarding a potential re-introduction, so not sure how we would address this issue.
Non-native fish	low	low	high	high	Not a high project priority.
Mexican spotted owl	medium	low	high	high	Could be part of a multi-park assessment; birds have not been seen in the park for about 6 years; 4 occupied territories in the 1990s – Steve speculates that canyons are drier, less preferred as habitat.
Mountain meadows	medium	low	medium	medium	tree encroachment in montane meadows;

Table 3-1. (*continued*) First draft list (July 2010) of all potential topics for the Bandelier National Monument Natural Resource Condition Assessment, herein sorted by priority. Items in bold were identified as being currently monitored or researched at some level.

Element	Management priority	Project priority	Data availability	Level of data summarization	Comments/ data availability
Migratory birds (Grace's warbler)	medium	low	low	medium	This is a national species of conservation concern, but not enough data to consider here; 50% decline in Grace's warblers, a ponderosa pine species.
Wilderness values (viewshed/ night sky/soundscapes)	medium	low	low	low	A soundscape project will begin in FY11, and night skies are being assessed by I&M and NPS night sky program.
Scenic values (Jemez Mtn trail/scenic byway)	medium	low	low	low	Important, but not appropriate for this project.
Bats	low	low	low	low	Mexican freetail bat population – intermittent use of BAND caves would be more appropriate as a multi-park assessment.

ing. Because this opportunity was available, it was important to identify resource topics that were not only amenable to such an approach but that were also of high importance to the park. In the case of Bandelier NM, after developing the draft resource topic list, park staff determined that describing vegetation changes in response to large-scale impacts of wildfire, climate change and ungulate herbivory were their highest priorities for more extensive data analysis and synthesis.

After the meeting, several follow-up conference calls were held with the NRCA Ecologist, SCPN Program Manager, and Bandelier NM and USGS staff participating in the project to better define the resource topics and goals of the project.

3.2 Study design

3.2.1. Ecological framework, reference conditions, reporting areas

3.2.1.1 Ecological framework

The group incorporated the NPS Ecological Monitoring Framework (Fancy et al. 2009) to identify and then synthesize the natural resource topics, indicators, and measures that would be emphasized in the study. This framework was selected due to the tight integration of the framework with the NPS Inventory and Monitoring program from which much of the data used in the NRCA would originate. Further, an element of each assessment is the identification of data needed to better determine current conditions. If there were areas where data gaps seemed important, this information could potentially be incorporated more easily into future I&M program reviews if the topics were organized using this framework.

3.2.1.2 Reference conditions

Reference conditions were developed separately for each topic. Generally the process utilized to develop relevant reference conditions was to first conduct a literature search to determine what types of measures had been or were being used to evaluate similar resources. Discussions were usually then conducted with local knowledge experts, and existing NRCA documents examined to compare references conditions applied to similar resources in other NPS units. In some cases determining reference conditions

Table 3-2. Final list of selected topics organized within the NPS Ecological Monitoring Framework (Fancy et al. 2009). Changes are highlighted in bold with comments.

Level 1 Category	Level 2 Category	BAND Elements
Geology and Soils	Geomorphology	**Erosion in piñon-junipe**r - removed, included in Vegetation Change Assessment (Bowker and Smith)
Water	Water quality	Water quality
Biological Integrity	Invasive species	**Cheatgrass** - removed, separate topic, included in Vegetation Change Assessment Exotic fish **Elk and deer** removed, separate topic
	Infestations and disease	**Widespread vegetation mortality** due to beetle outbreaks; included in Vegetation Change Assessment **Bats/white-nosed syndrome** - removed as a topic for the NRCA
	Focal communities	Aspen Mountain meadows **Migratory landbirds** - priority increased in response to BAND Species of Concern effort **Bats** - removed as a topic for NRCA **Bighorn sheep habitat** - priority increased in response to BAND Species of Concern effort
	At-risk biota	**Jemez Mtn. salamander** - priority increased in response to BAND Species of Concern effort and fire **High-elevation mammals** - focused changed to picas only **Peregrine falcons** - included with raptors Mexican spotted owls **River otters** - added as part of BAND Species of Concern effort **Lynx** - added as part of BAND Species of Concern effort **Mountain lion** - added as part of BAND Species of Concern effort
Human Use	Point source human effects	**LANL contamination** - removed as separate topic, included as threat to water quality **Contamination dynamics** - removed as separate topic, included as threat to water quality
	Visitor and recreation use	**Wilderness values** - removed, not a natural resource
Landscapes and Ecosystem Processes	Fire and fuel dynamics	**Historic fire patterns** - included in larger topic of Fire History and Ecology
	Landscape dynamics	**Large-scale vegetation change** - ultimately included piñon-juniper, ponderosa pine, and mixed conifer vegetation types Rio Grande riparian ecosystem upstream of Cochiti Reservoir **Riparian zone stability** - included in Rito de Frijoles and Capulin Riparian
	Viewscape	**Scenic values** – Jemez Mountain Trail - removed, not a natural resource

was straightforward, for example if a recovery plan had been developed for an endangered species. Conversely, in many cases, particularly for complex ecological processes such as large-scale vegetation change, there currently are no quantified reference conditions available. The process for determining reference conditions (or reasons why they are unavailable or unquantified) is included within each topic section in Chapter 4.

3.2.1.3 Reporting areas

As the project developed we determined that the use of reporting areas would not enhance the project. The two primary influences on park resources and processes at present - catastrophic fire and climate change – are acting across all park systems and management areas.

3.2.2. General approach and methods

Specific elements were approached differently, but in general the NRCA Ecologist and Bandelier NM and USGS staff approached each element in the following manner. First, all NPS, USGS, or other relevant participants were asked to contribute their expertise. In addition the group communicated with any cooperators or researchers recommended by staff or identified from published or unpublished literature. If a resource had been identified during I&M scoping, all supporting documentation for that process was examined. Then a thorough literature search was conducted, first for the specific resource in the park, then for the resource or process in other regions, and then for any restoration, management, or research efforts that might provide information on methods incorporated to assess similar resources.

3.2.3 Components included in each analysis

Per the NPS NRCA guidelines, each assessment includes the following elements:

1. **Background**. This section describes the resource and generally why it was selected for inclusion in the project. It includes threatened or endangered status if appropriate, biological and ecological descriptions and contexts, relevance to the NPS mission, and relationship to specific park planning and management efforts. If known, threats to the resource or process are included in this section.
2. **Reference conditions**. The measures used to evaluate the condition of the resource are defined here. If no clear science-based measures appear to exist and alternate evaluation methods were utilized, those are also described here. The absence of any valid reference is noted here as well.
3. **Data and methods**. This section can include references to both existing data and methodologies evaluated as well as specific assessment methods incorporated for the NRCA.
4. **Resource condition and trend**. This section summarizes what is known about the resource in relation to the described reference conditions.
5. **Level of confidence**. In some cases very little is known about the status of the resource, the conditions that should be used to make the assessment, or both. This section evaluates the level of confidence the team had in making the assessment.
6. **Data gaps/Research needs/Management recommendations**. This section varies in length and scope. In some cases there are clear recommendations for further research or data that would be needed to have a high confidence in making an assessment. If the team had specific management recommendations to improve the state of the resource those may be included here as well.
7. **Sources of expertise**. Subject matter experts not identified elsewhere are listed here.
8. **Literature cited**. Each section is followed by a complete list of citations. In addition, as part of the final product, a database of all references included in the full document will be delivered to NPS.

3.2.4 Project challenges and changes

In the fall of 2011, oversight of the NCPN/SCPN NRCA projects by the NRCA Ecologist was suspended due to an unanticipated funding issue. In October of 2011, FY 2012 budgets were uncertain; neither the NRCA Program, the Intermountain Regional Office Resource Management Division, nor the Inventory and Monitoring Program were willing to take the risk of renewing the term NRCA Ecologist position until FY 2012 NRCA funding levels were established. Consequently, the term position expired. In the summer of 2012 funding for a similar coordinating position was established through a cooperative agreement with the Institute for Wildlife Studies. Through this agreement, Cathy Schwemm renewed her role as the coordinating NRCA Ecologist.

In the spring of 2011, Matt Bowker, with the U.S. Geological Survey Colorado Plateau Field Station, was identified as an appropriate cooperator to analyze and synthesize a broad array of vegetation data for for the park. Later that summer an Interagency Agreement was initiated to fund the data analysis and synthesis work. The following year, Matt Bowker accepted an assistant professorship at NAU. Because the original agreement was between NPS and USGS, it took over six months for USGS and NPS to find a solution to continuing Matt Bowker's involvement in the project. Ultimately, he was rehired as a USGS employee to complete his work on this project, but these administrative issues delayed his work.

The catastrophic Las Conchas Fire, ignited on June 26, 2011, ultimately burned over 60% of the monument. Both the fire and the subsequent floods and soil erosion that followed had immense impacts on park resources, many of which are described in sections of Chapter 4. The disturbances caused by these events not only altered the ecology of many park systems, but required all NPS and USGS personnel involved in the NRCA to divert their workloads dramatically toward the post-fire response. When the NRCA project re-started in the summer of 2012, the NRCA Ecologist and NPS and USGS staff reviewed the direction of the project in light of fire and flood impacts.

Midway through the NRCA process the Bandelier NM staff decided to increase the number of species of management concern that would be addressed in the NRCA report. This was in part because they were already engaged in a systematic review and assessment of these species. This added some additional work and time to the project but did not fundamentally change the focus or direction.

In the fall of 2014, with the news that Barbara Judy would be leaving her post at Bandelier, Lisa Thomas and Barbara Judy scheduled a close-out workshop on November 5th and 6th 2014. The Bandelier team, including Craig Allen and JMFS-USGS Pathways Intern, Collin Haffey, re-convened to review the remaining work for Chapter 4 and to complete Chapter 5 of the NRCA report. Lisa Thomas provided a final content review of Chapters 4 and 5 following completion of the BAND team's work.

3.3. Literature cited

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Chapter 4: Natural Resource Conditions

Chapter 4 is organized thus: (1) big-picture landscape-scale disturbance dynamics and parkwide vegetation change; (2) conditions in the major park ecosystems, starting with high elevation montane meadows and descending to the Rio Grande River and (3) condition of wildlife communities and species within Bandelier National Monument (NM).

The individual topics are also organized within the National Park Service (NPS) Ecological Monitoring Framework. Table 4-1 provides a roadmap to the Chapter 4 resource topics within this framework. Appendix D provides a more comprehensive list of species of management concern.

The Bandelier NM staff added an additional resource category to the topics covered in Chapter 4—Species of Management Interest—to cover species that are not directly managed by the monument, but have been of recurring interest to internal or external conservation partners in recent decades. These are species that historically may have occurred on or near Bandelier NM, but are probably not integral to park ecosystem function.

Table 4-1. Chapter 4 resource topics organized within the NPS Ecological Monitoring Framework.

Level 1	Level 2	Resource – Section
Air and Climate	Air quality	Air quality – 4.23
	Climate and climate change	Climate and climate change at Bandelier NM – Appendix A
Geology and Soils	Soil erosion	Erosion in piñon-juniper – 4.06-1
	Riparian geomorphology	Rio Grande Corridor and associated riparian vegetation – 4.09
		Rito del los Frijoles, Capulin Creek and associated riparian vegetation – 4.10
Water	Hydrology	Rio Grande Corridor and associated riparian vegetation– 4.09 Rito del los Frijoles, Capulin Creek and associated riparian vegetation – 4.10
	Water quality	Rito del los Frijoles, Capulin Creek and associated riparian vegetation – 4.11
Biological Integrity	Communities of concern	Montane grasslands – 4.03 Mixed conifer forest – 4.04 Ponderosa pine woodlands – 4.05 Piñon-juniper woodlands – 4.06 Rio Grande Corridor and associated riparian vegetation – 4.09 Rito del los Frijoles, Capulin Creek and associated riparian vegetation – 4.10 Native fish – 4.12 Landbirds – 4.13
	Species of management concern	American Pika - 4.18 Aspen – 4.04-1
	Species of management concern: At-risk biota	Rare plant species – 4.07 Mexican Spotted Owl – 4.14 Jemez Mountain Salamander – 4.15
	Species of management concern: Apex predators	Mountain lion – 4.16
	Species of management concern: Over-abundant species	Elk and deer – 4.17
	Invasive species	Nonnative plant species – 4.08
Ecosystem Pattern and Processes	Landscape dynamics	Fire history and ecology – 4.01 Parkwide vegetation change – 4.02
	Natural Sounds	Soundscape – 4.22

4.01 Fire History and Ecology

4.01.1. Description

Fire has played an essential role in shaping and maintaining the vegetation communities and landscapes in Bandelier NM and the Jemez Mountains. Many factors, including climatic conditions, a high occurrence of lightning strikes, availability of surface fuels and flammable vegetation, and topography make fire one of the dominant natural disturbance processes in this region. Most of the vegetation communities and wildlife that have persisted through time here have evolved under the influence of frequent fire. However, natural fire processes, to which vegetation communities and species in the region have adapted, have been altered in multiple ways through human actions and now climate change. The term 'fire regime' is used to describe attributes, such as the frequency, intensity, extent, season, and duration, of naturally occurring fires as they would typically burn in a particular vegetation community or landscape.

The role of fire as an ecosystem driver is discussed in the following four sections, in relation to landscape-scale vegetation change, and in relation to three specific ecosystems—piñon-juniper woodlands, ponderosa pine forests and mixed conifer forests.

Climate

The warm, semi-arid southwestern environment contributes to fire occurrence in Bandelier NM and the Jemez Mountains. In May and June, before the onset of the summer monsoonal rains, temperatures near 100° F are possible and humidity is often extremely low. Strong southwesterly winds are also a common occurrence in the spring, and precipitation totals during this period are often less than half an inch. When fires start in May or June fire extents can be relatively large (Snyderman and Allen 1997). Conversely, the number of fire ignitions is highest in summer months (July, August, and September) when thunderstorms are more frequent; 86% of historic fires recorded in Bandelier were the result of lightning strikes Figure 4-1a (Allen 1984). Fire extents in the summer are generally smaller than in the spring because fuel moisture during these months tends to be higher as a result of monsoon storms Figure 4-1b.

The El Niño-Southern Oscillation (ENSO) phenomenon also has significant effects on precipitation and fire occurrence in Bandelier NM and the Jemez Mountains. ENSO involves fluctuating ocean temperatures: the El Niño or warm phase, where ocean temperatures are warmer than normal, causes enhanced precipitation across the southern U.S., while during the La Niña or cool phase, less precipitation falls (NOAA 2013). Higher precipitation levels stimulate the production of fuels, such as grasses, forbs, and shrubs, allowing these fuels to become more abundant and continuous. In the subsequent La Niña years, these fuels lose moisture and become available for fire ignition and spread (Touchan et al. 1995). Accordingly, major fire years in the Southwest were historically associated with drought conditions that followed periods of precipitation (Touchan et al. 1996).

Numerous sources reveal a long history of frequent low intensity surface fires that burned through continuous herbaceous understories to create relatively open woodlands and savannas in this region, as well as the occurrence of infrequent mixed-severity or stand-replacing fires. While Ancestral Puebloans certainly used fire and likely reinforced the frequent, low intensity fire regime, abundant lightning ignitions appear sufficient to explain most prehistoric fire patterns (Allen 2002a).

The pre-settlement forest structure and fire regime in the Bandelier NM area and the Jemez Mountains was maintained until about the 1880s, when railroads that linked New Mexico with other parts of the U.S. brought millions of sheep and cattle (Allen 2002b). The sheep and cattle overgrazed the land, removing the herbaceous vegetation and fine fuels that had previously carried surface fire.

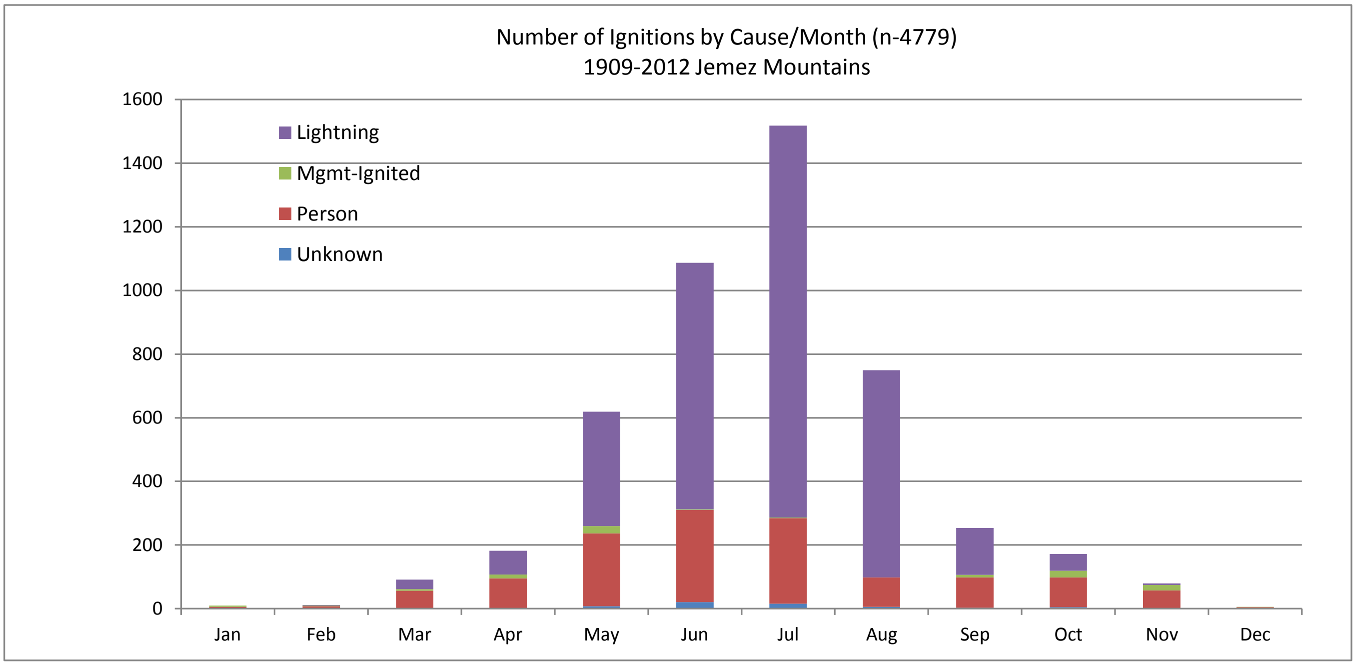

Figure 4-1a. Number of ignitions per month, by type, in the Jemez Mountains, New Mexico, from 1909–2012. Note the large number of ignitions occurring in June and July due to pre-monsoonal dry lightning storms.

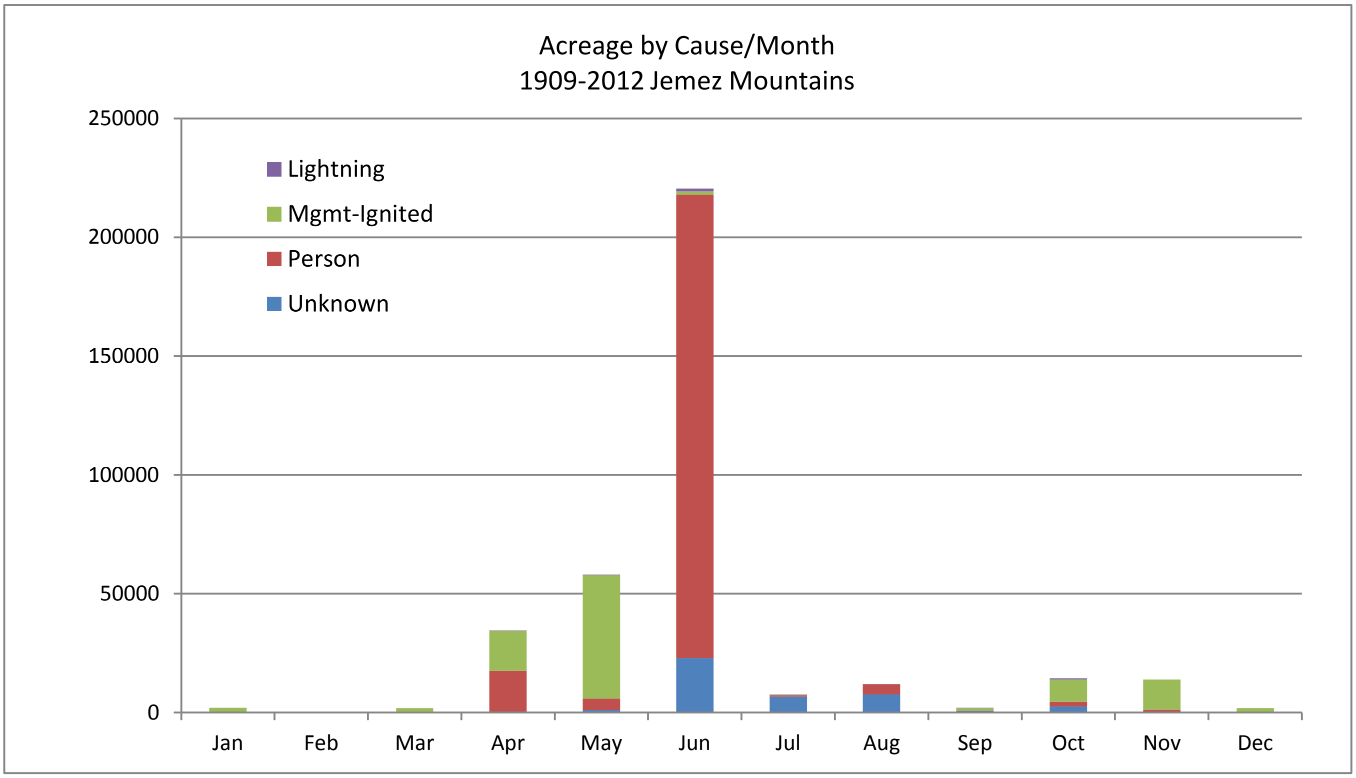

Figure 4-1b. Cumulative acres burned per month, by type of ignition, in the Jemez Mountains, New Mexico, from 1909–2012. Notable large fires burning May and June account for the majority of acres burned. Natural ignitions account for relatively few acres burned, in large part due to successful fire suppression. Human-caused ignitions that occur during seasonally dry conditions when natural ignitions are rare can often result in fires that exhibit unnaturally extreme fire behavior and severity.

In addition, widespread and effective fire suppression efforts began in the early 1900s, resulting in the near cessation of fire (Figure 4-2a).

Over the last century or so, the forest structures and fire regimes that once existed have been greatly altered, producing significant ecological effects on the fire prone-landscape (Potter and Foxx 1978; Allen 1989). Most notably, increased tree densities have resulted in 'crown fires' which can travel in the canopy across the forest at high speed and intensity, in stark contrast to the low intensity surface fires of the past. Fires of this type are now becoming more frequent in ponderosa pine forests, where large stand replacing fire events were once anomalous. The large amounts of surface fuels and compacted litter and duff can burn with high intensity and severity, causing a longer residence time (the amount of time fire burns in a location), and increasing the potential for lethal fire temperatures and deleterious effects on soil properties and vegetation adapted to less intense fires.

The absence of frequent, low intensity surface fire has altered and degraded the forests in Bandelier NM and the Jemez Mountains in many ways. The full range of tree age classes that existed historically has been replaced by extremely high densities of saplings and and mid-story trees. The high tree density also renders the forests more susceptible to disease and insect infestations, causing widespread mortality in trees. The forests no longer have an open canopy structure, limiting light penetration to the forest floor and causing herbaceous plant cover productivity and diversity to decrease. Forest fuels, such as branches, twigs, pine cones, and dead vegetation, have been accumulating to extremely high levels.

In recent decades, a number of unusually large and intense fires have burned across the the monument landscape. The La Mesa Fire of 1977 burned 14,250 acres; the Dome Fire of 1996 burned 16,500 acres; the Cerro Grande Fire in 2000 burned 43,000 acres; and the Las Conchas Fire in 2011 burned 156,593 acres, becoming the second largest fire in New Mexico history (Figure 4-2b).

4.01.2 Reference conditions

Fire-scar chronologies show frequent and widespread fire in the forests of the Jemez Mountains before the 1890s (Allen 2002a). Fire activity (perhaps including multiple ignition points) in Bandelier NM and the Jemez Mountains commonly occurred over extensive areas in some drought years, such that networks of fire-scarred trees record many years where fires burned widely across watersheds in the monument (Allen 1989, 1994, 2007), throughout most of the Jemez Mountains (Allen et al. 1998, Allen 2002a), and even across most mountain ranges in the Southwest (Swetnam et al. 1999).

The fire season was generally in the spring and summer, when climatic and fuel conditions were conducive to fire ignition: fall fires were uncommon (Allen 2002a). Fire duration varied widely, depending on variables such as precipitation, slope, aspect, and elevation. In persistent drought conditions, some fires in ponderosa pine forests could burn for months.

Fire-scar chronologies indicate significant spatial variation in past fire regimes (Allen 2004) which can be summarized for the 3 major forest/woodland ecosystems.

Mixed conifer forest

Prior to the 20th century, the mixed conifer forest experienced frequent fires with a fire return interval of approximately 4–9.7 years. Most fires were low intensity surface fires, but small, patchy crown fires also occurred in mesic mixed conifer forest sites outside Bandelier NM(Touchan et al. 1996).

Ponderosa pine forest/woodland

In the 400 years prior to the 20th century, ponderosa pine forests experienced frequent fires, most of which were low-intensity surface fires (Allen et al. 1995; Touchan et al. 1996). The fire return interval during this period was approximately 5–15 years. Fire frequency tended to be reduced at low-elevation sites and in sites that are topographi-

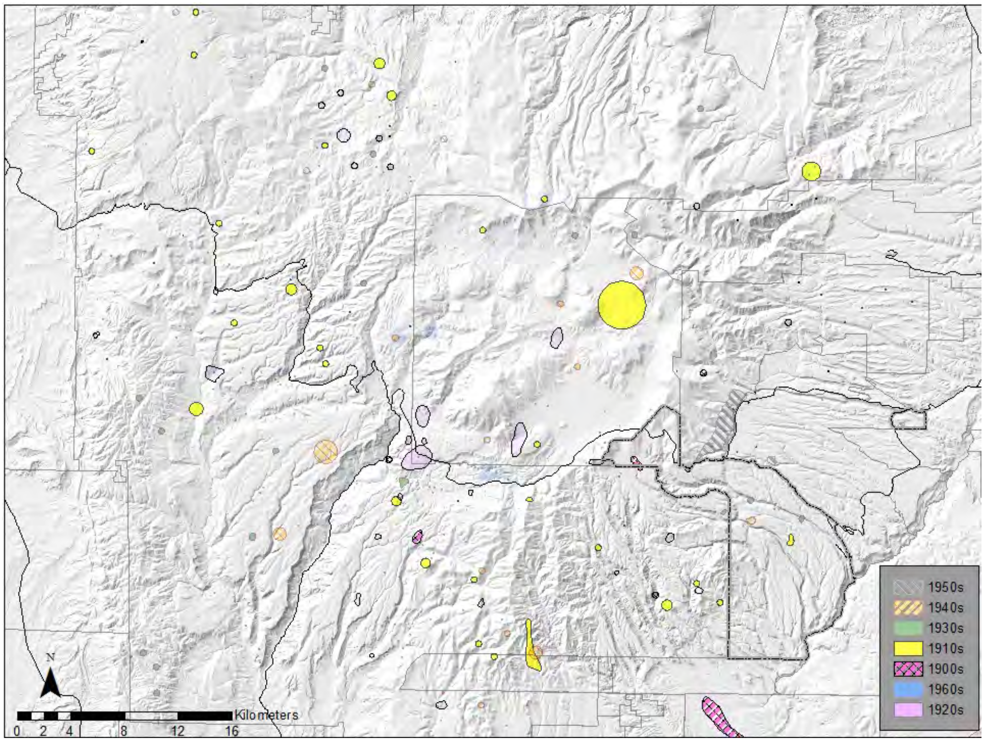

Figure 4-2a. Fires that occurred in the Jemez Mountains, New Mexico, by decade, from the 1900s to the 1960s. Bandelier NM boundary is in the lower right section. A period of high grazing numbers and successful fire suppression occurred from 1909–1969, and very few acres burned during this period. The circles represent the relative area of only those fires that had acreage records.

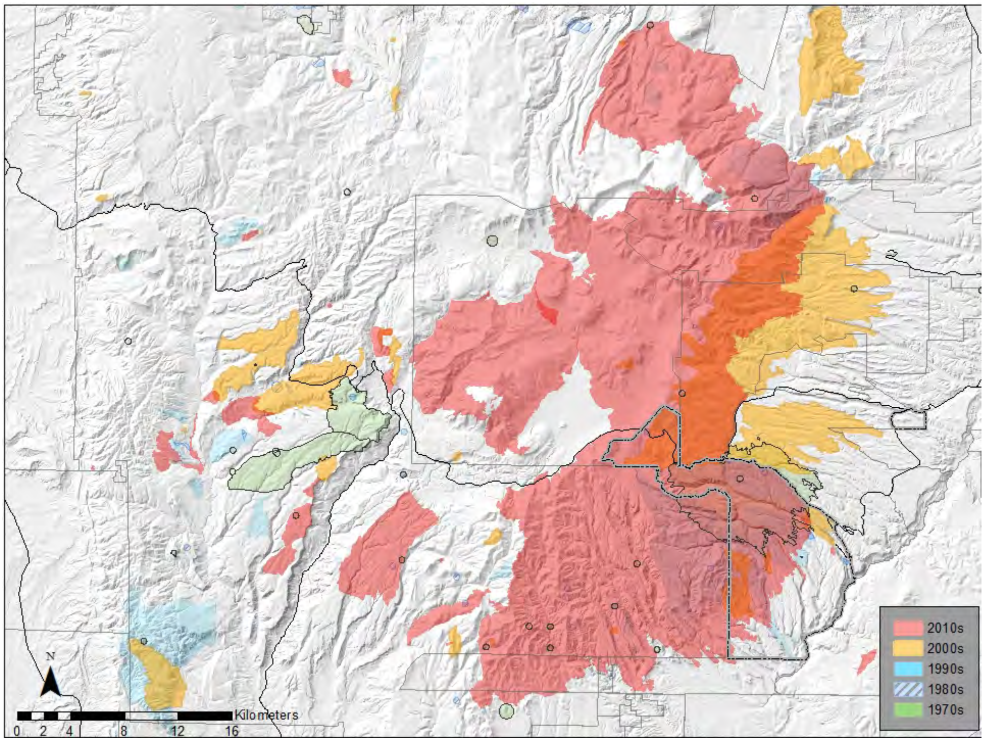

Figure 4-2b. Fires that occurred in the Jemez Mountains, New Mexico, by decade, from the 1970s to the 2010s. Bandelier is in the lower right section. After decades of effective fire exclusion, forest fuels became overly dense which resulted in a modern fire history of large fires with unprecedented severity both in size and degree. Large portions of the landscape, particularly around Bandelier NM, have effectively been reset. Understanding the trajectory of these post-fire landscapes is a high research priority for the monument.

cally isolated from the larger ponderosa pine forest matrix.

Piñon-juniper woodland

In contrast to our current knowledge of past fire regimes in the ponderosa and mixed conifer forests, the historical fire regime of the piñon-juniper woodland is not completely known and not without controversy (Bowker and Smith 2014). The fire history is difficult to know because piñyon pine and juniper trees are poor preservers of fire scars from which to reconstruct past fire cycles. Understanding the role of fire in the piñon-juniper woodlands is further complicated by the diversity of piñon -juniper community types that occur across the monument landscape. These include piñon-juniper savannas, piñon-juniper-oak communities, and most commonly, persistent piñon-juniper woodlands. At higher elevations, piñon-juniper woodland intersperses with, and, in some sites, has recently replaced ponderosa pine savanna. While some literature suggests frequent surface fire (Gottfried et al. 1995; Jacobs and Gatewood 1999), evidence is sparse and mostly from ponderosa pine stands near piñon-juniper stands (Bowker and Smith 2014). Patchy stand-destroying crown-fires are well-documented in piñon-juniper woodlands from elsewhere in the region (Romme et al. 2009). Pre-1900 disturbance regimes in piñon-juniper savanna are not well understood.

4.01.3 Data and methods

Numerous methods have been used to investigate historic fire conditions. Information gathered from packrat middens and pollen deposits support the concept of a long history of fire on the landscape by providing evidence that ponderosa pine (*Pinus ponderosa*), a fire evolved species, has existed in the southwestern U.S. region for approximately 8,000 to 11,000 years (Anderson 1989). Charcoal deposits provide evidence of continuous fire activity extending back at least 9,000 years (Brunner-Jass 1999, Allen et al. 2008). Core samples taken from bogs contain material that is approximately 9,000 to 15,000 years old, revealing a long history of high frequency fire activity for the past 9,000 years with an unusual gap in fire activity after 1880 (Allen et al. 2008).

Dendrochronological (tree-ring) analysis of fire scarred wood shows that, for many centuries prior to 1900, fires burned frequently and were widespread in these landscapes (Allen 2002a). Researchers have dated over one thousand sampled trees, logs, and stumps from more than fifty locations in the monument and the Jemez Mountains, thereby developing well replicated information regarding fire occurrence patterns, fire seasonality, and precipitation as far back as 1600 A.D (Allen 1989, Touchan et al. 1996, Morino et al.1998, Allen 2002a, Falk 2004, Margolis et al. 2007, Dewar 2011). For example, fire scar samples from multiple ponderosa pine sites in the Jemez Mountains recorded 1,858 fire events and 221 different fire years over a 400-year period, with an average fire return interval (or average number of years between fire events) ranging from 5-16 years (Touchan et al. 1996). Fire scar samples collected in ponderosa pine forests in Bandelier NM documented 113 separate fire years between 1480 and 1899, with widespread fires occurring every 5-15 years (Allen et al. 1995).

Historic records and journals also support the notion that natural fire was extensive in the Southwest. For example in 1875, Joseph Rothrock, a botanist with the expedition called the Wheeler Survey (predecessor of the U.S. Geological Survey), wrote of the region just south of Gallup, New Mexico: "… at an altitude of about 8,000 ft. above the sea, was a fine, open, park-like region with a large growth of yellow pine (*Pinus ponderosa*) and fir covering the hillsides. A diversified herbaceous vegetation was out in the most brilliant colors, beautifying alike the woods and open grounds. … Good forage was abundant" (Rothrock 1875).

Finally, aerial and ground based photography of landscape conditions in the monument and the Jemez Mountains demonstrate that fire was ubiquitous and played a major role in shaping and maintaining vegetation communities throughout the area. Repeat photography studies (Allen 1984 & 1989, Hogan and Allen 2000) show the vegetative and forest conditions as they existed under the influence of frequent fire, and also illustrate the striking expansion of local forests into previously open environments as well as the densification of some forests stands in the absence of periodic fire.

4.01.4 Condition and trend

Recent warm and dry climatic conditions are important drivers of increased forest drought stress and more high severity fire activity in the Southwest (Williams et al. 2010, 2013), including Bandelier NM and the Jemez Mountains.

In general, disruption of fire regimes during the 20th century has left many western forests susceptible to uncharacteristic wildfire. However, re-establishment of forest structure and fire regimes that predominated prior to European settlement can restore ecosystem function (Hurteau, Bradford, Fulé, Taylor, & Martin, 2013), mitigate destructive crown fire (Allen et al., 2002; Covington et al., 1997) and convey greater resiliency in the face of climate warming (Stephens et al., 2013). In most forests characterized by high-frequency low-severity fire regimes, ecological restoration entails mechanical thinning, followed by reintroduction of low-intensity fires that consume overabundant fuels, speed nutrient cycling, and encourage the reestablishment of a grass-dominated herbaceous understory, with multiple benefits to biodiversity. Wildfire alone may result in a forest structure that is subsequently more amenable to the reestablishment of historical fire regimes, with its concomitant benefits to ecosystem structure and function. However, recurrent fire also may have detrimental impacts on high-frequency low-severity fire regime ecosystems. In extreme cases, a single high-severity burn may cause near complete tree mortality over large areas, such as was the case in portions of the 2000 Cerro Grande (Allen et al. 2002) and 1996 Dome (USDI 1996) fires. Subsequent reburning of such areas can further eliminate seed trees and soil seed banks, reduce soil organic matter, encourage post-fire establishment of shrub and grass communities, and lead to long-term conversion of forest to non-forested ecosystem types—exemplified by extensive reburning by the 2011 Las Conchas Fire (Allen, 2012). Thus, the impacts of multiple, recurrent fires may have strikingly different long-term effects on the future of southwestern U.S. forests, depending on the influences of multiple factors, including initial forest composition and structural characteristics, fire size and intensity, fire return intervals, site characteristics, and climate.

Combinations of fire and drought are hypothesized to interact non-linearly, potentially pushing currently forested systems beyond thresholds that, once crossed, exceed the capacity of some ecosystems to recover (Allen, 2007; Williams et al. 2013), resulting in conversion to novel vegetative communities. Once across such resiliency thresholds, additional fires may drive positive feedback loops, perpetuating novel communities and establishing new fire regimes (Parks et al. 2013; Thompson et al. 2007; van Wagtendonk et al. 2012). Research that addresses the new fire regime characteristics and resiliency of emerging novel ecosystems is important as managers in fire-prone areas already face questions about appropriate burn frequencies and their influences on sustaining or encouraging desired ecosystem characteristics, as ongoing climate change makes restoration of previous forest conditions less likely.

4.01.5 Level of confidence

High level of confidence for history; moderate-to-low level of confidence for future projections.

4.01.6 Data gaps/Research needs/ Management recommendations

It will be important to understand the patterns and drivers that sustain more resilient forests following multiple fires. Resilience in forests of the Southwest is often defined from the perspective of single species or groups of species, and rarely assessed at the landscape level. We define landscape resilience in forests once dominated by surface fire (e.g., ponderosa pine) as resistance to widespread crown fire that allows for the incremental adaptation of understory and overstory vegetation to warming climate conditions. Fire has been a key tool for returning ecosystems to a resilient and adaptive state.

We provide the following recommendations:

- Continue with planned fuels treatments and restoration efforts of extant forests.

- Monitor disturbed landscapes to identify indications of ecosystem trajectory (such as changed fire regime and successional patterns).Identify areas where subsequent fires may have detrimental effects on landscape trajectories and develop a wildfire management strategy that aims to reduce fire impact to those areas. Likewise, identify areas where subsequent fires would have beneficial effects on ecosystem processes and develop strategies to allow, or encourage, wildfire into these areas.
- Consider experimental management techniques to alter trajectories based on information from monitoring and research.

4.01.7 Sources of Expertise

Craig Allen, Matthew Bowker, Terelene Foxx, Laura Trader, and Tom Swetnam

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4.02 Parkwide Vegetation Change

Introduction

Bandelier NM is rather unique for a monument of its size in that several decades of vegetation data are available, some of which is from long-term repeatedly measured studies. The park is also located adjacent to a National Laboratory with substantial research infrastructure, which has produced additional environmental observations over the same multi-decadal timespan. Thus, in many ways, the monument has already been serving as a barometer of change. During the initial project scoping for this Natural Resource Condition Assessment, park staff identified synthesis and analysis of a substantial body of vegetation data, collected by park staff and collaborating researchers for a variety of purposes, as an important area of focus for the NRCA.

In 2011, NPS entered into an Interagency Agreement with Matt Bowker, U.S. Geological Survey, Southwest Biological Science Center to conduct a synthesis and analysis of Bandelier NM vegetation data. With a focus on 3 major ecosystems within the park (piñon-juniper woodlands, ponderosa pine forests, and mixed conifer / aspen forests) the project's objectives were to

1. evaluate the direction of landscape-scale vegetation change that has occurred in recent decades
2. identify the primary drivers of observed change
3. suggest, to the extent possible, the likely direction of future change.

In 2014, Matt Bowker, assisted by David Smith, and in consultation with Craig Allen, Brian Jacobs, Stephen Fettig, Laura Trader, and Jim DeCoster, completed a report of his findings. *Assessment of Vegetation Change in Bandelier National Monument, a "Barometer of Change" in the National Park System* is a first attempt at compiling several of the most information rich datasets available for Bandelier NM and surroundings, and synthesizing these to assess the condition and trajectory of the park's major ecosystems, and possibly the future trajectory of southwestern woodlands and forests. The Bowker and Smith report forms the basis for this section, and for the sections on mixed conifer forests (4.04), ponderosa pine forests (4.05), and piñon-juniper woodlands (4.06), covering the major park ecosystems. A portion of their analysis was directed specifically toward aspen recovery following fire in relation to elk herbivory—those results are included in section 4.04-1 Aspen. Their full report is provided in Appendix B of this document.

4.02.1 Description

Bandelier NM encompasses vegetation communities ranging from semi-arid woodlands to subalpine forests and meadows (Muldavin et al. 2011). In total, 55 distinct plant communities are mapped in the park, which can be roughly divided into three major zones based upon dominant plant communities: mixed conifer / aspen forests, ponderosa pine forests, and piñon-juniper woodlands.

Vegetation changes can be simultaneously rapid and occur over a large spatial extent. A severe drought occurred in the 1950s across much of southwest North America, causing widespread tree mortality (Betancourt et al. 1993, Marshall 1957). The low elevation ponderosa pine forests within Bandelier NM died during the drought, resulting in a 2 km upslope retreat of ponderosa pine (Allen and Breshears 1998). The retreat of the pines released piñon-juniper woodlands from competition and those species co-dominated the low elevation mesa tops until the early 2000s. At the turn of the century a drought, driven by warmer than average temperatures, decimated the piñon population by killing over ninety percent of the mature piñon pine (Breshears et al., 2005). The low elevation mesa tops are now dominated by drought resistant one-seed juniper, and although piñon pine seedlings are present across much of the pre-drought range, few ponderosa seedlings are found in the pre-1950 range of the species.

4.02.1.1 Major drivers of change in Bandelier National Monument

At the scale of the entire monument, there are four tightly-linked phenomena responsible for most of the change that has occurred in recent decades: 1) climate change, 2) fire, 3) drought/insect and disease outbreaks, and 4) past land use. Other factors that strongly affect particular ecosystems, such as elk herbivory or persistent accelerated erosion, are discussed in later sections.

Before discussing these drivers of change individually, it is worth viewing them as a linked complex of the following interacting change agents (Figure 4-3; cf. Allen 2007):

1. Climate change, a pervasive force, modifies soil moisture and vapor pressure deficit, which leads to plant drought mortality (Breshears et al. 2005, Williams et al. 2013).
2. Climate change modifies distributions of some bark beetles (Bentz et al. 2010).
3. Warmer and drier conditions favor fire (Williams et al. 2010).
4. Drought stress leads to greater susceptibility of some tree species to bark beetle attack (Wallin et al. 2004).
5. Both drought- and beetle-killed plants contribute to fuel load.
6. Greater fuel load creates larger and more intense fires.
7. Accumulated fine fuel load has been strongly reduced by past grazing, and fire suppression activities, which results in fire exclusion and buildup of woody fuels (Allen 2004).

Climate change. Climate change influences vegetation both directly and indirectly. Recently, Williams et al. (2013) developed a forest drought-stress index, using a comprehensive tree-ring data set representing the past one thousand years. They found the index to be equally influenced by warm season vapor pressure deficit and cold season precipitation (together explaining 82% of the forest drought-stress index). These two parameters are likely to change in the near future. All available models converge on a

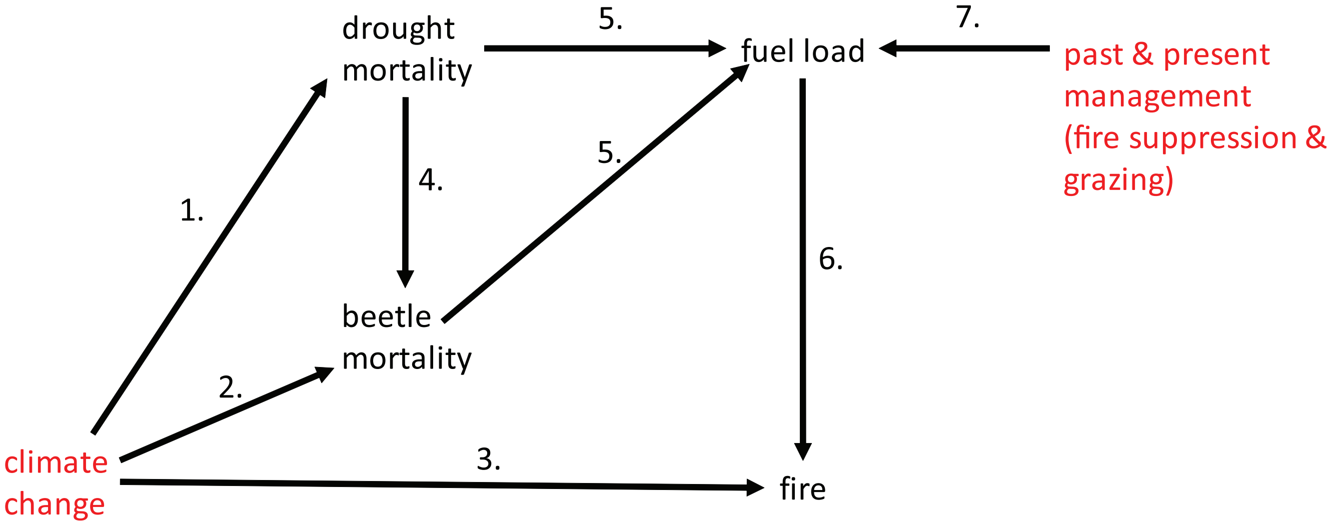

Figure 4-3. Linked drivers of vegetation change in Bandelier National Monument and the southwestern U.S. Climate change and past and present management have and continue to exert a variety of direct and indirect effects on forests and woodlands. See text for explanation.

scenario where northern New Mexico will experience warming (Karl et al. 2009). Precipitation projections are much less certain and likely to vary among models, although reduced spring precipitation is predicted by a consensus model (Karl et al. 2009). Both of these changes can be expected to influence soil moisture and therefore drought mortality directly, and indirectly affect fire susceptibility (Figure 4-3). Williams et al. (2010, 2013) project that by 2050, forest drought stress values will routinely be equal to or more extreme than mega-droughts seen in the 1200s and the 1500s. This suggests that areas currently occupied by Southwestern forests will, to a large degree, no longer be forest habitat. Breshears et al. (2011) argue that such climate change impacts on vegetation communities may not be gradual, incremental and homogenous (e.g. a small mortality rate every year). Rather, climate change impacts on ecosystems may be "big, fast, and patchy". Climate change-linked drought and fire have already caused high mortality in 18% of southwestern forests (Williams et al. 2010). Such impacts give managers and stakeholders few options and little time to respond to climate change-driven vegetation change, such as the mass tree mortality events now being observed globally (Allen et al. 2010).

Fire suppression and subsequent large-scale crown fire. Fire is a part of the evolutionary environment of most of the forest, woodland and savanna types within Bandelier NM (Allen 1989, Moore et al. 1999, Touchan et al. 1996). It is well demonstrated that two phenomena— livestock reduction of fine fuels, and fire suppression—coincide with a shift to less frequent fire cycles (Touchan et al. 1996). We consider fire here as a stressor and a driver of vegetation change because the scale of recent fires has been unusual (Figure 4-4). The 1977 La Mesa Fire burned 6354 ha in 1977. The 1996 Dome fire burned 6677 ha, and was followed shortly by the 1997 Lummis Fire which burned another 865 ha. The Cerro Grande fire of 2000 burned 16,909 ha. Most recently, the Las Conchas fire burned 63,250 ha in 2011. These fires are occurring at higher frequency and intensity and at increasing spatial scale. Fire severities varied within these fires, but each fire had areas of significant stand replacement.

Drought and insect outbreaks. Periodically, severe droughts may be associated with pulse mortality of overstory trees (Allen 1998; Breshears et al. 2005, 2011). One such drought occurred in 2002. It caused major mortality to *Pinus ponderosa* and *P. edulis*, especially in lower elevations, but also impacted *Pseudotsuga menziesii* and *Abies concolor* at higher elevations (Muldavin et al. 2011). Primarily mature trees were affected rather than saplings. This extreme drought was not the driest on record, but was the most impactful in centuries. The severity of die-back events has been linked to the fact that warming climate interacts with drought to induce mass mortality. This was hypothesized by Breshears et al. (2005), and experimentally tested by Adams et al. (2009) using *Pinus edulis*. Using 50 years of tree ring data from multiple species along an elevation gradient in Arizona, Adams and Kolb (2005) found that growth rates were related to both water availability and temperature, and that species in the hotter and drier parts of their range were more sensitive to these influences (i.e. their growth tracked climate more closely than conspecifics from higher sites). Insects, such as bark beetles and the fungi they disperse, cause mortality of water stressed trees because water stress can inhibit defense mechanisms, such as production of resins (Wallin et al. 2004, Negron et al. 2009). Our data cannot distinguish between temporally congruent drought mortality episodes and bark beetle outbreaks; thus we treat them as a tightly-linked stressor complex.

4.02.2 Reference conditions

At the scale of the entire monument we can make three broad generalizations about historical reference conditions. First, most of the ecosystems have a tree canopy component. It is not thought that large shrublands and grasslands without trees were abundant on the landscape. Second, in most ecosys-

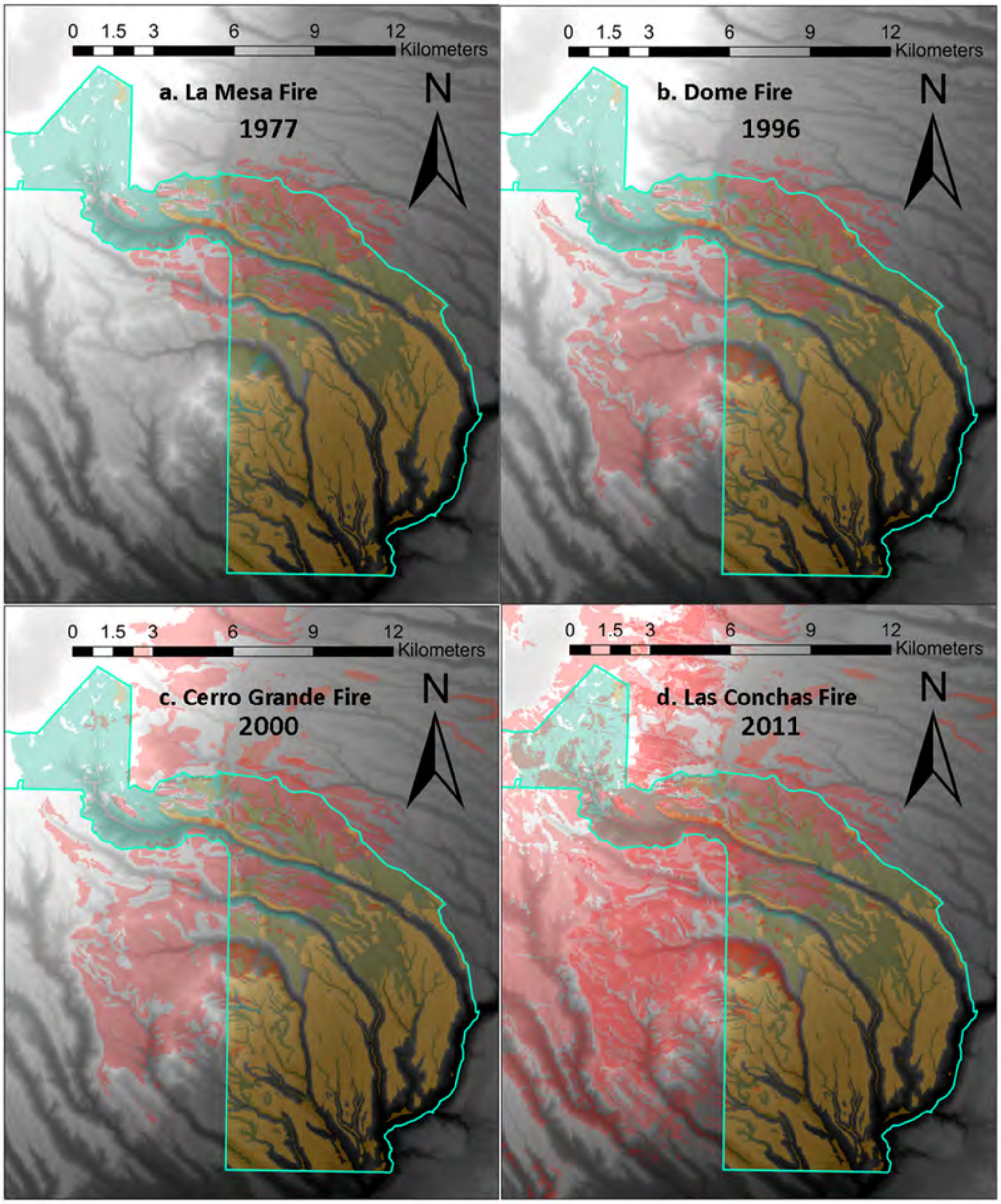

Figure 4-4. Cumulative footprint of medium and high severity wildfires in the distributions of the mixed conifer-aspen forest zone (blue green), ponderosa pine forest zone (drab green), and piñon-juniper woodland zone (orange), since 1977. Light red shades indicate a high or medium severity burn. Darker red shades indicate areas where high or medium severity burns have occurred twice.

tems it is thought that woody canopy cover increased through the 20th century, and under a more natural disturbance regime, forests would be less dense. Nearly a century of fire suppression has allowed many forests and woodland stands to attain unusually thick canopies, sometimes at the expense of understory herbaceous biomass. Finally, although fine scale shifting mosaics of different forest and woodland types would be expected, based upon normal disturbance and succession cycles, there should be no major directional shifts in the distribution of major vegetation types. At the scale of the entire monument, a vegetation trajectory toward an overall decrease in tree canopy cover, without loss of a tree component from the various ecosystems, and without a directional shift in the broad distribution patterns of the major ecosystem types, would approach reference conditions.

4.02.3 Objectives of vegetation analysis, data sources, and methods

Two vegetation maps have been produced in recent decades, encompassing most or all of Bandelier NM. Allen (1989, hereafter `Allen') produced a map (resolution 1.5 ha) based upon aerial imagery from 1981. Muldavin et al. (2011, hereafter `Muldavin et al.') produced a map using supervised automated digital processing, assisted by multispectral imaging techniques at a resolution of 0.5 ha. This map was based upon 2004 imagery.

The two maps used a different system of map units. Allen developed a system that consisted of 43 forest and woodland types and 23 additional patch types, identified primarily by the dominant tree component with additional understory information. Muldavin et al. collected a series of ground plot surveys which they analyzed using multivariate statistical methods and classified into a system consistent with the guidelines of the National Vegetation Classification Standard that identified 95 plant associations in Bandelier NM. To facilitate comparisons among the two maps, both were reduced to their

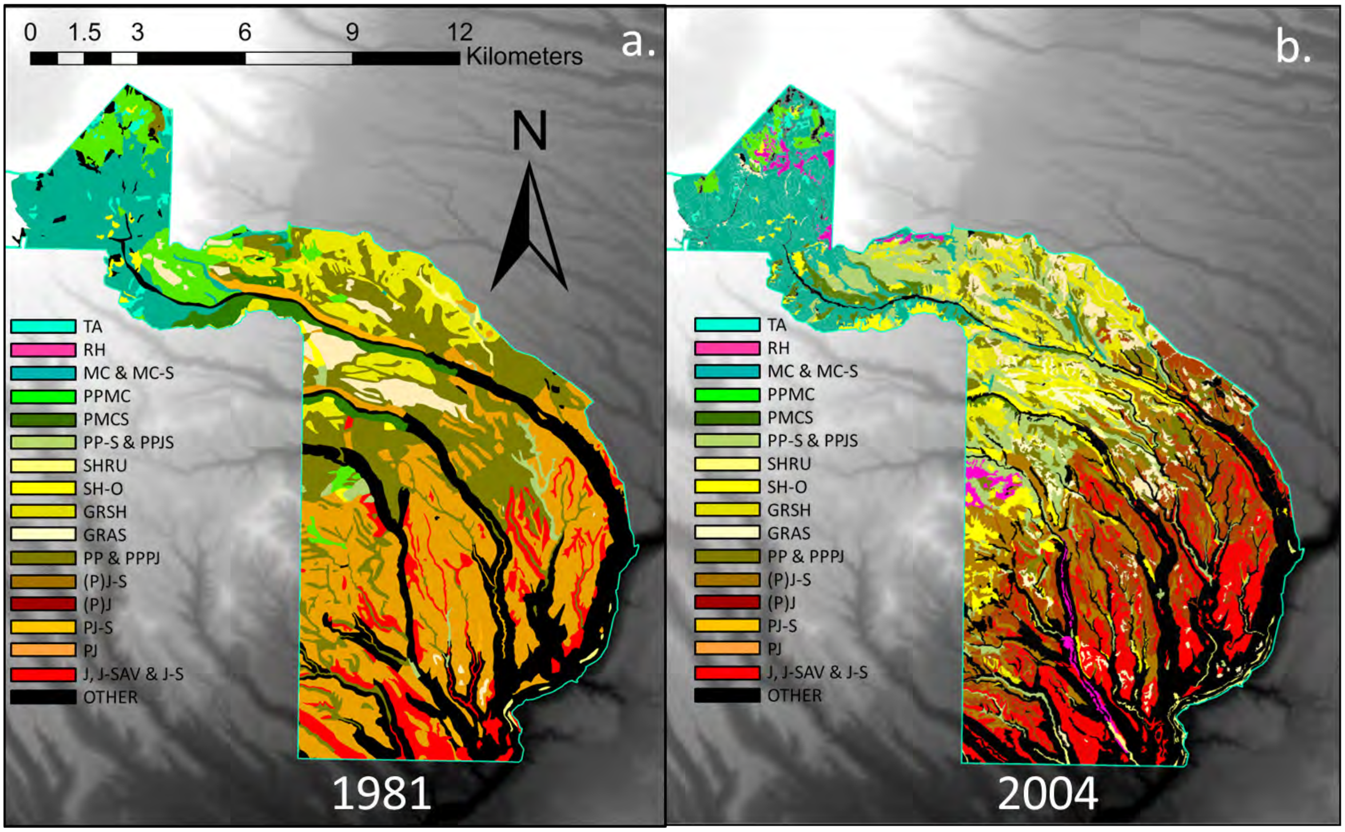

Figure 4-5. Change in spatial distribution of major vegetation map units from 1981–2004. a.1981, modified from Allen (1989). b. 2004, modified from Muldavin et al. (2011).

LEGEND

TA - Quaking Aspen Woodland/Forest
RH – Ruderal Herbaceous
MC – Mixed conifer Woodland/Forest
MC-S – Mixed conifer-Shrubland
PPMC – Ponderosa Pine/Mixed Conifer Woodland
PMCS – Ponderosa Pine/Mixed Conifer Shrubland
PP-S & PPJS – Ponderosa Pine Shrubland & Ponderosa Pine /Juniper Shrubland
SHRU -Shrubland
SH-O – Oak Shrubland
GRSH – Grassland/Shrubland
GRAS - Grassland
PP & PPPJ – Ponderosa Pine & Ponderosa Pine/Piñon-Juniper Woodland
(P)J-S – Juniper Shrubland (previously with piñon)
(P)J – Juniper Woodland (previously with piñon)
PJ-S –Piñon-Juniper Shrubland
PJ – Piñon-Juniper Woodland
J, J-SAV & J-S - Juniper Woodland, Juniper Savanna & Juniper Shrubland

intersection, and the Muldavin et al. map was reclassified to conform to the Allen map (complete details of this process are provided in Bowker and Smith 2014). Although this translation was imperfect, most major Muldavin et al. units corresponded reasonably well to major Allen units. Comparisons of the Allen and Muldavin et al. vegetation maps are reported here and in the 3 following sections.

In addition to the spatial comparison of the maps, two point datasets associated with each of the maps, respectively, were used to estimate where and to what degree canopy cover has changed since the 1980s. The Allen data were analyzed in three ways to estimate canopy cover during the 1980s: 1) using the existing map attribute estimating canopy cover at the spatial scale of polygons; 2) applying an inverse distance weighting interpolation technique (Franke 1982) to the point data; and 3) by averaging the two methods. The three analyses of the Allen data were compared with the Muldavin et al. data to quantify cover change from the 1980s to post-2003; positive values indicate canopy loss, and negative values indicate canopy gain.

4.02.4 Condition and trend

4.02.4.1 Map analysis

Vegetation has changed drastically in 23 years between the 1981 and 2004 imagery (Figures 4-5 and 4-6). At the lowest elevations of the monument, it appears that woodlands and savannas, with *Juniperus* as the canopy dominant, are expanding. This pattern appears independent of any changes to other species, such as *Pinus edulis*. Further upslope, the most striking difference in vegetative dominance to have occurred is the complete loss of mature *P. edulis* as a co-dominant species due to drought and beetle mortality. This change occurred in association with the 2002 drought. These former *P. edulis*-dominated or co-dominated stands have shifted to *Juniperus* woodlands via subtraction of *P. edulis* rather than migration of *Juniperus*. Although *P. edulis* is retained mostly as saplings, it can no longer be said to be a co-dominant. Further upslope, there is a clear shrinkage and fragmentation of stands with *Pinus ponderosa* as the dominant, or co-dominant with *P. edulis*. Grass-dominated stands expanded at the apparent expense of *P. ponderosa* coverage.

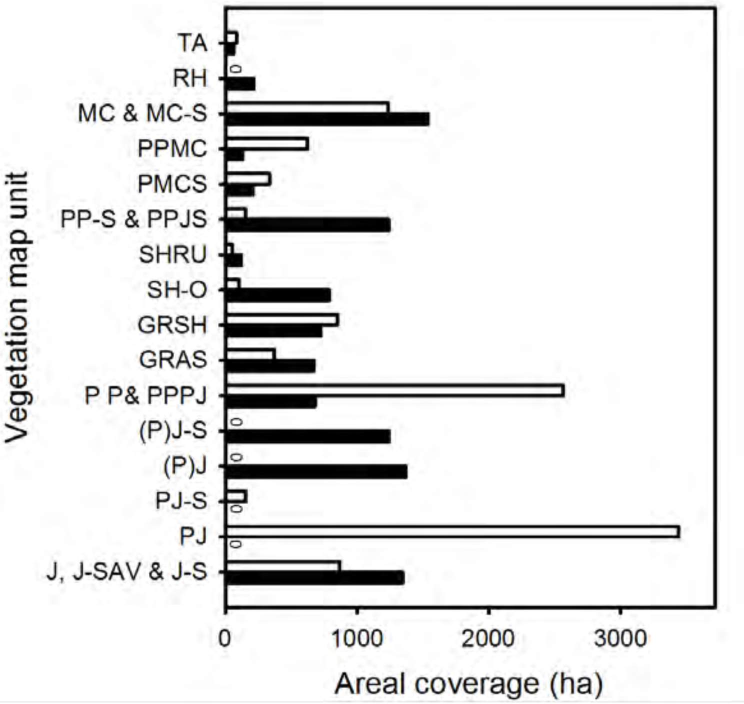

Figure 4-6. Change in total area of coverage of major vegetation map units from 1981 (white bars) – 2004 (black bars). See Figure 4-5 on page 48 for definition of vegetation codes..

Many *P. ponderosa* stands also seem to be gaining a shrubby understory. Across Bandelier, but particularly in the middle elevations where *P. ponderosa* was dominant, shrubs are becoming more prevalent, and the coverage wherein shrubs are a major component has more than tripled (Figure 4-7). It is difficult to ascribe such a large pattern to differences in vegetation classification or mapping. Allen (1989) was very cognizant of the distinctiveness of communities with and without shrub components in the 1981 imagery, devoting several map units to making this distinction. Compared to *P. ponderosa*, growth-ring widths of the shrub *Quercus gambelii* are less sensitive to interannual changes in drought severity (Adams and Kolb 2005), providing one possible reason for shrub proliferation

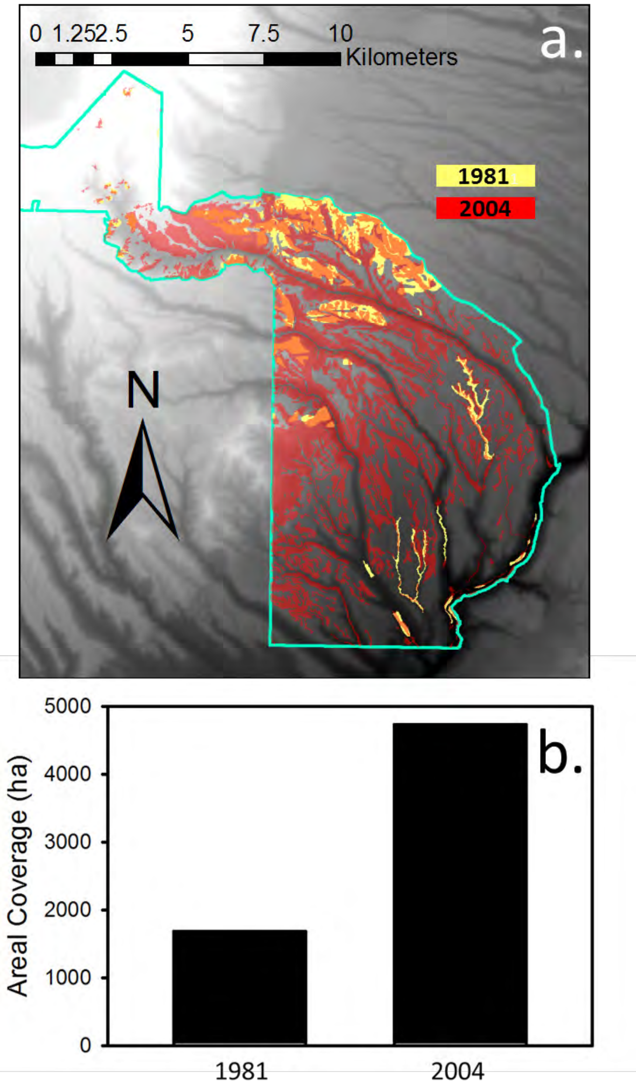

Figure 4-7. Expansion of plant communities with a shrub component (J-S, PJ-S, SH-O, SHRU, PP-S, PPJS, PMCS, MC-S) from 1981–2004. a. Spatial distribution of shrubs as a major community component. b. Areal coverage of shrubs as a major community component. See Figure 4-5 on page 48 for definition of vegetation codes..

in the middle elevations of the monument. Another major reason might pertain to the ability of *Q. gambelii* to resprout after fire (Abella 2008, Strom & Fule 2007). Some portions of the monument have experienced repeated fires in the last few decades.

At the highest elevations the spatial distribution of mixed conifer-dominated stands has declined by probably less than one third (Figures 4-5 and 4-6). (However, changes resulting from the recent Las Conchas Fire are not captured in this analysis.) Mixed conifer stands with *P. ponderosa* as a component may be increasing, as are shrubs overall. Finally, it is difficult to draw conclusions about *Populus tremuloides* stands; they appear to represent a shifting mosaic, as would be expected for a species associated with disturbance, but do not strongly change in coverage on the landscape from the earlier map to the later one.

4.02.4.2 Canopy cover analysis

The analysis of canopy cover change supports the conclusions discussed above. Averaged over the whole extent of our analysis, the monument appears to have lost between 14 –15% canopy cover. The lower elevation mesa tops, almost without exception, indicate some degree of canopy loss, generally about 30%, but some areas were even greater. These areas were dominated by piñon-juniper woodlands and ecotones of this ecosystem.

The area within the boundary of the La Mesa Fire (at a higher elevation) is experiencing canopy gain, usually less than 20%, indicating that there is some slow recovery of forest canopy after about 30 years. Still higher in elevation in the mixed conifer/aspen forests, the overall pattern is one of very heterogeneous net loss. Near complete loss of canopy is estimated for the upper portions of the canyon, and further north in areas which were burned by the Cerro Grande Fire as well as outside the fire perimeter.

4.02.4.3 Synthesis

There are several clear directional changes in the spatial distribution of focal vegetation communities. At higher elevations, *Pinus edulis* has declined dramatically, leaving these areas dominated by juniper. The lower boundary of *P. ponderosa* forests has shifted upslope and these forests have become increasingly fragmented, colonized by shrubs, or otherwise altered. *P. ponderosa*-mixed conifer stands appear to have been replaced by *P. ponderosa*-shrub communities, but otherwise mixed conifer forests have not strongly shifted in their spatial location.

The loss of distinctive overstory species, directional shifts of some major ecosystem types, proliferation of former subdominants, and drastic or outright loss of some dominant species on the landscape indicate that the ecosystems of the monument are largely well outside of desired conditions. Given that these recent vegetation shifts are directly or indirectly driven by climate change, it is likely that similar patchy, big, fast transformations will continue to occur on the monument landscape. The role of climate change in driving vegetation changes across the landscape is discussed in detail in Bowker and Smith 2014.

4.02.5 Level of confidence

There are inherent difficulties in making comparisons between two maps that were produced by different researchers using different techniques. While some error can be attributed to methods, (and many of these are discussed in detail in the appendices), confidence in the assessment of the broadest and strongest patterns is high.

4.02.6 Data gaps/Research needs/ Management recommendations

Bowker and Smith suggest management of fuel conditions within the remaining stands that have not been burned intensely may yet discourage similar events from occurring.

4.02.7 Sources of expertise

Matt Bowker, Dave Smith and Craig Allen

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4.03 Montane Grasslands

4.03.1 Description

Montane grasslands are found in a distinctive landscape pattern on upper, south-facing slopes (5–40%) with moderate to high solar exposure on nearly all of the larger summits and ridge crests in the Jemez Mountains above 9500 ft (2900 m) feet, including the slopes of Cerro Grande within Bandelier NM (Allen 1984 & 1989; Coop and Givnish 2007a) (Figure 8).

Primarily grasslands, these communities are dominated by the large bunchgrasses Thurber fescue (*Festuca thurberi*) and oatgrass (*Danthonia parryi*) (Allen 1984; Muldavin et al. 2011). Kentucky bluegrass (*Poa pratensis*) dominates some patches within these grasslands and may represent the legacy of historic livestock grazing (Moir 1967). Along with the dominant grasses, forbs can be common and diverse. Soils of montane grasslands are relatively deep mollisols with significant eolian inputs. These soils are well-developed, indicating the long-term presence of grassland vegetation on these sites and possible feedback mechanisms (Allen 1984; Coop and Givnish 2007).

The topographic situations of montane grasslands are consistent with the idea that fires were important to their long-term maintenance, and confirmed by fire-scar records of high frequency surface fires pre-1900 from old trees adjoining these grasslands (Allen 1989, Dewar 2012). While the climatic conditions of montane grasslands are clearly suitable for conifer establishment and growth, natural fires at relatively high frequencies, but low intensities, acted to rejuvenate existing meadows by burning pedestaled grass clumps and killing conifer regeneration (Allen 1984 & 1989; Coop and Givnish 2008; Dewar 2012).

Figure 4-8. Montane meadow. NPS photo.

4.03.1.1 Threats

The loss of montane meadows and grasslands (Figure 4-9) reduces biodiversity and other ecosystem services provided by grassland and herbaceous systems (Patton and Judd 1970, Kremer et al. 2014), so the observed occurrence of woody species encroachment and associated loss of herbaceous open spaces is of concern (Dyer and Moffett 1999, Moore and Huffman 2004, Vankat 2013, Kremer et al. 2014). Trees are common in areas adjacent to these grasslands, so the decrease in grassland/forb cover at high elevations indicates the increasing effects of factors that promote woody species persistence (Kremer et al. 2014).

In particular, prior to 1900, historic high fire frequencies and vigorous grass competition acted to maintain largely treeless grasslands (Allen 1984 & 1989). In the Jemez Mountains there is abundant documentation that, with fire suppression since 1900, conifers have invaded sites, creating high montane savannas where there used to be open grasslands and meadows (Allen 1984 & 1989; Swetnam et al. 1999; Coop and Givnish 2007a, Dewar 2012). Encroaching tree overstories suppress grasses and forbs and eventually displace them as tree densities increase, needle litter accumulates, and mature canopies close (Allen 1984). Recent large-scale wildfires have reversed some historic encroachment, killing trees primarily along the forested grassland boundaries. However, mechanical reductions are still required to remove well-established, large diameter trees scattered at lower densities in grassland interiors.

Overgrazing by many sheep from the 1880s to about 1920 caused failure of the previous surface fire regime, yet precluded conifer establishment (Allen 1984, 1989). Subsequent reductions in sheep numbers apparently allowed conifers to establish in degraded grasslands where appropriate seed sources existed. Moist spring weather may have promoted pulses of tree establishment. Increasing temperatures may also be supporting increased rates of tree establishment (Dyer and Moffett 1999, Coop and Givnish 2007b).

Because the persistence of montane grasslands is largely dependent on fairly frequent but low intensity fires, fire condition is extremely relevant to determining the condition of these systems. The current fire regime of the Jemez Mountains is very different from what it was pre-settlement, and has diverged even further in the last several decades. This topic is discussed at length in other sections of this report.

Geographic patterns of montane grassland occurrence in the Jemez Mountains and specifically in Bandelier NM was best documented through aerial photographs (Allen 1984 & 1989), though these only captured changes since 1935, well after grazing and fire suppression had begun.

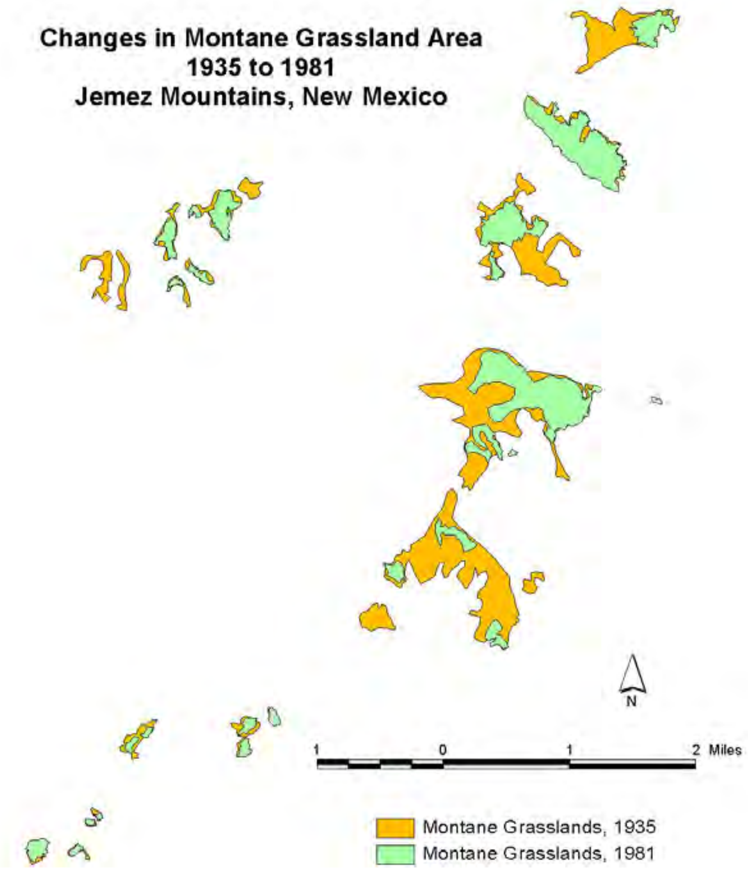

Figure 4-9. Changes in montane grassland area, 1935–1981, Jemez Mountains, New Mexico.

4.03.2 Reference conditions

Many higher elevation south-facing mountain slopes were dominated by montane meadows prior to the effects of fire suppression and historical grazing.

4.03.3 Data and methods

In his Masters thesis, Allen used historic photo analysis, tree cores, and soil analysis to examine historic conditions of montane grasslands (Allen, 1984). Allen (1989) identified grassland and meadow vegetation types using GIS and fieldwork.

Montane grasslands were also identified by Muldavin et al. (2011). Allen compared vegetation changes (including montane grasses) from 1935–1981, however no rigorous comparison has been made between Allen’s delineations and that of Muldavin et al. (2011).

In fall of 1983 twelve permanently-marked photopoints were established in the upper portions of the Cerro Grande montane grassland to visually document changes (4–6 photos/pt, including the cardinal directions). These have been visited periodically and rephotographed (on file with Bandelier Ecology Group).

4.03.4 Condition and trend

Change analysis using historic air photos clearly reveals recent conifer tree invasion of Jemez montane grasslands, with an 85% decrease in open grassland area on Cerro Grande, from 110 ha in 1935 to 17 ha in 1981 (Allen 1989). Overall, in the southeastern portion of the Jemez Mountains, open montane grassland area decreased 55%—from 554 ha in 1935 to 250 ha in 1981. Several small montane grasslands present in 1935 have disappeared, while the larger grasslands have been fragmented (Allen 1989).

Encroachment continued through the mid-1990s until the Cerro Grande (2000) and Las Conchas (2011) fires removed additional trees (personal communications with Brian Jacobs and Craig Allen), although neither of these fires killed many mature invasive trees in the interior of the meadows. The fires also rejuvenated existing grasses by burning pedestaled clumps, decadent with dead growth, and promoting forb growth, though mechanical reductions are still required to remove well established large diameter trees scattered at lower densities in meadow interiors (Brian Jacobs, personal communication, Halpern et al. 2012). Ironically most of the trees that have invasively infilled the ancient montane grassland have survived the recent fires, while the adjoining upper slope old-growth forests were completely killed by the Cerro Grande and especially Las Conchas fires. For several years, from the mid-1980s to the early 1990s, there was sporadic management cutting to harvest trees (mostly small) that were expanding into upper slope montane grasslands on Cerro Grande. Management fires have also been utilized during the past several decades to remove mature invading trees, which has had limited but important effects on preserving grassland areas (Craig Allen, personal communication, NPS unpublished GIS data).

4.03.5 Level of confidence

The level of confidence for this analysis is high for changes occurring up to the present.

4.03.6 Data gaps/Research needs/ Management recommendations

A proposal to girdle larger trees and leave them standing to fall naturally in coming years was suggested as a minimally invasive approach to incrementally address the encroachment issue without the negative aesthetic impacts of cutting, or environmental concerns associated with herbicides. This would partially mitigate heavy fuel loads which could sterilize soils locally when burned. Prescribed fire could be used in concert with these practices to maintain herbaceous open spaces within mostly forested systems.

An NAU graduate student is inventorying montane meadows using Southern Colorado Plateau Network vegetation monitoring methods. Results will be available in 2016.

4.03.7 Sources of expertise

Craig Allen, Esteban Muldavin, Division Leader, New Mexico Heritage Program.

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4.04 Mixed Conifer / Aspen Forest

This is a brief summary of the information provided by Bowker and Smith (2014) in Appendix B. The data used, analysis methods, and all references are included therein. The information below is mostly interpreted or taken directly from that report.

4.04.1 Description

The Mixed Conifer / Aspen vegetation type is a mosaic of distinct forest types that differ in both overstory and understory components. Within this group, forest stands, including significant amounts of aspen (P*opulus tremuloides*), are of particular interest for both their aesthetic and ecological value. Aspen stands will be discussed separately in section 4.04-1.

Muldavin et al. (2011) described five mixed conifer alliances within Bandelier NM. On north aspects at the highest elevations, dense Engelmann spruce forests occur in the Jemez Mountains, but occupy minimal area within the boundaries of the park, particularly after recent high-severity fires. In the elevation zone between 2130 and 2750 m, forests are co-dominated by mid-elevation conifers (Douglas-fir, white fir, blue spruce, ponderosa pine, southwestern white pine, limber pine). Muldavin et al. describe two distinct mixed conifer forest groups within this zone that reflect an inherent moisture gradient. Which tree species dominate a site is a function of site conditions and disturbance history. For example, white fir and blue spruce (shade tolerant but fire intolerant) dominate mesic sites. In contrast, Douglas-fir (less shade tolerant, more fire resistant and drought tolerant) is more common on dry sites with the potential for higher fire frequency (Muldavin et al. 2011).

Over the last 125 years, the structure and composition of local mixed conifer forests have changed dramatically (Allen 1984, 1989, 2002). Fire suppression in the 20th century allowed the development of dense sapling understories in many mixed conifer forests, with tree regeneration dominated by Douglas-fir and white fir, and also aspen and ponderosa pine in some areas.

4.04.2 Reference conditions

Mixed conifer forest includes a mosaic of forest types in various stages of disturbance. Old-growth stands tend to be uneven-aged and contain multiple species and age classes, with an open structure and patchy understory.

On south-facing sites in Bandelier NM, such as much of the headwaters area of the Rito de los Frijoles, dry mixed conifer forests generally experienced frequent, low-intensity surface fires, with a mean fire interval of about 10 years before 1900, and crown fires rare or absent (Allen 1989). In mesic mixed conifer forest—largely absent on Bandelier but found elsewhere in northern New Mexico forests – past fire regimes included a mixed-severity fire regime with a combination of surface fires and patchy crown fires of low frequency, which generally corresponded with periods of drought (Touchan et al. 1996; Margolis et al. 2007; Margolis & Swetnam, 2013). Mesic mixed conifer crown fire gaps would be expected to become aspen stands if the species was locally present prior to fire. With post-1900 suppression of surface fires, both xeric and mesic mixed conifer forest understories in-filled with high densities of tree regeneration, particularly aspen and pine species initially, in the then-open forest stands, but later, increasingly dominated by shade-tolerant Douglas-fir and white fir regeneration as stands densified. With the long-term absence of fire, overstory aspen stands have gradually in-filled and been overtopped by conifer species, until fire or other disturbances allow aspen regeneration again.

4.04.3 Data and methods

Numerous methods and data sets were included in Bowker and Smith's analysis. A vegetation map based upon 1981 imagery was used to delineate the former location of various types of mixed conifer forest and aspen stands (Allen 1990). Available maps of fire severity were overlayed on the 1981 distribution of Mixed Conifer / Aspen forests to illustrate the cumulative footprint of moderate to high severity fire since 1977.

Two vegetation maps, Allen (1989) and Muldavin et al. (2011)), were compared to detect changes in vegetation cover over time. See section 4.02.3 (p. 11) for an explanation of the two different maps and how they were compared. See also section 4.04-1.4 Condition and trend for analysis of elk exclosure and refuge plot data to determine the relative impacts of browsing on aspen growth and regeneration. See Bowker and Smith (2014) for detailed descriptions of data and methods.

4.04.4 Condition and trend

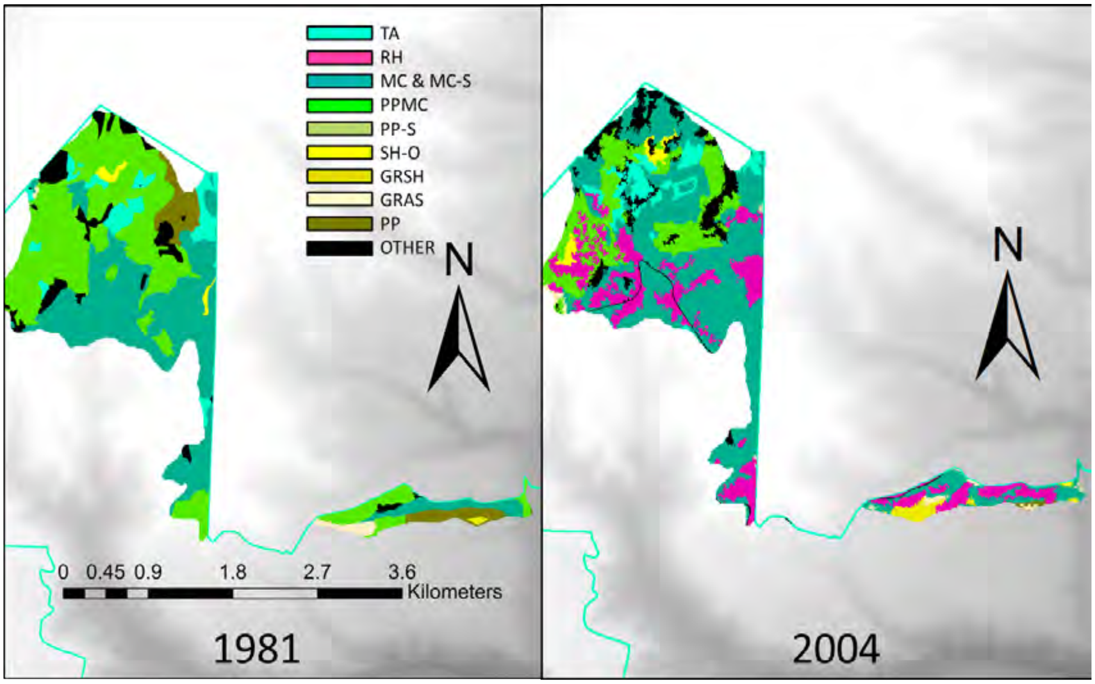
Vegetation change, specifically in the mixed conifer zone of Bandelier NM, is largely driven by fire, herbivory, and drought and drought-associated insect and disease outbreaks. After about a century of fire suppression and relatively few fires, large and often intense fires have been strongly impacting the mixed conifer/aspen forests of the park. The current millennium has been characterized by "mega-fires" in the region. The 1977 La Mesa Fire, 1996 Dome Fire, and 2001 Cerro Grande Fire all burned mixed conifer/aspen forests within Bandelier NM (Figure 4-4, p. 47). Each of these fires burned moderately to intensely in patches, opening canopies in patches up to hundreds of hectares in size. The scale of these openings was reasonably close to historical conditions of the mixed conifer/aspen forests within the park. The Las Conchas Fire was much larger and followed only 10 years after Cerro Grande. Virtually the entire southern half of the 1981 distribution of Mixed Conifer / Aspen forests was impacted by Las Conchas (Figure 4-4). The scale of this event is clearly outside of historical conditions. The cumulative outcome of these fires is that more than half of the 1981 distribution of Mixed Conifer / Aspen has experienced at least moderate severity fire.

It is too early for substantial data to be available to track the long-term response of the recent Las Conchas Fire in mixed conifer forests. However it is possible to observe change among major cover types from 1981–2004 within the Cerro Grande Fire perimeter. About 40% of areas mapped as mixed conifer by Allen (1989), retained this cover type following the Cerro Grande Fire, and approximately 25% best matched a ponderosa pine/mixed conifer/shrub community, perhaps signaling a transition to fire- adapted species. One notable fire disturbance-linked community type that does not appear to be increasing is aspen. After the Cerro Grande Fire, ruderal herbaceous communities comprised about 25% of the former area of communities mapped as mixed conifer within the burn perimeter. Spatially, the overall trends are the apparent decline in total coverage of ponderosa pine/ mixed conifer communities, the increase in ruderal herbaceous vegetation, and a decline in contiguous patches of similar community types (Figure 4-10).

However, most impacts of the Cerro Grande Fire within Bandelier NM were actually low intensity, so we cannot attribute all vegetation change within this time period to the fire, which occurred in 2000. Another important event was the 2002–2004 drought-induced pulse of tree mortality which affected many white fir and Douglas-fir trees within portions of Bandelier's mixed conifer forests, though less dramatically than the extreme mortality of piñon in the park's lower-elevation woodlands.

Finally, saplings and suckers of aspen are extremely palatable to elk, and following fire, aspen recovery can be suppressed and even nearly eliminated due to elk herbivory (Kay 1990, Kay 1997, Kay and Bartos 2000, Wooley et al. 2008).

The recent post-fire trajectories of mixed conifer or aspen forests in Bandelier do not seem to be toward proliferation of aspen clones. Aspen is a difficult community type to map unless it is a pure stand. Relatively pure stands could be expected to increase



LEGEND

TA - Quaking Aspen Woodland/Forest
RH – Ruderal Herbaceous
MC – Mixed conifer Woodland/Forest
MC-S – Mixed conifer-Shrubland
PPMC – Ponderosa Pine/Mixed Conifer Woodland
PP-S & PPJS – Ponderosa Pine Shrubland
SH-O – Oak Shrubland
GRSH – Grassland/Shrubland
GRAS - Grassland
PP & PPPJ – Ponderosa Pine

Figure 4-10. Vegetation communities within the Cerro Grand Fire perimeter as mapped based on 1981 imagery (Allen 1990) and 2004 imagery (Muldavin 2011).

after stand-opening fires, which in small patches likely allowed pre-1900 recruitment of some current mature aspen groves. However, other current aspen groves in Bandelier reflect early 1900s infilling into glades, small openings, and canopy gaps in formerly much more open mixed conifer forests due to post-1900 suppression of historic surface fires, just as associated conifer species regeneration similarly infilled these stands. Between the 1981 and 2004 maps, aspens were mapped as neither gaining nor losing total area within the area of the 2000 Cerro Grande Fire; rather they show a shifting mosaic on the landscape, gaining area in some places while losing a similar amount in others. As discussed in the separate section on “Aspen”, there is evidence that browsing herbivory on aspen by ungulates (elk and deer) is suppressing recent post-fire aspen regeneration in portions of Bandelier’s mixed conifer forests. One clear change between the two maps is the new ruderal herbaceous communities, which would be expected in a fire-disturbed environment for this ecosystem type. According to Muldavin et al. (2011) these are dominated by Pascopyrum smithii along with Bouteloua spp. and a high diversity of forbs. We cannot yet determine if these ruderal patches will eventually be colonized by trees (aspen or conifers), as these patches are still young and there are insufficient data available currently to determine longer-term trends. It also appears as though a finer-scaled heterogeneity is emerging (Figure 10), although this observation may be influenced by a slightly smaller grain size used in the Muldavin (2011) map compared to the Allen (1989) map.

4.04.5 Level of confidence

The level of confidence for this analysis is moderate to high.

4.04.6 Data gaps/Research needs/ Management recommendations

Research needs are to update the park vegetation map to reflect post-Las Conchas Fire vegetation, monitor post-disturbance ecosystem recovery, and investigate potential fire management actions to contribute to ecosystem resilience. These are all discussed in more detail in chapter 5.

4.04.7 Sources of expertise

Matthew Bowker and David Smith

4.04.8 Literature cited

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4.04-1 Aspen

Portions of this section are a summary of information provided by Bowker and Smith (2014). Their full report is provided in Appendix B.

4.04-1.1 Description

Quaking Aspen (*Populus tremuloides*), is the most widely distributed native tree in North America and the only aspen species in western North America (DeByle and Winokur 1985, Jones 1985, Worrall 2013). Vegetation communities dominated by aspen have numerous ecological and human values including high biological diversity (Chong et al. 2001), means by which they can indicate greater ecological health of a region (White et al. 1998), importance as wildlife habitat, ability to yield high-quality water in greater volume than other forest types in similar settings, ability to function as fire mediators in highly flammable coniferous forests (Fechner and Barrows 1976; DeByle and Winokur 1985), and the generally high value that humans place on the visual contribution of aspen stands in western landscapes.

The reproductive biology of aspen is of biological interest and greatly affects their current ability to persist. In the West, aspen does not typically regenerate from seed; aspen seeds are very sensitive to drying and need to stay wet for extended periods of time as well as in contact with mineral soil during early seedling growth. Intermittent dry conditions, or insufficient contact between the roots and mineral soil lead to seedling death (Quinn and Wu 2001; Turner et al. 2003). However, abundant aspen regeneration by seed can occur, particularly after fires when mineral soils are widely exposed (e.g., Turner et al. 2003).

In contrast, aspen vigorously reproduce vegetatively from existing clones (Barnes 1996). A clone is specifically defined as a group of individual stems and the associated root system that has been vegetatively propagated from a single seedling, called an ‘ortet’. Many living aspen clones may be ancient, for example it has been proposed that most extant aspen clones in western North America established by 10,000 years ago (Mitton and Grant 1996), and the long-lived root systems of these clones make aspen one of the oldest living life forms on earth (Mitton and Grant 1996).

There is a long history of concern about browsing on aspen across the western United States (Sampson 1919; Leopold et al. 1947; Beetle 1962; Basile 1979; Mueggler 1988; Mueggler 1989; Krebill 1972; Murie 1994; Mueggler and Bartos 1997; Kay and Bartos 2000; Ripple and Larsen 2001), including in national parks (Wright and Thompson 1935; McLaren and Peterson 1994; White et al. 1998; Baker et al. 1997; Ripple and Larsen 2000).

Historic rise and recent declines of aspen

Aspen regeneration and succession are closely linked with historical and current fire regimes. Thus the story of recent aspen declines needs to be prefaced with an understanding of historical aspen expansions. Based on extensive fire-scar studies (Allen 1989), over at least the last 400 years, the dry mixed conifer and ponderosa pine forests of Bandelier NM were characterized by frequent surface fires (approximately 10-year mean fire intervals) with less common, patchy mixed-severity fires. With the advent of fire suppression by 1900, aspen began to expand due to release from this fire regime. Over the last century or so, the forest structures and fire regimes that once existed have been greatly altered, producing significant ecological effects on the fire prone-landscape (Foxx and Potter 1978; Allen 1989). Most notably, increased tree densities have resulted in ‘crown fires”, which can travel in the canopy across the forest at high speed and intensity, in stark contrast to the low intensity surface fires of the past. The patterns of aspen forest structure that emerged by the late 1900s were just as altered by fire suppression as the rest of the mixed conifer forests, with infilling of gaps by aspen sprouts that no longer were killed or thinned by frequent landscape-scale fires.

Although it is the most widespread tree species in North America (Worral et al. 2013), *Populus tremuloides* is generally declining in the western U.S. (Kay 1990, Kay 1997, Kulakowski et al. 2013, Schier 1975), with trends varying from place to place. Declines may be caused by a variety of factors. First, without fire, *Populus tremuloides* generally cannot compete with conifers. In some ecosystems, such as spruce-fir forest or high elevation, mesic mixed conifer forest, where long fire intervals allow conifers to close canopies and build high fuel loads, fire suppression throughout the 20th century largely omitted the fire-caused openings that typically provide opportunities for *Populus tremuloides* clones to proliferate. However, this sort of fire regime characterizes very little of the Monument landscape, as its high elevation forests are southerly in aspect and were connected as part of high-frequency firesheds pre-1900 (Allen 1989, Touchan et al. 1996,Morino et al. 1998). Secondly, herbivore populations in the Southwest have seen a substantial increase in the last century due to predator suppression, increase in ground water availability in the form of cattle tanks (Binkley et al. 2006), and re-introduction efforts (Truett 1996). Now when fires occur, herbivory can constrain the growth of new ramets (Kay and Bartos 2000, Binkley et al.2006). Sheppard and Fairweather (1994) estimate some 10-15 years of protection from herbivores might be necessary to allow an aspen ramet to attain a sufficient size to make it safe from herbivory. Finally, in recent years, the so-called “Sudden Aspen Decline” has led to mass mortality of mature *Populus* clones. Recently researchers have asserted the cause to be the combination of severe drought and warm temperatures (Anderegg et al. 2013, Worral et al. 2013). While this is a concern regionally, thus far in Bandelier NM there are no such cases of unexplained death of mature aspens (Stephen Fettig, personal communication), though it remains a possibility in the future.

History of elk and aspen in or near Bandelier National Monument

Elk were not abundant before the 20th century. Despite, or because they were relatively easy for Native Americans to hunt, they appear only relatively scarcely in archaeological records regionally in the Southwest (Truett 1996) and locally in the Jemez Mountains (Allen 2004). Mexican gray wolves and grizzly bears inhabited the local area until extirpation in the 1930s, and likely played some role in reducing elk numbers. Another predator, the cougar, has been the target of eradication efforts throughout the 20th century but has persisted in low numbers. Although the elimination of these predators was likely a boon for elk, human hunting pressure and limited surface water were probably greater constraints. Another difference between the 19th century and now is that elk are not naturally a forest species. They prefer open areas where they feed on herbaceous plants and can see their predators form a distance. With the advent of the rifle, elk can be killed from a distance and, therefore, they have found an advantage in spending greater amounts of time under forest cover in areas with hunting. Consequently elk are browsing forest species more heavily. A reference state would ideally be significantly after Puebloan occupation, before large-scale introduction of livestock and fire suppression and during a period of time when elk were present in low densities.

There is no reason to believe that aspens did not regenerate well after high-frequency watershed-wide fires in the Bandelier NM landscape prior to 1900, as they have been shown to do so when not constrained by other factors, like herbivory (Patton & Avant 1970). Aspens were much less likely to have been constrained by elk herbivory because elk are believed to have been a relatively minor species (<3% of large mammals) in the western U.S. before Euro-American settlement since they are poorly represented in archaeological and paleontological records (Allen 2004). This contrasts strongly with the current situation in which there has been major concern about elevated levels of elk herbivory cross the West (Murie 1944, Beetle 1962, Basile 1979), and a paucity of aspen regeneration for much of the 20th century (Ripple & Larsen 2001). Rocky Mountain elk (*Cervus canadensis*) were native in the Jemez Mountains until extirpated by about 1900 (Bailey 1931). This same subspecies was reintroduced to the Jemez Mountains in 1948 with elk from Yellowstone National Park, and in 1964-65 with elk from the Jackson Hole, Wyoming area (Allen 1996). The main difference between the pre-1900 and current role of elk in the ecosystem is based on population numbers (See section 4.17).

4.04-1.2 Reference conditions

Aspen is not browsed at a level that suppresses stand development.

4.04-1.3 Objectives of data analysis, data sources and methods

2001 through 2006 elk exclosure data.

Five elk exclosures were established in mixed conifer forests to track the influence of elk herbivory on aspen recovery, in addition to impacts on higher trophic levels. For each of the five exclosures and their adjacent controls, surveys counted the number of aspen individuals, and their frequency by height class. One pair of plots was burned in the 2000 Cerro Grande Fire. Bowker and Smith (2014) used a paired repeated measures design to examine the impact of elk herbivory on aspen growth following the Cerro Grande Fire.

Aspen and shrub recovery, random plots in Cerro Grande Fire vicinity; 2002 and 2005 data.

Maximum height and number of growing tips of aspen and elk herbivory-susceptible shrub species were measured in one set of random plots in 2002 (mostly located within the Cerro Grande Fire boundary) and in another set of random plots in 2005 (mostly located outside the Cerro Grande Fire boundary). Between 1992 and 2004, 3 distinct but similar studies were conducted that manipulated piñon-juniper canopy cover (Chong 1993; Jacobs and Gatewood 1998; and Loftin 1999). Bowker and Smith (2014) used classification and regression trees to model aspen height, and percentage of tips browsed as a function of spatial coordinates, elevation, pre-fire vegetation type, slope, aspect, and whether or not a site was located within the Cerro Grande Fire.

2006 aspen recovery in "refuge" plots in the Cerro Grand Fire vicinity; 2006 data.

During the collection of the previous dataset, it was observed that palatable saplings and shrubs appeared to be refuged by on-site elements, such as downed logs or rocks. This dataset focused on characterizing the properties of sites in which *Populus tremuloides* or elk-susceptible shrubs may be refuged from elk browsing. Field crews searched the entire Cerro Grande burn area looking for examples of an aspen or shrub that may be benefiting from a refuge element. When such a case was located, crews marked and sampled a 100 m^2 plot centered on the refuged individual or individuals. A total of 115 plots were sampled in this way. Bowker and Smith used classification and regression trees to model aspen height and cover as a function of refuge site characteristics.

Pooled data.

Bowker and Smith (2014) pooled the above data to produce maps of percent aspen browsing in 2002 and 2005, and maximum aspen height in 2006. All of these data were plotted through time as a simple means of gauging park-wide aspen recruitment and the impact of elk browsing.

4.04-1.4 Condition and trend

Aspen is currently decreasing in abundance because of the lack of successful regeneration on uplands with level or moderate slopes that are not protected from elk browsing.

Bowker and Smith (2014) found clear evidence that elk herbivory constrains aspen growth (Figure 4-11). In the first five years following the Cerro Grande Fire, elk exclosures and nearby controls did not have clearly different numbers of aspens. The numbers of aspens were highly variable among the different exclosure-control pairs, and in some pairs the exclosure had a greater number of sprouts, while in others control plots had more sprouts. This variability likely results from three linked factors:

- whether the plot burned
- the degree of shading in the understory
- soil temperature fluctuations due to different degrees of canopy closure

This suggests that the ability of aspen to initially resprout in an area, as suckers from roots of a surviving clone, is not clearly reduced by elk in this dataset. Elk browsing and debarking of mature aspens could reduce sprouting vigor, but these data account only for exclusion of such impacts since 2000.

Regardless of their numbers, aspen individuals are currently subject to intense herbivory which constrains their size. On average, saplings were constrained to around 40 cm in height and did not show an upward growth trend. Inside exclosures, aspens were clearly becoming larger over time and the growth rate increased at each increment. In 2006, they had attained a height about 4 times that of their elk-affected counterparts. While in the short-term, this herbivory does not seem to result in mortality of aspens, it is reasonable to assume that a long-term continuation of this pattern would eventually result in the depletion of energy reserves and death of these stems as observed in two 10-acre areas within Bandelier NM and elsewhere (Lindroth &and St. Clair 2013). Perhaps most interesting was the fact that fire intensity and elk exclusion seem to interact. In Figure 11c, we can see that control plots have similar, small aspen heights regardless of fire intensity. But, when protected from herbivory, the burned plot had aspens sprouts nearly double the size of the unburned plots. This suggests that aspen are well-equipped to recover from fire as long as they are not constrained by herbivory. These results are not surprising, and are in line with several other studies of the elk-aspen interaction in

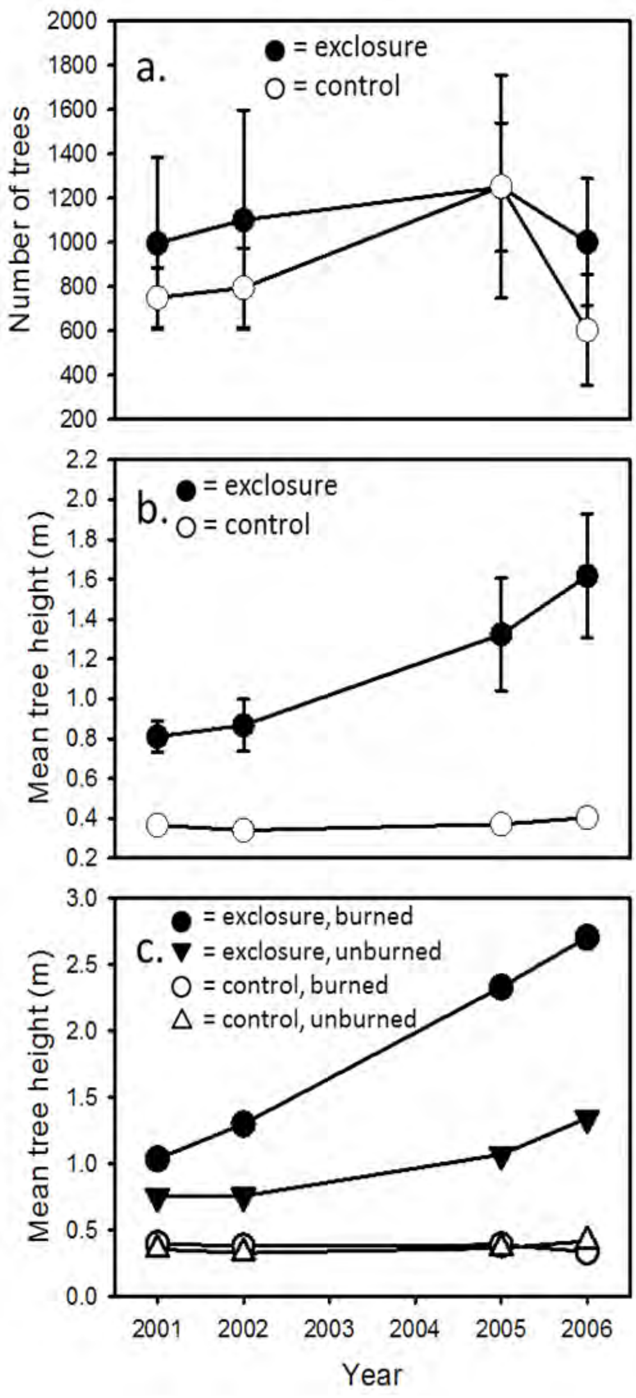

Figure 4-11. Number and mean height of aspen stems inside elk exclosures and in control plots outside of exclosures over five years. a. None of the considered factors affected number of aspen stems (Exclosure: F=1.6, P=0.26 Time: F=1.3, P=0.39), b. Mean aspen height was much greater inside exclosures, increased over time inside exclosures, and increased most inside burned exclosures (Exclosure: F=137277, P<0.0001; B × E : F=57.2, P=0.0003; Time: F= 160.0, P 0.0001; E × T: F=8.9, P=0.10), c. Illustration of the B × E interaction.

western North America.

Several of Bowker and Smith's analyses focused on the variation in indicators of elk herbivory on aspen across the landscape. They wished to determine which were the strongest influences upon intensity of herbivory, and whether these could be predictively modeled. Three datasets were available, dating from 2002, 2005, and 2006, that were pertinent to the recovery from the Cerro Grande Fire. In 2002 and 2005 there were direct observations of browsed tissue on aspen, supplemented with information on maximal sapling height, which would be expected to be under the influence of elk herbivory, but also influenced by climate and soil characteristics. In 2006, two indirect indicators of elk activity were available: aspen maximal height and cover.

Perhaps the most striking conclusion that can be drawn from this set of analyses is that elk herbivory has a strong spatial component, regardless of the year. Krantz (2001) compiled elk observations in Valles Caldera National Preserve in 2001, and found that the Rincon area of Valle Grande consistently had the largest number of elk, numbering in the hundreds for most observations. This result is attributed to the fact that it is a large area with considerable forest edge. Valles Caldera elk prefer to spend the night in open areas and spend days under tree cover, so the Rincon area attracts large numbers of animals. This demonstrates that the natural distribution of elk is highly variable spatially, and that there can be consistent hotspots of elk activity in a given year.

The spatial pattern of elk browsing indicators in Bandelier NM may be demonstrating something similar. Interestingly, in 2002, it was the far eastern edge of the northern portion of the monument which experienced the greatest herbivory. However, the northernmost portion of this area exhibited the smallest aspen heights. In 2005, models were much poorer in explanatory power (not shown: % browsing R2 = 0.05; maximum height R2 = 0.18) but the % browsing model indicates the greatest browsing north of the monument in lower lying areas, i.e. Valle Grande. Also in 2005, the smallest maximal tree heights were observed in the Bandelier NM portions of the Cerro Grande Fire area, but were about twice as large east of the park boundary. The 2006 aspen height data suggests more northerly locations had shorter aspens, and that in between 3967997N and 3968536N, no aspens were observed. Again this suggests a spatial heterogeneity in herbivore pressure. Some places experience much greater herbivory than other locations, and this patterning may shift from year to year (Wolf 2003).

4.04-1.5 Level of confidence

High; this is a highly researched topic across the western U.S.

4.04-1.6 Data gaps/Research needs/ Management recommendations

Trophic cascades associated with the loss of key predators has been shown to be an urgent global issue known to specifically be acting in U.S. national parks (Estes et al. 2011). The loss of large predators in the 1900s has left a characteristic signal in reduced tree growth rates at Isle Royal National Park (McLaren and Peterson 1994) and recruitment failure at Yellowstone National Park (Beschta and Ripple 2009). The monument's studies of aspen supports the synthesis provided by Estes et al. (2011).

Two management recommendations surface clearly from our current scientific understanding. First, land management planning efforts need to address the cascading ecosystem changes when consumers such as ungulate increase with the loss or continued absence of predators (Estes et al. 2011). Through planning efforts (e.g. compliance with NEPA), managers should seek to educate the public regarding the ecological role of apex predators and how their loss threatens biodiversity (Estes et al. 2011). Second, the best management solution is likely the restoration of an effective predation regime (Estes et al. 2011)—clearly not an easy task.

Management considerations in the face of climate change

Beschta et al. (2012) summed up the topic, saying that if effective adaptions to the adverse effect of climate change are to be accomplished on western public lands, large-scale reduction or cessation of ecosystem stressors associated with ungulate use are crucial. Active ecosystem management, including the elimination of non-climate in situ threats, such as herbivory, are important in the face of climate change (Welch 2005).

The park should conduct monitoring to determine regeneration of unprotected aspen stands within the footprint of the Las Conchas Fire.

4.04-1.7 Sources of expertise

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4.04-1.8 Literature cited

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4.05 Ponderosa Pine Forest and Woodland

This is a brief summary of the information provided by Bowker and Smith (2014) in Appendix B. Detailed descriptions of ponderosa pine forest ecology, the data used, analysis methods, and all references are included therein. The information below is mostly interpreted or taken directly from that report.

4.05.1 Description

Ponderosa pine (*Pinus ponderosa*; 'ponderosa') vegetation communities can be described as forest, woodland or savanna, depending on canopy density, but generally have ponderosa as the dominant overstory species. Ponderosa pine forests across the southwestern U.S. are threatened by several factors, primarily changes in fire regimes that have led to unnaturally high tree densities, and the interacting impacts of climate change and drought (Negron et al. 2009, Wallin et al. 2004), factors that are described in detail in Bowker and Smith (2014) and explained briefly below.

4.05.1.1 Climate

One of the most important expressions of climate change is increasing warm drought periods (Williams et al. 2012). The western U.S. is currently experiencing the warmest temperatures observed in a millennium (Intergovernmental Panel on Climate Change [IPCC] 2007), and models strongly suggest climate scenarios wherein northwestern New Mexico will very likely experience continued warming, possibly along with less spring precipitation (Karl et al. 2009).

Within Bandelier NM, the 1950s drought and associated bark beetle outbreak killed mesatop ponderosa pines across a several kilometer band along its lower elevational distribution over a brief 5-year timespan Figure 4-12 (Allen and Breshears 1998). Regionally, in the years following the 2002 drought many individuals or even entire stands died (Breshears et al. 2005; Gitlin et al. 2006).

4.05.1.2 Fire

Fire patterns in southwestern ponderosa pine forests have changed considerably over the last several centuries. Until about the late 1800s, fires were common, but remained mostly on the ground (rather than burned in the crowns of trees). Fire behavior was driven by forests that had fewer trees and a greater understory of grasses and shrubs that maintained surface fires and minimized long-term damage to mature trees (Allen 1989, Allen et al. 1995). The average fire return interval was about 5–15 years, which limited survival of all but a few tree cohorts (Figure 4-13).

With the introduction of grazers (cattle and sheep), much of the herbaceous understory was consumed or trampled, fires were routinely suppressed, and ponderosa pine forests across the southwest grew thicker and older. The result was much higher tree densities, a situation that greatly increases the potential for crown and high intensity fires that kill large trees which would otherwise

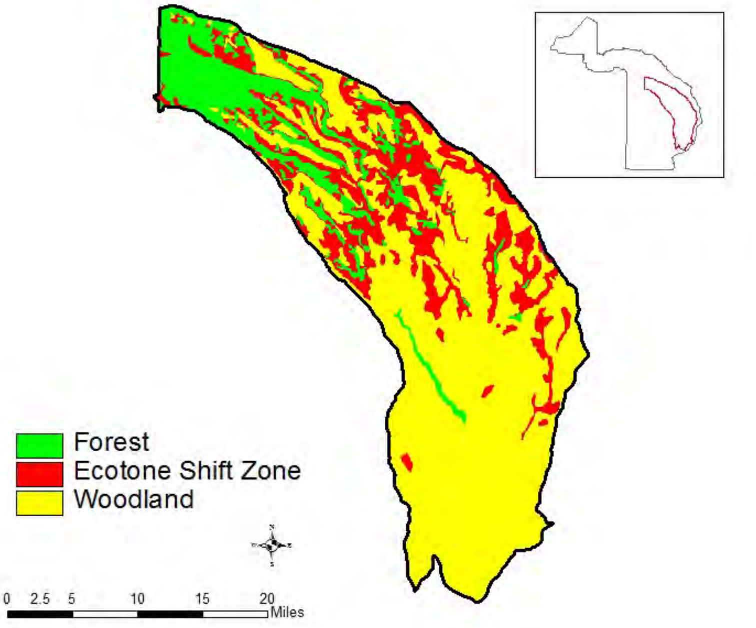

Figure 4-12. Changes in vegetation cover, 1954–1963, in the study area at Bandelier National Monument, showing persistent ponderosa pine forest (green), persistent piñon-juniper woodland (yellow), and the ecotone shift zone (red) where forest changed to woodland.

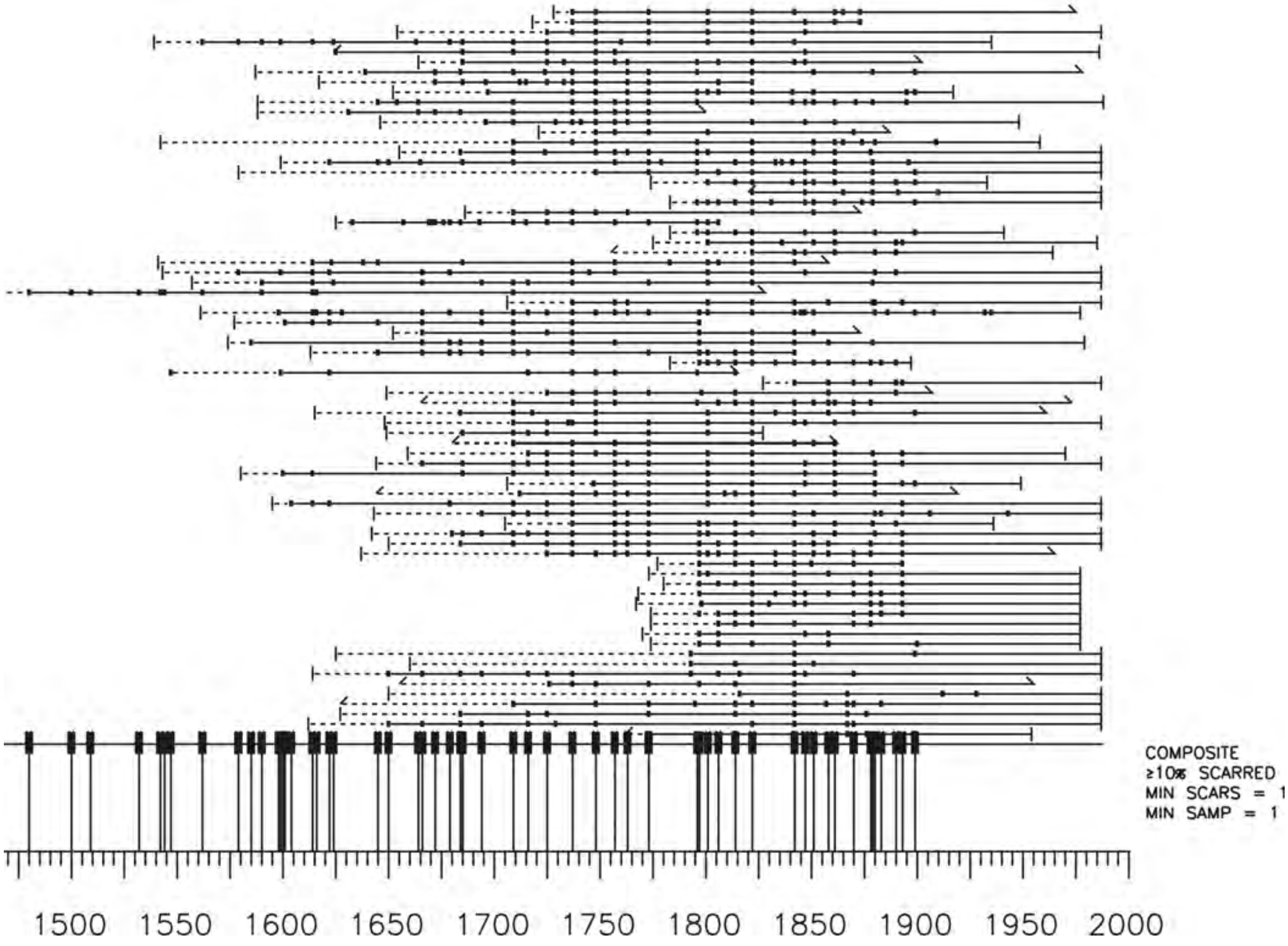

Figure 4-13. Fire scar chronology for samples collected in Frijoles Canyon. The horizontal lines depict individual tree lifespans. The dots along the lines represent recorded fire scars. Many trees record fire in the same year. Fires burned at a regular interval until after 1900.

survive low intensity surface fires (Potter and Foxx 1978, Allen 1989, Moore et al. 1999, Allen et al. 2002). In addition, exotic grasses (particularly cheatgrass, *Bromus tectorum*, and other brome species) are now permanently present in the seed bank of many southwestern forests or areas adjacent to them, and following fires, can now outcompete many native herbaceous species (Korb et al. 2003, Barclay et al. 2004, McGlone et al. 2009).

Fire is a part of the evolutionary environment of the ponderosa pine forest (Moore et al. 1999). However, fires in recent decades have been dissimilar to the surface fires that occurred from ca. 1500–1900, and can be characterized as having a much greater extent of high-severity fire effects. Since the 1970s, several major crown fires have occurred in and around Bandelier N. M. The 1977 La Mesa Fire (6,354 ha), the 1996 Dome Fire (6,677 ha), the 1997 Lummis Fire (607 ha), the 2000 Cerro Grande Fire (16,909 ha), and the catastrophic 2011 Las Conchas Fire (63,250 ha) all burned at least some ponderosa pine forests (Figure 4-14a), with the La Mesa, Dome and Las Conchas impacting ponderosa pine forests most severely(Figure 4-14b). These large, stand-replacing

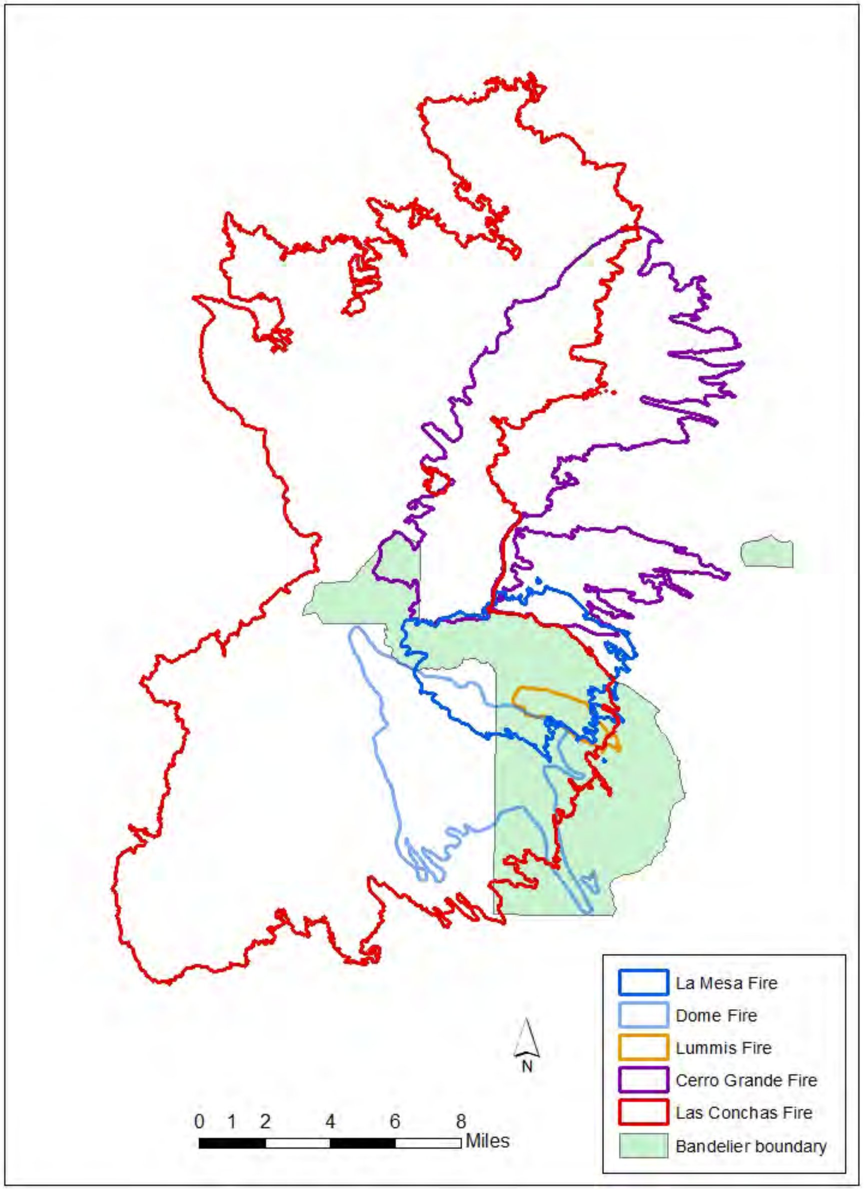

Figure 4-14a. Bandelier National Monument and outlines of 1997 La Mesa Fire, 1996 Dome Fire, 1997 Lummis Fire, 2000 Cerro Grande Fire and 2011 Las Conchas Fire.

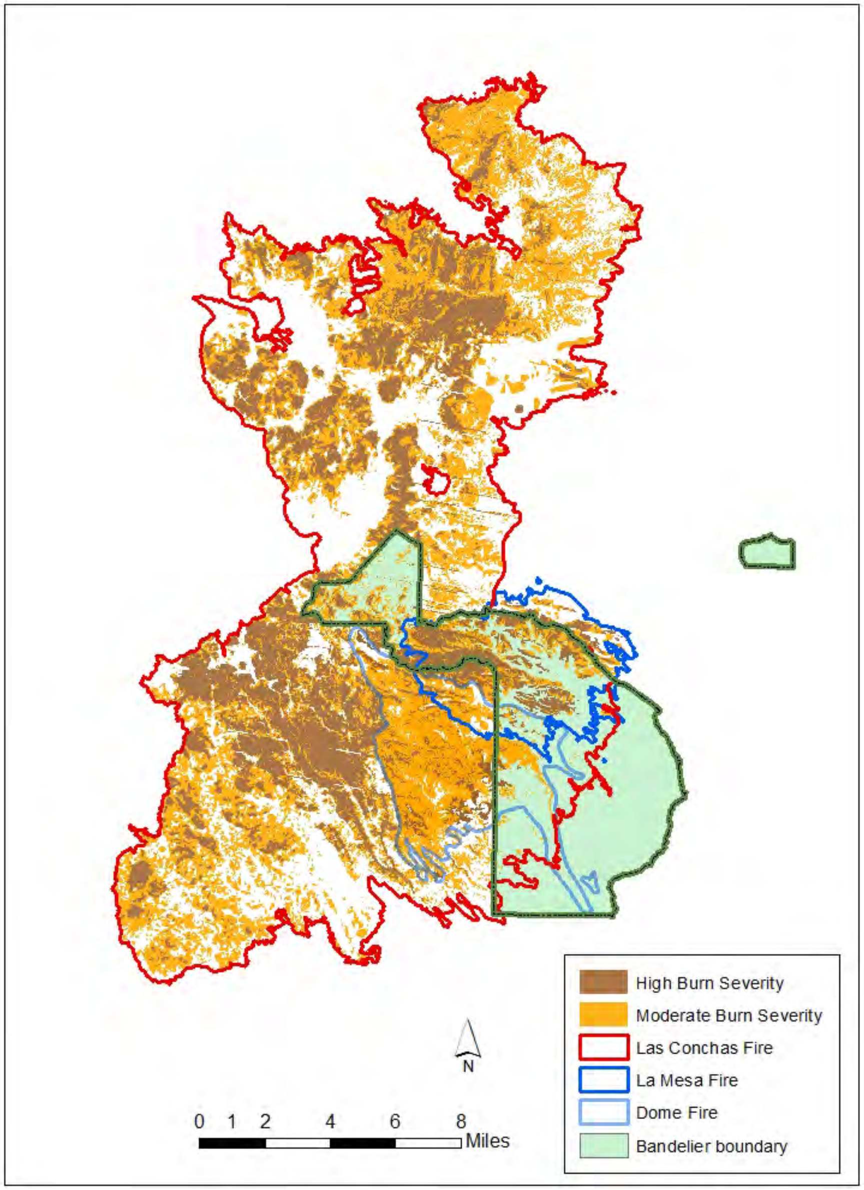

Figure 4-14b. Bandelier National Monument and the areas of high and moderate burn severity for the three fires (1997 La Mesa Fire, 1996 Dome Fire and 2011 Las Conchas Fire) that represent areas of tree cover loss.

fires were a result of over a century of fire suppression and fuel build-up interacting with warmer temperatures brought about by recent climate trajectories.

As has occurred elsewhere in the region, 20th century fire suppression resulted in rapid recruitment of ponderosa pine saplings, resulting in “doghair thickets” of small diameter trees. One local example of old-growth ponderosa pine densities, and subsequent stand thickening, comes from the Monument Canyon Research Natural Area of Santa Fe National Forest to the west of Bandelier NM. This stand displayed a two-tiered tree distribution with an old-growth density of 100 stems/ha and an understory thicket of saplings and poles with a tree density of 21,500 stems/ha (Allen 1989). Similar stand conditions and high fuel loads were evident in the area burned by the 1977 La Mesa Fire which converted the heart of the monument’s ponderosa pine forest into grasslands (Allen 1989).

The historical fire regimes of ponderosa pine forests in the Jemez Mountains have been relatively well reconstructed using tree-ring methods and charcoal (Touchan et al. 1996, Allen 2002, Anderson et al. 2008). Prior to the 1860s, Utes, Apaches and Navajos using the Jemez Mountains area were a deterrent to settlement by Euro-Americans (Allen 2004). Shortly afterward, in the 1870s and 1880s, livestock were introduced in large numbers into what is now Bandelier NM, at very high stocking rates (Allen 2004). The grazing pressure diminished fine herbaceous fuels, curtailing the spread of fire and the size of fires. The previously high-frequency fires were further curtailed by the adoption of fire suppression policies in the early 20th century. Long-term fire scar records confirm that this omission of fire since the late 1880s in the Jemez Mountains is an anomaly. The fire regime can be tracked over periods of centuries rather than just a window of time just before the 20th century. Touchan et al. (1996) and Allen (2002), documented the last 500 years of burn scars, and provide strong evidence of a near complete cessation of fire by around AD 1900. In the 400 years prior to the 20th century, ponderosa pine forests experienced frequent fires, most of which were low-intensity surface fires. The fire return intervals were largely about 5–15 years. Large crown fires were absent from ponderosa pine forests in local and regional fire scar records (Swetnam and Baisan 1996).

4.05.2 Reference conditions

An extensive body of research has addressed the definition of reference conditions for southwestern ponderosa pine forests in general (e.g. Covington and Moore 1994, Fule et al. 1997, Moore et al. 1999 Swetnam et al. 1999). The conclusions from these studies are that prior to European settlement, ponderosa pine communities were mostly open canopied forests and savannas, with tree densities ranging from 8–25 trees per hectare (Fule et al. 1997, Allen et al. 2002, Woolsey 1911). Understories included native grasses and shrubs, such as *Quercus gambelii*— compositions maintained by frequent, low-intensity surface fires (Allen et al. 1995; Abella 2008).

In the Jemez Mountains, open ponderosa pine forests covered the middle and upper portions of the Pajarito Plateau 125 years ago. By 1900, heavy livestock grazing had degraded the grassy understory, leading to de facto fire suppression, which was later continued as 20th century fire suppression policy (Allen 1989).

4.05.3 Objectives of vegetation analysis, data sources, and methods

The data used and all analysis methods are described in detail in Bowker and Smith 2014 (Appendix B).

4.05.3.1 Vegetation maps

Two vegetation maps, Allen 1989 (`Allen’) and Muldavin et al. 2011 (`Muldavin’), were compared to detect changes in vegetation cover over time. The Allen map was based upon 1981 imagery while the Muldavin map was based upon 2004 imagery. To facilitate comparisons among the two maps, both were reduced to their intersection and the

Muldavin map reclassified to conform to the Allen map (complete details of this process provided in Bowker and Smith 2014). Although this translation was imperfect, most major Muldavin units corresponded reasonably well to major Allen units. For ponderosa pine, Muldavin's eight types were reduced to Allen's four types (Bowker and Smith 2014).

4.05.3.2 Fire maps

Fire severity maps for each of the fires mentioned above (or fire perimeter, only if severity was not available), were spatially oriented with the vegetation maps. Multiple methods were used to compare seeded vs. unseeded, and variable fire intensities with vegetation composition to analyze changes in cover of various functional groups (forbs, grasses, shrubs, subshrubs and trees).

4.05.3.3 Fire effects monitoring plot data

Fire monitoring plots were established to examine shifts in vegetation characteristics after prescribed burns, mechanical thinning, and wildfire. This analysis used only data from plots representative of various types of ponderosa pine stands, or former ponderosa pine stands. Plots were defined as being from one of four areas: upper elevation ponderosa, low elevation ponderosa, previous ponderosa (area of La Mesa Fire) and the area of mechanical thinning. Burned plots were surveyed before a prescribed burn, immediately after the burn, and at one, two, five, and 10 years after the burn. Similar to burned plots, thinned plots were surveyed immediately before and after thinning.

4.05.3.4 Post-Dome Fire transects

In 1996, following the Dome Fire, an emergency seeding operation was initiated on the Santa Fe National Forest (SFNF) portion of the burn, but not on the monument's portions. In 1997, 49 vegetation sampling transects were established in and around the burn, ten in the park and the remainder in SFNF. Transects were located to sample areas that varied in pre-fire forest type, elevation, fire intensity, and seeding treatment. At each location, herbaceous vegetation was sampled using line-intercept methods, and tree regeneration counted within a plot centered around the transect (Barclay et al. 2004). All of the plots were measured in 1997 and 1998, the park's plots were re-measured in 2002, and the SFNF plots were re-measured in 2008.

4.05.3.5 Data analysis

PerMANOVA (Anderson 2001) was used to statistically test for differences in the overall cover composition of functional groups. Using the same community matrix, non-metric multi-dimensional scaling (NMDS) (McCune and Grace 2002) was used to graph functional group composition across fire intensities (high, moderate and unburned) and seeding (seeded vs. unseeded). An ANOVA was used to evaluate changes in cover of individual functional groups, species richness, total live canopy, and specifically for *Bromus tectorum* and *B. inermis*.

4.05.4 Condition and trend

Vegetation change in the ponderosa pine zone of Bandelier NM is largely driven by fire and associated invasive species impacts, and by drought and associated pathogen outbreaks. After about a century of fire suppression and relatively few fires, large, and often intense, fires are strongly impacting the ponderosa pine forests of Bandelier NM (Figure 4-4, p. 49). The 1977 La Mesa Fire had a major effect on the distribution of ponderosa pine forests (Potter and Foxx 1978, Allen 1989). Within Bandelier NM, nearly all of the area that experienced at least moderate severity burns, was mapped from 1981 air photos as vegetation types other than ponderosa pine forests—usually grasslands or shrublands (Allen 1989). Since it is known that most of the burned areas were ponderosa pine forests, this observation suggests a fire-triggered type conversion. Most of these areas have still not reverted back to ponderosa pine forests based on more recent 2004 mapping.

The 1996 Dome Fire impacted ponderosa pine forests, but mostly forest outside of the Bandelier NM boundary on the national forest side. Within the park, west of Capulin

Canyon, a sizable block (~ 1km^2) of contiguous ponderosa pine forests experienced moderate to high intensity fire (Fig. 4-2b, p. 39).

The 1997 Lummis Fire was almost entirely within ponderosa pine habitat, including some areas previously converted during the La Mesa Fire). This fire occurred in the largest remaining contiguous block of ponderosa pine forest remaining in the monument, and its perimeter encompassed more than half of that block. The Las Conchas Fire was a much larger fire, but its impacts on ponderosa pine forests within the monument were smaller, perhaps because of the fuel consumption effects of other recent fires (Figure 2d). On adjacent National Forest lands, many ponderosa pine and former ponderosa pine stands that were impacted by the Dome Fire reburned in Las Conchas.

In summary, ponderosa pine forests with herbaceous understories have been greatly reduced in coverage on the monument landscape by high severity fires since 1977, and are becoming increasingly fragmented (Figure 4-15). Shrubs appear to be emerging as either dominant species or major community components, along with grasses. Shrublands or shrub-grassland associations, in which shrub species re-sprout after fire, are common in various parts of the U.S. and the world. It seems likely that increasing fire frequency would perpetuate these community types and constrain recruitment of *Pinus ponderosa*.

Bowker and Smith (2014) used the Post-Dome Fire transect data (transects were primarily in ponderosa pine, but ranged from mixed conifer forest to piñon-juniper woodland) and fire monitoring plot data to examine the components of ponderosa pine community structure and composition that changed with fire. They observed that species richness was highest at moderate burn severities. Their analysis also suggested that total vegetative cover differed across burn severities, such that cover was highest in unburned plots and lowest at high burn severities. In examining canopy cover of individual functional groups, they found significant differences in the cover of forbs, shrubs, and trees across burn severities. Tree cover in unburned sites was approximately 58%, but

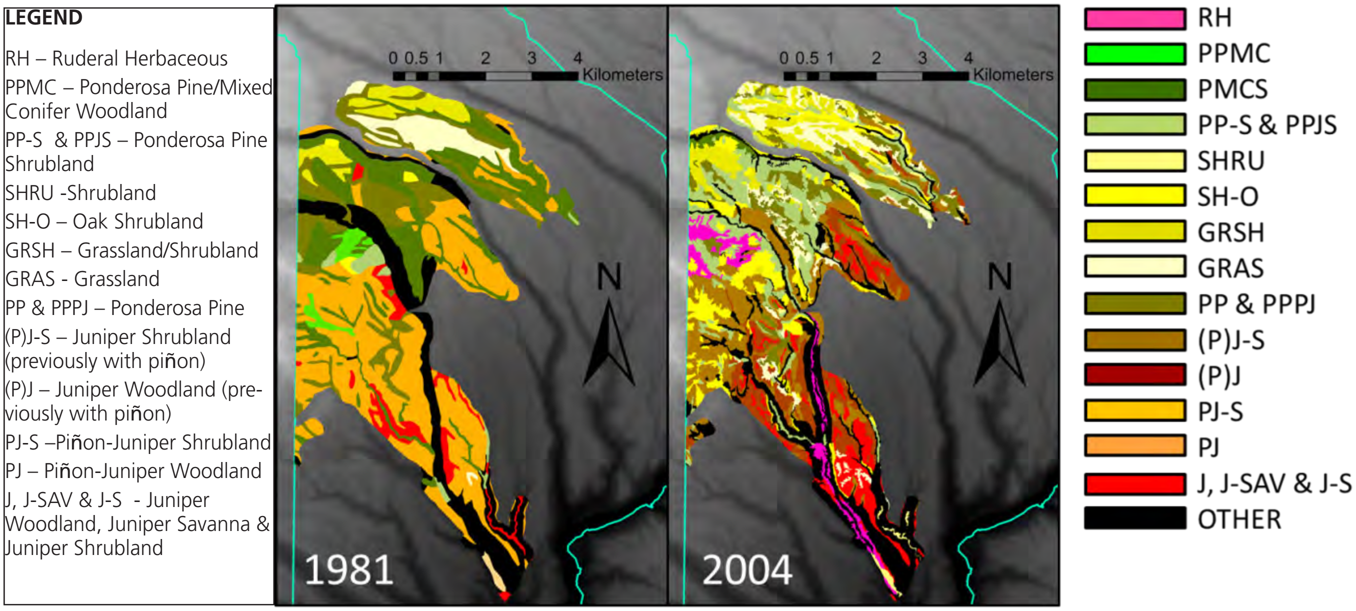

Figure 4-15. Vegetation communities within the Dome (1996) and Lummis (1997) fire perimeters as mapped based on 1981 imagery (Allen 1990) and 2004 imagery (Muldavin 2011).

only 3% and 1% respectively, in moderate severity and high severity burn sites. Shrub, forb and grass covers in moderately or severely burned sites were roughly two-fold that of unburned sites.

Over the past several decades, there have been major transformations in the distribution and vegetation structure of the ponderosa pine forests of Bandelier NM. Some of these changes have been induced by deliberate human action (thinning, prescribed burning and seeding), while others have been brought about by inadvertent human perturbation (climate change and wildfire). Wildfire in particular has had the greatest impact, resulting in fragmentation of the ponderosa pine forests, and establishment of large grassland patches. Wildfire has also promoted shrubs, both as a dominant species, and as an understory subdominant. As climate change increasingly favors fires, it can be expected that their frequency and magnitude will continue increasing (Williams et al. 2010).

The recent fires were the outcome of changing climate and fuel and canopy conditions brought about by a century of fire suppression. Since canopy cover and the continuity of ponderosa pine forests has already changed so much, we can expect that future fire behavior and outcomes could be different. Grassland patches are likely to experience frequent low intensity fire, but are being colonized by shrubs which will change their fire behavior. Shrublands and forests with shrub understories are likely to experience moderate to high intensity fire. If trees are part of the community or adjacent to these communities, the fire may be capable of jumping to tree crowns. The major shrubs (oak and locust) are fire resprouters. While we are seeing significant departure from historical vegetation conditions which formerly supported a surface fire regime, one positive aspect is that a more crown fire-resilient vegetation type is emerging. These scenarios would logically lead to a future where grasses and shrubs become more prevalent and promote their own persistence by modifying fire behavior and fuel characteristics. It would seem that ponderosa pine may have a lesser role in the resulting systems. If future climate conditions include multi-decadal wet spells between drought episodes, ponderosa pine may persist, but perhaps at lower densities, especially in shrub dominated portions of the landscape.

Major drought events, such as that of 2002 (Breshears et al.2005), are another force influencing vegetation dynamics. As stated elsewhere in this report, the future will very likely be characterized by increasingly warm, and therefore more severe droughts (Adams et al. 2009, Williams et al. 2010). It is clearly a factor which reduces the probability of the persistence of ponderosa pine stands (Allen & Breshears 1998, Gitlin et al. 2006). Less well understood are its effects on the recruitment of shrubs and exotic grasses. In the fire monitoring plots, shrub cover increased consistently each year from 1993 to 2000, resulting in an approximately 18-fold increase in shrubs. After the drought of 2002, shrub cover decreased to 1993 levels, consistent with observations of shrub canopy diebacks that year (Brian Jacobs and Craig Allen, personal communication).

In the remaining ponderosa pine stands that have withstood crown fire and drought, management activities may alter the successional course. Prescribed burning and mechanical thinning reduced overstory tree density by about 50 and 40%, respectively, from undisturbed controls. The reduction in overstory trees is consistent with historical conditions, and would likely avert or reduce the extent of crown fire runs in the future, allowing the persistence of ponderosa pine as a major community member. Interestingly, in contrast to wildfire, we did not find an influence of prescribed burning or thinning on shrub or understory herbaceous cover vegetation, suggesting that the treatments have not thinned the overgrown forests sufficiently to restore pre-1900 surface fire conditions.

In this ecosystem, which is so prone to change, desired conditions should be viewed as a moving target. The extent, abundance, and continuity of ponderosa pine woodland have been compromised, and the best guess at the future is a continuation of this trend; an increasing "shrubification" and an increasing presence of exotic grasses. This future does depend on which management actions are undertaken. There is some evidence that thinning and prescribed burning could slow this trend, by reducing the probability of crown fire in the remaining stands, and by removing the force which converts forests to shrublands—high-severity fire.

4.05.6 Data gaps/Research needs/ Management recommendations

- Generally, post-fire seeding is not likely to lead to desirable long-term vegetation outcomes and is not effective in reducing erosion. Seeding results in increased cover of exotic species that persists indefinitely.
- There is some evidence that thinning and prescribed burning could slow the process of conversion from forests to shrub and grasslands by reducing the probability of crown fires in remaining stands. See discussion of forest management under Jemez Mountain salamander.
- Climate change scenarios strongly indicate that northwestern New Mexico will experience continued warming and possibly less spring precipitation (Karl et al. 2009), conditions that will continue to inhibit, through various mechanisms, the regrowth of ponderosa forests. This increases the importance of landscape-wide conservation measures targeted at ponderosa pine forests.

4.05.7 Sources of expertise

Matthew Bowker, David Smith, and Stephen Fettig

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4.06 Piñon-Juniper Woodland

This is a summary of the information provided by Bowker and Smith (2014) in Appendix B. More in-depth descriptions of piñon-juniper woodland ecology, the data used, and analysis methods, are included therein. The information below is mostly interpreted or taken directly from that report.

4.06.1 Description

Piñon-juniper woodlands occur on nearly 40 million ha in North America (Romme et al. 2010) and include many different community types. In all cases, however, they occur in semi-arid climates, usually at medium elevations (~1200 –2500m), and contain at least one *Pinus* species and a *Juniperus* species. In Bandelier NM, until recently, piñon-juniper woodlands were dominated by *Pinus edulis* and *Juniperus monosperma*, with *Juniperus deppeana* either a subdominant or co-dominant in local areas (Muldavin et al. 2011). While a variety of woodland types that occur in Bandelier NM can be described as piñon-juniper (described in detail in Bowker and Smith 2014), most piñon-juniper woodlands in the monument best fit the persistent woodland type.

4.06.1.1 Climate

One of the most important expressions of climate change is increasingly warm drought periods (Adams et al. 2009). The western U.S. is currently experiencing the warmest temperatures observed in a millennium (IPCC 2007), and models strongly suggest climate scenarios wherein northwestern New Mexico will very likely experience continued warming, possibly along with less spring precipitation (Karl et al. 2009).

In the Southwestern U.S., piñon pine is susceptible to mass mortality during drought (Allen and Breshears 1998, Breshears et al. 2005, Gitlin et al. 2006). The impact of the 2002 drought across the Bandelier NM landscape provides a stark example. While major mortality of ponderosa pine and some impacts to Douglas-fir and white fir were observed, the 2002 drought decimated piñon pine, with over 90% mortality of mature trees. This has resulted in substantial transformations in vegetation community distribution and canopy cover on the landscape (Bowker and Smith 2014). *Juniperus monosperma*, the most common co-dominant of *Pinus edulis* in the monument, is more drought-tolerant than piñon but also experienced stand-level mortality during the recent drought periods (Bowker et al.2012, Brian Jacobs and Craig Allen, personal communications and unpublished data from Bandelier NM plots and transects).

4.06.1.2 Fire

In addition to variation in species composition, piñon-juniper woodlands can differ fundamentally in disturbance regimes. The natural fire regime of these ecosystems is not completely known because, unlike larger fire-resistant species, such as ponderosa pine, piñons and junipers often do not survive fire, so tree-ring data commonly are unavailable for use in reconstructing fire histories. Understanding the role of fire in the piñon-juniper woodlands of Bandelier NM is complicated by the diversity of piñon-juniper community types that occur across the landscape. These include juniper savannas, piñon-juniper savannas, piñon-juniper -oak communities, and most commonly, persistent piñon-juniper woodlands.

At higher elevations, piñon-juniper woodland intersperses with, and in some sites, has recently replaced ponderosa pine savanna. Some have suggested that a major herbaceous understory component historically existed in piñon-juniper communities of the region and that surface fires may have preserved this condition (Gottfried et al. 1995, Jacobs & Gatewood 1999). However, evidence is sparse and mostly from ponderosa pine stands near piñon-juniper stands (cf. Allen 1989).

Patchy stand-destroying crown-fires are well-documented in piñon-juniper woodlands from elsewhere in the region (Romme et al. 2009). Pre-1900 disturbance regimes in piñon-juniper savanna are not well understood. Piñon-juniper woodlands have been less affected by fire than other ecosystems, but large areas of piñon-juniper have been lost in recent fires, particularly in the 1996 Dome Fire and the 2011 Las Conchas Fire.

Persistent erosion has also had substantial negative impacts on piñon-juniper communities in Bandelier NM. Overgrazing and the resulting loss of the herbaceous understory likely accelerated this process (Allen 1989, Miller and Wigand 1994, Brockway et al. 2002). High erosion rates create a positive feedback loop, whereby herbaceous plants cannot colonize the unstable surface, and because there are few plants in between trees, erosion is unchecked.

The past conditions of the piñon-juniper woodlands are characterized by change, and can be coarsely reconstructed based on historical or archaeological evidence. Significant portions of piñon-juniper woodlands may have been cleared on the monument's mesa tops during the period of Ancestral Puebloan settlement (Allen 2004). The period of abandonment coincided with a megadrought in the 1500s. Both settlement and mega-drought could be viewed as likely triggers of ecological state transition. Recovery and regrowth from this event must have occurred during the cool period known as the Little Ice Age.

It is known that the ecosystems that grew back from these earlier perturbations were woodlands with a substantial grass cover, as noted in the Ramon Vigil Land Grant of 1877 and later adjacent Land Grants surveyed by the General Land Office (Allen 2004). This land grant and the resettling of nomadic Native Americans onto reservations around the 1860s marked the introduction of large scale livestock grazing (Allen 2004), since the Navajo, Apache and Utes had previously served as a deterrent to Euro-American use of the area.

Although livestock had certainly been introduced earlier, their numbers have not been documented and cannot be estimated well, but are thought to be much lower than after the 1860s. Thus, the period from 1860–1870 is one of the better periods to use as a reference to compare current conditions to because it is over 300 years after occupation by Ancestral Puebloans, yet just before the introduction of large livestock herds. The introduction of livestock quickly compromised herbaceous plant cover, as stocking rates were an order of magnitude greater than what would be considered carrying capacity today (Foxx and Tierney 1984) By 1913, grass cover was described by the General Land Office as "scant", compared with descriptions from 30-40 years before as "excellent" or "fine" growth of grass (Allen 2004).

One major result of a degraded herbaceous layer was erosion. The creation of the national monument in 1916, and the transition to management by the National Park Service in 1932 led to the termination of commercial ranching in Bandelier NM, but trespassing or feral livestock has remained a problem (Allen 2004). Erosion and hydrological impacts lingered after the cessation of commercial grazing. As early as 1938 the inability of the soil to retain moisture was noted by the General Land Office (Allen 2004). A 1948 internal park memorandum described the large extent of the problem and suggested means to correct it. High erosion rates may have been further accelerated by the 1950s drought which resulted in vegetation mortality. Accelerated erosion has persisted up to the present as possibly the foremost management issue in the piñon-juniper woodlands.

The sharp decrease in herbaceous vegetation corresponds with a cessation of fires in the general area. An oft-stated belief is that because this major herbaceous component had previously existed in the understories of piñon-juniper communities in the park and nearby, surface fire may have preserved a relatively open canopied configuration (Gottfried et al. 1995, Jacobs and Gatewood 1999). The fire history is difficult to know with certainty because piñon-juniper woodlands are poor preservers of fire scars from which to reconstruct past fire cycles. The evidence for frequent surface fire is indirect but comes from two lines of reasoning:

1. Most individual trees are young, whereas older trees are more widely scattered (Jacobs et al. 2008). Since woodlands generally become thick without maintenance by fire, this implies the presence of fires, which did not destroy whole stands, i.e. low intensity surface fire.
2. A frequent fire cycle is well established for directly adjacent and interfingered ponderosa pine savannas (Allen et al. 1995), and it is reasonable to believe that these fires carried into piñon-juniper stands with herbaceous understories. This series of events may have led to thickening tree canopies, which reinforce a more closed woodland rather than an open grassy woodland. Jacobs et al. (2008) use a predictive model to conclude only about a third of the monument's current piñon-juniper woodlands were recently encroached savannas. The other two-thirds likely were persistent woodlands that may have thickened in the past century and which did not have a frequent-fire regime.

4.06.2 Reference conditions

Prior to historic grazing, Colorado piñon pine (*Pinus edulis* Engelm.) – one-seed juniper (*Juniperus monosperma* (Engelm.) Sarg.) woodlands were largely restricted to unproductive (e.g. steeper and rocky) settings adjoining and interspersed with the (swales, pumice patches, and gentle mesa) locations dominated by ponderosa savanna (Allen 1989, Gottfried et al. 1995, Jacobs et al. 2008). Age-class and observational data suggest the historic woodland component was relatively open canopy, but mostly persistent in nature (sensu Romme et al. 2009) with sparse understories and only supporting patchy crown fire (Jacobs et al., 2008).

4.06.3 Objectives of vegetation analysis, data sources, and methods

Bowker and Smith incorporated the literature on known ecosystem dynamics, expert knowledge, and ecological principles to develop an a priori state-and-transition model for the park's piñon-juniper ecosystems (Figure 16). They then used hierarchical cluster analysis of data from the watershed-scale thinning study to validate the existence of the proposed states (Figure 17).

Between 1992 and 2004, three distinct but similar studies were conducted that manipulated piñon-juniper canopy cover (Chong 1993; Jacobs and Gatewood 1998; and Loftin 1998). The goal of the analysis was to determine the relative importance of interannual climate variability, overstory reduction and slash redistribution, ground surface manipulations, age of treatments, and fire on the relative functional group composition of the understory. Bowker and Smith pooled data from the three studies and analyzed it using regression tree models.

Bowker and Smith used a dataset from 46 monitoring plots established in piñon-juniper woodland in 2008–2010 by the Southern Colorado Plateau Network to examine the relationship between soil aggregate stability (a negative index of erodibility; low scores indicate high erodibility) and factors thought to contribute to soil stability. These factors include mechanical thinning, canopy closure, total biocrust cover, total litter cover, and cover of major plant functional groups (forb, annual grass, perennial grass, and shrub).

Bowker and Smith used two Bandelier NM datasets: 1) line intercept vegetation data collected in 1999 and 2003 from nine 300 m long transects (so-called JRM LTER transects); and 2) erosion bridge measurements of microtopographic soil surface changes from the same nine transects (plus two additional erosion transects measured annually, 1999-2007), to examine possible relationships between vegetation structure and erosion. They used perMANOVA and non –metric multidimensional scaling (NMDS) for the analysis.

4.06.4 Condition and trend

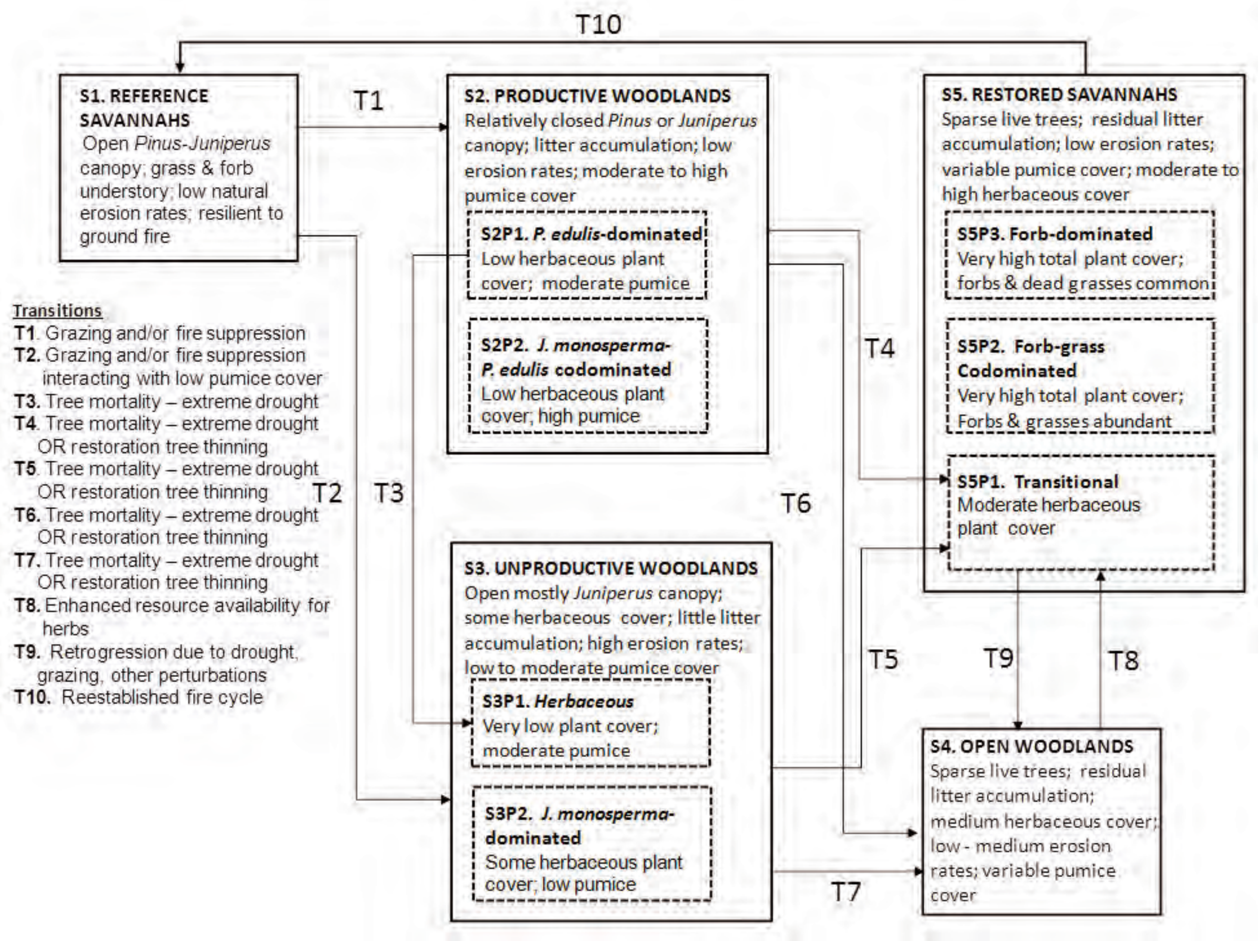

Figure 4-16. State-and-transition diagram for piñon-juniper mesa tops. Solid boxes represent ecosystem states. Dashed boxes indicate phases within states. Arrows indicate transitions. In some cases, phases within the reference state are not connected to any others by arrows; this is our method of representing spatial variants of the reference state that are determined by abiotic factors, or cases where we simply do not have a strong hypothesis for the relationship among phases.

The trajectory of piñon-juniper woodlands in Bandelier NM has diverged from reference conditions. However, it is important to note that these woodlands are heterogeneous, and not all sites have the potential to support the same desired outcome. Woodlands would have interfingered with ponderosa pine savannas which may have played a large role in spreading frequent fire to grassier, open piñon-juniper woodlands and savannas. Highly productive persistent woodlands would have been subject to low frequency crown fire. Generally speaking, many of the woodlands and savannas recently supported too much tree canopy compared to desired conditions, and some of them support much less herbaceous cover than desired. The current woodlands can be thought of as a mosaic of persistent, unproductive woodlands (averaging 21% or less tree canopy cover, up to 45% bare ground cover, and < 5% herbaceous cover), persistent productive woodlands (averaging 38% or more tree canopy cover, 17% or less bare ground cover, and < 5% herbaceous cover), open woodlands (20% tree cover, 55% litter cover), and thinning-induced savannas which have the potential to support surface fire (averaging 7% or less tree cover, and up to 32% herbaceous cover).

The first characteristic, overgrown canopy, is reversing due to three drivers: 1) intentional canopy reductions (thinning), 2) fire, and

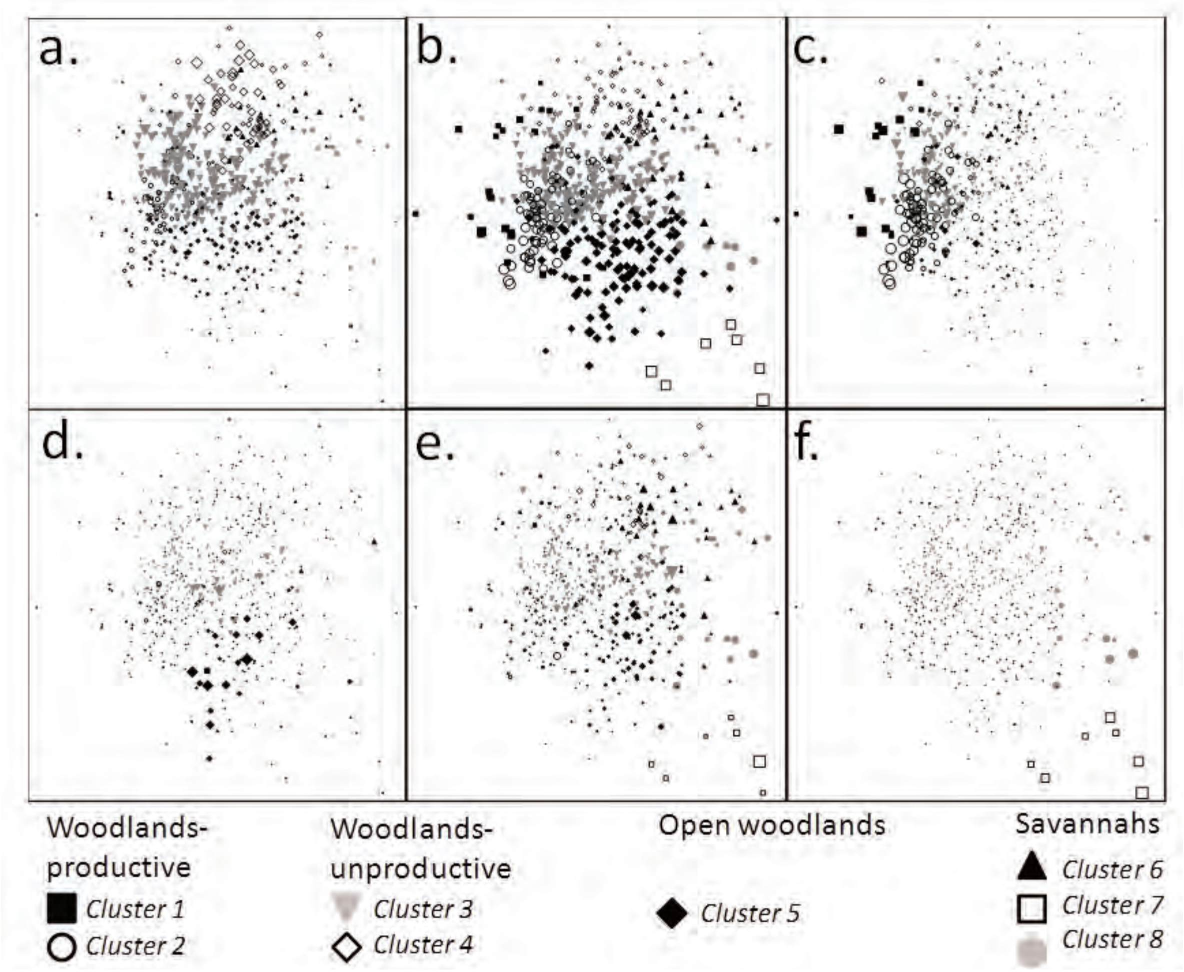

Figure 4-17. Diagrams showing six versions of the same non-metric multidimensional ordination with symbols coding for the eight clusters found in our cluster analysis. In each panel a point is a transect in a given time point; the position of each point is identical in all panels. The panels differ in that the size of symbols is scaled proportionally to a particular element of the community that is characteristic of one or more clusters. (a.) Symbols are scaled according to percent cover of bare ground, which is especially high in clusters 3 and 4. (b.) Symbols are scaled according to percent cover of litter, which is high in clusters 1, 2, 5 and 7. (c.) Symbols are scaled according to live *P.edulis* cover, which is high in clusters 1, 2, and to a lesser extent 3. (d.) Symbols are scaled according to dead *P. edulis* cover, which is high in cluster 5. (e.) Symbols are scaled according to live forb cover, which is high in clusters 6, 7 and 8. (f.) Symbols are scaled according to dead grass cover, which is high in cluster 7.

3) drought-induced dieback and mortality of trees. Thinning may induce ecosystem state transitions which are favorable, but are probably best applied to situations where the large majority of trees are younger rather than older. Stands that are primarily young are likely stands that have thickened relatively recently, and would change in a desired trajectory if thinning was applied. These areas are more likely to be former savannas or former open grassy woodlands (Jacobs et al. 2008). There are also situations where thinning is not highly likely to promote a transition, i.e. unproductive woodlands. This was actually expected to some degree. The watershed-scale thinning study attempted to avoid such areas by selecting mesatops with favorable soil types, gentle slopes, and little exposed bedrock. However, individual transects were heterogeneous mosaics of productive and unproductive types. Treatments were applied uniformly, regardless of individual transect characteristics, in the process, treating some unproductive

woodlands. This allows us to experimentally demonstrate that less productive woodlands are less responsive to canopy treatment.

There is comparatively little data on the effects of fire in piñon-juniper woodlands of Bandelier NM and environs, though various fires have burned woodlands in the park since 1996, including the Dome Fire of 1996, the 2005 Capulin Fire, the 2011 Las Conchas Fire, and various other wildfires and prescribed burns. Thinning and drought mortality make stand destroying crown fires less likely in the future because affected canopies are less continuous. The 1996 Dome Fire encompassed thousands of hectares of piñon-juniper woodlands, apparently creating some conversions to shrublands, grasslands or ruderal herbaceous cover (Figure 4-15, p. 78). However most fire-impacted stands within the Dome Fire boundary in the Bandelier NM fire atlas remained as woodlands, and apparently did not experience crown fire, though that would be the expected normal outcome of a fire in a piñon-juniper woodland. Another dataset mapping fire severity does not include most of the woodlands in question, suggesting that burning had been low severity or patchy, allowing the woodlands to persist. Another reason why there may have been relatively little conversion of woodlands to other vegetation types is that a large proportion of these woodlands in question are co-dominated by *Juniperus deppeana* and *Quercus undulata*, both resprouters post-fire. These dominant species are atypical of most of the woodlands in the park. Virtually all of these stands lost mature piñon pine between 1981 (before the fire) and 2004 (after the fire), but these transformations were almost certainly due to the drought rather than the fire.

Drought mortality in the tree canopy is the most problematic of these three drivers. Piñon-dominated woodlands in Bandelier NM and on the Pajarito Plateau recently have been decimated by the related impacts of

Figure 4-18. Landscape transformation associated with a die-off of piñon pine trees triggered by a global change-type drought. Piñon pine trees, evergreen when alive, (left) exhibiting reddish-brown foliage indicating mortality (October 2002); (right) after they have lost their needles, exposed gray trunks of standing dead tree carcasses remain (May 2004). Almost all of the surviving green trees in (b) are junipers. (Breashears et al. 2008)

drought and beetle infestation; between 2002 and 2003, over 90% of the mature piñons were killed by a combination of these factors (Figure 4-18). Despite the fact that canopy has been reduced overall, we cannot confidently say that the woodlands are moving closer to desired conditions because mature individuals of the previously co-dominant *Pinus edulis* have been lost. The future of this species is uncertain within Bandelier NM. Currently it persists in the understory as saplings established before the early 2000s mortality pulse, but because future climate is likely to bring increasingly severe and warm droughts (Williams et al. 2013), it is not clear that this species will rebound (Adams et al. 2009). Just as troubling are observations of substantial *Juniperus monosperma* mortality since 2003 in portions of the monument, documented in multiple park data sets (e.g., PJ demography plots, LTER transects, SCPN plots). If both species are lost, affected stands will effectively undergo a shift in growth form dominance to grasslands and/or shrublands.

The second major element of these ecosystems is the understory and its control over accelerated erosion (Davenport et al. 1996). The effect, at the scale of a whole watershed, of a thinning and slash redistribution study on promotion of understory herbs is well documented (Chong 1993, Hastings et al. 2003, Jacobs 2002, Loftin 1999). We interpret successful promotion of understory using slash redistribution as an ecosystem state change—a transition to a structural and functionally distinct ecosystem.

This state change is very much in line with the desired conditions of more open woodlands with herbaceous understories. Our examination of long-term transects confirms that greater development of a grassy understory is associated with less emitted sediment. In our analysis of the SCPN monitoring plots, greater soil aggregate stability seems to be promoted by greater canopy cover, despite that thicker-canopied woodlands tend to yield more sediment in erosion events (Hastings et al. 2003). Though this was not the expected result, it likely indicates that most sediment yield in the park is regulated more by the connectivity of patches which can intercept and store runoff (Davenport et al. 1996) than by the erodibility of soil per se.

In general, the literature and our analyses of existing data indicate that: 1) herbaceous understories can be promoted by slash redistribution, though this does not always happen, 2) erosion is reduced by herbaceous understories and slash, and 3) promotion of herbaceous understories and reduction of erosion are consistent with desired trajectories for the piñon-juniper ecosystem.

Since a large proportion of the park's piñon-juniper woodlands have undergone thinning and slash redistribution, can the understory stratum of these woodlands be said to be on a desired trajectory? There are reasons to be optimistic, but recent drought conditions cast some doubt on the permanence of older treatments, and whether similarly successful results will continue to be replicable. Bowker and Smith (2014) demonstrate that understory productivity is just as tightly controlled by climate as by overstory reduction and slash redistribution. A fortuitous redistribution of slash during a wetter than average period is most likely to promote understory production because two constraints (adequate moisture, and unstable soil surfaces) are simultaneously omitted.

However for reasons poorly understood, wet years in the recent past seem to lead to lesser understory production. Long-term transects suggest that established herbaceous understories can be compromised by extreme drought, and that, proportionately, grasses may experience more cover loss than the tree canopy (Figure 4-19). Other results from the small-scale thinning plots of Jacobs and Gatewood (1999) seem to directly conflict with this observation. Most of the grass species would be able to rebound from a drought die-back due to vegetative growth from surviving patches, but it is not known how resilient these herbs are to repeated drought events which may be more com-

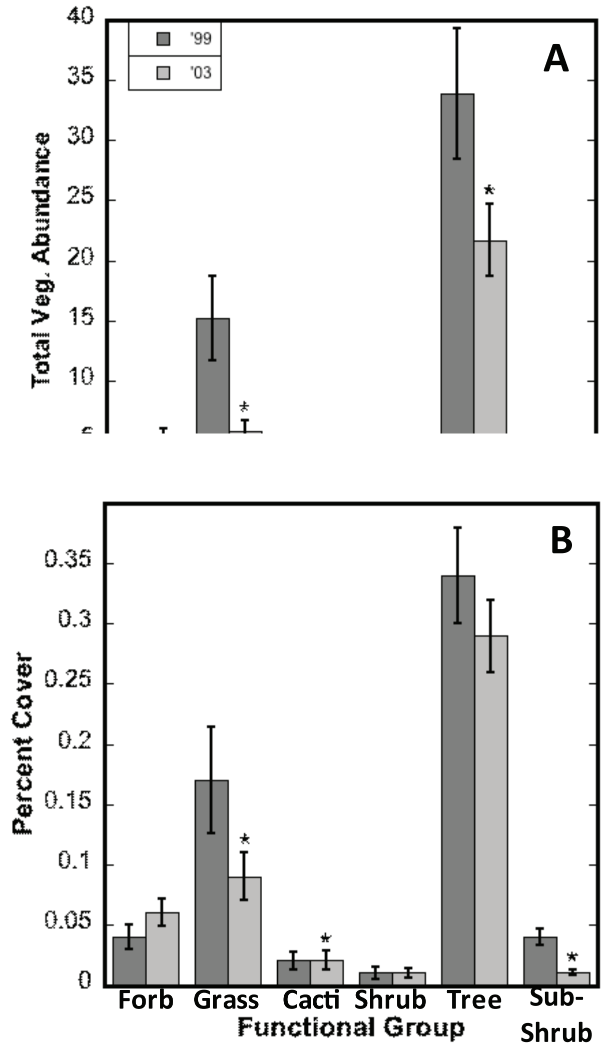

Figure 4-19. Bar graphs showing the mean and standard error for total (A) and percent (B) abundance of individual functional groups. Asterisks denote significant differences between years. Even though it is difficult to tell from the graph, the percent cacti is different in the two years (means are .0186 and .0249 for 1999 and 2003, respectively).

mon in the future (Karl et al. 2009). Some short-lived forbs produce large quantities of long-lived seed in favorable years, thus may retain considerable resiliency. As drought eliminates more and more canopy, the understory will become increasingly important. Since herbaceous plants are less long-lived, and leave less recalcitrant residues, we can expect an emerging dynamic ecosystem that varies strongly from wet years to dry years.

We would conjecture that very high herbaceous biomass would be possible in wet years, whereas much reduced cover and accelerated erosion could characterize drought years.

4.06.5 Level of confidence

Reference conditions are poorly understood due to successive intensive human uses, high confidence for last approximately 300 years of history and moderate confidence for trajectory.

4.06.6 Data gaps/Research needs/ Management recommendations

- Develop fire management recommendations for treated piñon-juniper woodland areas to promote grassy understory and maintain open canopy.
- Continue SCPN monitoring of piñon-juniper woodlands.
- Develop remote sensing monitoring methods to track post-treatment vegetation response.

4.06.7 Sources of expertise

Matthew Bowker and David Smith

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4.06-1 Erosion in Piñon-Juniper Woodlands

4.06-1.1 Description

Accelerated rates of soil erosion within large portions of the semi-arid piñon-juniper woodland zone of Bandelier NM (occurring primarily in mesa-top settings between 1830–2135 m [6000-7000 ft] elevation) were first identified as a management issue in the 1970s in connection with early soil mapping and efforts to control a feral burro (*Equus asinus*) population (Earth Environmental Consultants 1978, Chong 1992). However, it was not until the mid-1980s that park-wide archaeological survey efforts began to systematically document erosional impacts to cultural sites. At the same time, a graduate student researcher (later the park ecologist) articulated the erosional problem in piñon-juniper systems (Allen 1989); and these and other efforts cumulatively presented the widespread impacts that erosion was having on park resources.

In the early 1990s, long-term monitoring of vegetation and erosion within the woodland zone was initiated. The goal was to determine whether erosion was accelerating or the systems were still recovering from prior landuse impacts. A number of focused research efforts were also initiated to quantify rates of runoff and erosion within woodlands (Wilcox et al. 1994, 1995a, 1995b, 1996a, 1996b; Wilcox et al. 2002), assess methods to mitigate runoff and erosion (Chong 1993, Chong 1994, Hastings et al. 2003, Jacobs 1999, Jacobs and Gatewood 1999, Jacobs et al. 2002b), and reconstruct historical vegetative and soil conditions (Allen and Breshears 1998, Allen 2004, Julius 1999,). Several collaborative efforts attempted to provide ecohydrologic frameworks linking observations and data, arguing that the spatial arrangement and critical threshold of effective cover within an area could be used to model hydrologic response in degraded woodland systems (Davenport el al. 1998).

From 2007–2010, nearly 5,000 acres (constituting about ½ of the total woodland acreage within the park, and representing nearly all of the woodland area deemed favorable for mechanical restoration treatments, minus several hundred acres of uncut control) were mechanically treated using a thinning-slash restoration method. This method was developed and tested at several scales, including within a 100-acre paired watershed study area (ref; Hastings et al. 2003). Results from these studies demonstrated that mechanical thinning and slash treatments increase understory cover by 3-fold, while reducing runoff and sediment yields by 1–2 orders of magnitude (Hastings et al. 2003, Jacobs et al. 2002b).

History

Most researchers agree that prior to Euro-American settlement the region of the Jemez Mountains now occupied by Bandelier NM was predominantly an open woodland system that was abandoned around 1600 after several hundred years of intense human occupation by Ancestral Pueblo farmers. For the next 200 hundred years, until around 1800, the area recovered in part, and presumably both tree and herbaceous understory cover increased in many locations. In ponderosa pine communities, conditions were somewhat different, as a frequent surface fire regime may have kept productive grassy savanna areas more open and woodlands restricted to shallow rocky soil locations (Bowker and Smith 2014).

Arguably the biggest changes to the landscape took place when Euro-Americans arrived in the area. Spanish colonists in the 1600s brought sheep, cattle, goats, horses, and burros, along with tools such as metal axes. What had been small-scale use of the Bandelier NM area for livestock changed dramatically with when the railroad arrived at Buckman in the 1880s and very heavy grazing ensued, promoted at the federal level. This quickly proved unsustainable. The railroad also turned what had been sporadic timber collection into a full-fledged industry, though the inaccessible terrain on the southern part of the plateau to some extent limited the exploitation of the lands that would became the monument. Even so, the combination of grazing and logging proved devastating for the fragile soils of the Pajarito Plateau, and began a cycle of erosion that continues to this day. Research indicates that a pattern of increasing woodland cover reinforced initial grazing impacts on the understory and facilitated initiation of accelerated surface runoff and soil erosion.

An intense multi-year drought during the mid-1950s shifted the distribution of ponderosa pine communities several kilometers up-mesa, representing a shift up in elevation at the lower end of ponderosa distribution. This shift is now thought of as a tipping point when ecohydrologic degradation processes in the woodland (piñon-juniper) system became self-reinforcing (Allen and Breshears 1998, Bowker and Smith 2014d). During the 1970s, a feral burro population was likely a contributing factor in woodland soil erosion (Allen 1989). In the 1990s large populations of elk and deer further increased soil erosion (Chong 1992). More recently a dry period that began around 1996 and intensified with several severe multi-year droughts (2000–2004, 2010-present) resulted in widespread mortality of piñon (exceeding 90%). However, there has been a piñon seedling response in many locations.

Finally, fire has become a more common disturbance in regional woodlands with increased severity and extent. Consequently, post-burn woodlands are initially somewhat sterile and may be more vulnerable to weed invasion and accelerated erosion. However, prior mechanical treatments may prime the systems response to burning and promote post-fire recovery of the understory (Jacobs and Gatewood 2002a).

4.06-1.2 Reference conditions

Erosion levels were likely low due to large amounts of herbaceous cover in a landscape dominated by savanna type woodland with piñon-juniper and ponderosa pine trees. A rough estimate of pre-grazing erosion rates can be inferred from treated areas where sediment production was reduced by an order of magnitude relative to untreated control (Hastings et al, 2003).

4.06-1.3 Data and methods

Surface runoff and soil erosion has been assessed by a variety of methods and at several spatial scales. The most reliable methods are based on direct measurements of vegetative cover, precipitation, and runoff and sediment transport using instrumented watersheds with line transects, rain gauges, flumes, and sediment traps. Integrated ecohydrologic measures have been collected primarily at two study sites located on the mesa immediately south of park headquarters. These provide comparative data across multiple spatial scales within untreated woodlands (Frijolito Experimental Watershed) and across matched treatment-control areas (Paired Watershed Study). Data collected within these discrete study areas are then incorporated into predictive models that can infer hydrologic condition indirectly using measures of effective soil cover.

Vegetation cover (% of total) by species was collected for basal intercept and canopy overstory along permanent line transects. Sediment production (kg-ha) was estimated using sediment traps for defined contributing areas (i.e. 0.1-ha microwatersheds).Runoff (cfs) was estimated using instrumented flumes for defined contributing areas (i.e. 0.3-ha microwatersheds). Detailed data and methods for individual studies are documented in the cited references.

4.06-1.4 Condition and trend

After the 1950s drought, through the mid-1990s, piñon-juniper woodlands increased in both extent and density, in part facilitated by an unusually wet period during the 1980s. By the time long-term monitoring efforts were initiated in the early 1990s, average woodland tree densities exceeded 1000 stems/ha, compared to forest age reconstructions that suggested tree densities were <250 stems/ ha prior to 1850 (Julius 1999). Higher tree densities were accompanied by a generally sparse herbaceous understory cover.

Recent measures of runoff and erosion suggest that rates are highly accelerated because extrapolated over time, current rates yield unsustainable levels of sediment production. For example, at measured soil-loss rates of ~10 mm/decade (Wilcox 1994), the existing soil resource in these locations, which is less than 10 cm, could be eliminated within one century.

The long-term benefits of landscape-scale woodland restoration treatments in the face of developing warmer and drier conditions are unclear. Recent accelerating climate-induced changes in woodland overstory include increased fire activity and near complete loss of mature piñon. Partial, but ongoing and progressively more expansive die-off of juniper overstory due to drought stress suggests that type conversion of this system to more of a scrubland (sparse grass and shrub cover with scattered trees) could happen within 20-50 years (Bowker and Smith 2014).

4.06-1.5 Level of confidence

Historical reference conditions are inferred from a number of lines of evidence, but these are mostly indirect and confidence is generally fair to poor. Measurements of soil erosion are highly variable from year to year and driven by the relative strength of summer monsoonal patterns with a few large events producing most of the observed erosion in a given year. Hydrologic measures are extremely scale dependent, with the response mediated by antecedent conditions and subject to threshold effects. Current conditions and response to restoration treatment are well documented for a few well-studied locations, but future trends are highly speculative relative to anticipated effects of climate change on this system.

4.06-1.6 Data gaps/Research needs/ Management recommendations

- Reconstruct the temporal and spatial patterns of recent drought-beetle induced tree mortality.
- Characterize ecohydrologic system response to cumulative effects of mechanical thinning, drought, and fire disturbance.
- Implement long-term monitoring of the restoration response using remote sensing methods to complement ground based (erosion and vegetation) measurements.
- Conduct research to assess the role of biological soil crusts in degraded and restored semi-arid woodland systems.

Going forward, we are developing new methods to remotely sense understory response to restoration treatment at landscape scales using satellite imagery which would support monitoring of the woodland soils and embedded cultural resources. Monitoring of the entire 5,000-acre treatment footprint will allow management to determine spatial and temporal patterns of treatment response and identify locations which may benefit from supplemental treatment.

4.06-1.7 Sources of expertise

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Biological Integrity • Species of Management Concern – At-risk Biota

4.07 Rare Plant Species

4.07.1 Description

Five plant species of management concern have been identified by monument staff because they require monitoring due to their restricted distribution and/or existing threats to their persistence in the park (Brian Jacobs, personal communication). At present, no federally-listed species occur in the park, but several are (or have been) identified by the State of New Mexico as endangered, or by NatureServe as imperiled or vulnerable (http://www.natureserve.org). See Appendix D for a complete list of species of management concern for Bandelier NM.

All of the mesic species (yellow ladyslipper, rattlesnake orchid, grape fern, wood lily and cerro hawthorn) have been significantly impacted by recent fire, flood, and drought events, to the point where source material (seeds, spores, rhizomes and bulbs) may now be extremely rare, or even extirpated within the park. In addition, the historic riparian landscape where these species were once found has been permanently altered. Consequently, in establishing reference conditions for these species, managers should consider that many sites that once supported them are now in the very early stages of secondary successional processes. It may or may not be reasonable to consider long-term measures of abundance and distribution that fall within the historic ranges of variation noted for each species. For the near-term, the park may simply want to document the presence or absence of each as riparian systems recover. Reference conditions for the single xeric species (grama grass cactus) are likewise problematic due to the absence of existing information on current or historic populations.

Regardless of current ecological condition, negative anthropogenic impacts (e.g. from grazing, recreation, collecting) to habitats where populations of these species are currently or have historically been found, would indicate some level of impairment.

4.07.2 Reference conditions

All of these species were likely present in appropriate settings, but locally rare even prior to Euro-American contact primarily due to restricted habitat requirements

4.07.3 Data and methods

Data exist for the abundance or distribution of the species described below. In addition, previous surveys have been conducted and local botanists and park staff are generally familiar with historic locations and the general number of individuals that were present at each site in past years.

Data from surveys conducted in the late 1980s for grama grass cactus in the Tsankawi Unit of the park, and from ladyslipper and grape fern surveys in the 1990s, are available in Bandelier NM. Plot data collected during the pilot study for the Southern Colorado Plateau Network (SCPN) integrated riparian protocol (Scott and Reynolds 2007) in Capulin Canyon did not include observations of any of the mesic species of concern. Plot data collected for the park's vegetation map (Muldavin et al. 2011) likewise did not include any of these species. All other information on abundance and location comes from park staff observations.

4.07.4 Condition and trend

Greater or large yellow ladyslipper (*Cypripedium parviflorum* var. *pubescens*)

Description

Cypripedium parviflorum var. *pubescens* (yellow ladyslipper) is an orchid that produces one or two bright yellow flowers between May and August. Plant height ranges from 10–50 cm, and the large, showy lip petal of the flower is between 15 and 55 mm long. The species can reproduce

both vegetatively from rhizomes, (so that individual plants observed growing close together are likely one clone), and sexually (Deller). Reproduction by seed is rare, however, because seeds are very small and dependent on mycorrhizal fungi for germination, and because seedlings that do emerge rarely survive (Deller). Rhizomes may be viable below ground even though they do not produce flowers every year (Deller).

Distribution

Greater yellow ladyslipper is widely distributed throughout North America, occurring across much of Canada and in 42 of the lower 48 states. In New Mexico it has been documented in both the Jemez and the Sangre de Cristo Mountains at elevations between 1,830 and 2,900 meters (Coleman 2002).

Ecology

This species tolerates shade to nearly full sun conditions in fairly open sites within riparian-associated forest communities, meadows and clearings. Individual plants are found in generally mesic conditions but not directly adjacent to water, and prefer neutral to slightly acidic soils. Many of the populations documented in the monument prior to the Las Conchas Fire in 2011 were located near small spot fire locations from the 1977 La Mesa Fire (Brian Jacobs, personal communication).

Status

NatureServe assesses the global status of this species as secure and the national status as apparently secure, though there are currently few documented occurrences of large and secure populations anywhere in North America (NatureServe). The species is classified by NatureServe as imperiled in New Mexico where it was previously listed as endangered. It is listed by Los Alamos National Laboratory (adjacent to the park) as a sensitive species (Hathcock et al. 2010). Extirpation of two populations has been documented in Arizona and populations in New Mexico and Arizona are considered to be in decline. In Bandelier NM most, if not all, of the known populations were likely destroyed by floods and debris flows in 2011 and 2012. Some individual clones may have survived in areas above flood zones, but if so their locations are at present unknown and it will likely be some time until they can be discovered (Brian Jacobs, personal communication).

Threats

The high-intensity fires and post-fire floods and debris flow events that followed the Las Conchas Fire in 2011 destroyed much of the riparian habitat upon which this species depends. While occasional low-intensity fires may provide areas of open habitat for ladyslippers, the extreme fires that have occurred recently have probably eliminated most, if not all, of the previously documented occurrences of this plant along Capulin and Frijoles creeks (Brian Jacobs, personal communication). Because flowering individuals re-sprout from rhizomes and sexual reproduction is rare, significant re-establishment from seed is unlikely for this species. Continued drought may further restrict population growth if it alters the mesic environment this plant requires.

Condition and trend

The current condition of this species in the park is poor; the 2011-2012 floods probably destroyed or buried most of the existing individuals, source populations are extremely limited, and habitat has been altered or lost. Trend is a steep decline precipitated by the events of 2011-2012.

Rattlesnake Orchid (*Epipactis gigantea Douglas* ex Hook.)

Description

Epipactis gigantea is an erect perennial reaching anywhere from 30 centimeters to one meter in height. Its stems have wide or narrow lance-shaped leaves 5 to 15 centimeters long and inflorescences of two or three showy flowers near the top. Each flower has three straight sepals which are light brownish or greenish with darker veining, each one to two centimeters long. The fruit is a hanging capsule 2 or 3 centimeters long which contains thousands of tiny seeds.

Distribution

Widespread and locally abundant in suitable habitats, native to western North America from British Columbia to central Mexico

Ecology

This plant grows in wet areas in a variety of wetland settings; including riverbanks, hot springs, and meadows.

Status

Populations of this species are widespread and abundant across western U.S. but local populations have been impacted by recent widespread fire and flood disturbance. NatureServe assesses the status of this species as secure nationally and it is has no protected status in New Mexico.

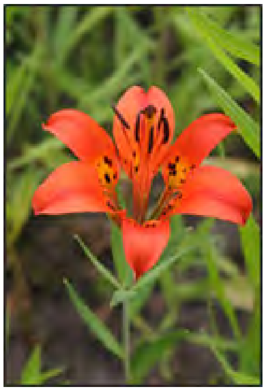
Threats

Spring habitats within White Rock Canyon were destroyed during reservoir holding events during the mid-1980s. Recent high-intensity fires and post-fire floods and debris flow events have subsequently degraded much of the riparian habitat upon which this species depends. Continued drought may further restrict population growth if it alters the mesic environment this plant requires.

Resource condition and trend

The current condition of this species in the park is poor; the 2011-2012 floods likely destroyed or buried most of the existing individuals, source populations are extremely limited, and habitat has been altered or lost. Current population trends are on a steep decline.

Wood Lily (*Lilium philadelphicum* var. *andinum*)



Description

The wood lily is a geophytic monocot that flowers from June to August. A single stalk grows up to 0.5 meters and produces one to three orange-red flowers.

Distribution

This species is distributed across most of Canada and much of the eastern, midwestern, and Rocky Mountain regions of the U.S. It is rare, however, in New Mexico and Colorado, and absent from Utah and Arizona. In New Mexico it is patchily distributed in only four or five counties. In the Jemez Mountains it ranges from 2,130 to 2,440 meters in elevation, and in the monument is found only in upper Frijoles Canyon.

Ecology

In the southwest this lily prefers moist, open understory habitats in mixed conifer riparian forests and higher mountain meadows. Periodic low-intensity fires may enhance habitat by providing openings in forests canopies.

Status

NatureServe assess the global and national status of this species as secure. In New Mexico it is assessed as vulnerable, and was previously listed by the state as endangered. Wood lilies were formerly common in the upper Frijoles riparian zone but few individuals have been observed since the events of 2011 (B. Jacobs).

Threats

Similar to the species described above, recent fires and floods in riparian zones have likely extirpated most of the previously known wood lily populations. If populations do re-establish, threats noted to wood lily populations in other locations have included woody encroachment and risk from collectors.

Condition and trend

The condition of this species in Bandelier NM is currently poor. As with the ladyslipper and the grape fern, the riparian habitats upon which wood lilies depend have been greatly altered and existing populations largely destroyed. Given their much higher reproductive rates from seed, wood lilies may be more likely than ferns or orchids to re-establish if conditions are good. Trend is a steep decline, precipitated by the events of 2011-2012.

Cerro hawthorn (*Crataegus erythropoda*)

Description

This shrub or small tree grows to a maximum height of about five meters. Plants have dark green leaves and thorny branches. Clusters of white flowers are produced in late spring, and the fruit (berries) are palatable to birds and rodents. Due to its relatively low stature in riparian woodlands, individual hawthorns also provide important habitat for songbirds and other small vertebrates. The species is often cultivated, and appears to have a fairly wide range of drought and wind tolerance in urban settings.

Distribution

This species is only found in the Rocky Mountain states of Colorado, Arizona, New Mexico, Wyoming and Utah. In the Jemez Mountains it is found between about 2,130–2,440 meters in elevation, and within Bandelier NM it occurs in upper Frijoles Canyon.

Ecology

Cerro hawthorn is found in relatively open, well-drained sites within riparian woodlands. It is not found where there is a closed canopy above it.

Status

NatureServe assesses the global status of this species as secure. It is assessed as vulnerable in Colorado, Arizona and New Mexico, and critically imperiled in Wyoming and Utah. It may have been extirpated from Bernalillo and Sandoval counties in New Mexico. Prior to 2011 it was uncommon in the monument, but locally abundant in moist areas of upper canyons (NPS 2007). Though directly impacted by the 2011 and 2013 flood events, re-sprouting by several individuals has subsequently been observed (B. Jacobs).

Threats

Relatively little seems to be known regarding the status of this species in the wild or threats to it throughout its range. Many individuals in the Frijoles drainages were either damaged or killed in the Las Conchas Fire, though re-sprouting is occurring. Minimal published information is available on the ability of this species to recover from fire. Populations in New Mexico seem to be scattered, and the general lack of information on the species has been identified as a concern.

Condition and trend

The condition of this species in the park is likely poor, given that it was only known from riparian sites that were both burned and flooded in 2011. The number of surviving individuals is likely quite low. Without surviving reproductive individuals, significant population growth in the short term is unlikely. Trend is likely a steep decline precipitated by the events of 2011-2012, though it was probably historically rare.

Gramma Grass Cactus (*Sclerocactus papyracanthus*)

Description

This small cactus is a perennial, but relatively short-lived species that grows only to about 8 cm in height. Small, white flowers are produced in April or May, but during the remainder of the year the plant is cryptic, covered with papery-like spines, making the plant appear similar to the gramma grass (Bouteloua spp.) with which it is often associated.

Distribution

Gramma grass cactus is distributed across New Mexico and adjacent portions of Arizona and Texas at elevations between about 1,525 and 2,290 meters.

Ecology

This species prefers open sites in fine, sandy clay loams on basaltic canyon rims and mesa tops, usually in piñon-juniper woodlands and desert grassland habitats. In Bandelier NM, the plant is found at lower elevations in open and grassy piñon-juniper habitats (NPS 2007) but basalt exposures and associated soils that this plant prefers are relatively rare in the main park unit (B. Jacobs).

Status

NatureServe assesses this cactus as critically imperiled in Texas, imperiled in Arizona, and apparently secure in New Mexico, though it may have been extirpated from Cibola and Grant counties. This is inherently a cryptic species within the habitats where it is found. (This species was reported as occurring fairly frequently during the SCPN upland vegetation monitoring in 2009 [DeCoster and Swan 2011], however, in 2013 these occurrences were field-checked and found to represent a mixture of Echinocereus and Escobaria species. The identification of this species in the upland vegetation plots has since been corrected. Sclerocactus papyracantha has only been encountered in one plot [42] during upland monitoring from 2008 to 2010). A project to enhance a population of this cactus in the monument was undertaken in 1989 when hundreds of individual plants were transplanted to the park's Tsankawi unit (from a salvaged site outside the park), however, none of the plants survived after five years.

Threats

Current threats to this species within the park are unknown. Given likely habitats that exist in exposed soils, erosion may be a factor. In other locations the species is at risk from cattle grazing and off-road activities, but these impacts are not a factor in the park.

Condition and trend

The status of this species in the park is generally unknown. Assuming that plants were historically common prior to grazing but are now rarely observed, the condition is likely poor. The trend is unknown.

Grape or Rattlesnake Fern (*Botrychium virginianum*)

Description

This small deciduous fern grows to about 30 cm in height. The plant produces two leaf types: a single sterile leaf which is pinnately divided into subleaflets and appears early in the spring and lasts through the summer, and a shorter, fertile leaf which produces sporangia and is short-lived compared to the sterile leaf. Wind-dispersed (anemochorous) spores are released during the early summer.

Distribution

Grape ferns are widely distributed across North America and exist often at relatively high elevations.

Ecology

In Bandelier NM, this fern is found in habitats similar to those described for the ladyslipper: shady to mostly-sunny open patches and bogs in riparian mixed conifer forests. Prior to the 2011 floods it was only known to occur in one bog location that was subsequently destroyed (Brian Jacobs, personal communication).

Status

NatureServe assesses the global and national status of this species as secure. The status of the species in New Mexico is unknown, but it is listed as critically imperiled in California, Arizona and Colorado. Known locations for this species in the park were inundated by the post-fire floods and debris flows in 2011 and 2012, likely extirpating grapefern from the park. Targeted surveys in 2013 have found no occurrences remaining (B. Jacobs).

Threats

Catastrophic high-intensity fires and associated floods, such as those that have occurred recently have probably eliminated the populations that previously existed along Frijoles creeks (B. Jacobs). Like the ladyslipper orchid, the absence of a source population for this species greatly reduces the likelihood of short-term re-colonization at historic sites. Because spore producing plants are particularly dependent on moist habitat conditions, continued drought may also impede establishment in this species.

Condition and trend

The current condition of this species in the park is poor, though it may have always been rare. The riparian habitats in which this fern was found have been greatly altered or destroyed and there is no regional source population. Trend is a steep decline precipitated by the events of 2011-2012.

4.07.5 Level of confidence

Confidence in the presence or absence of the mesic species at likely locations is fairly high given the existing knowledge of park staff and ongoing monitoring efforts. Confidence in the status of the riparian species in relation to successional processes, however, will probably be very low for the foreseeable future. Confidence in the condition or trend of grama grass cactus populations is low.

4.07.6 Data gaps/Research needs/ Management recommendations

The park botanist conducts annual informal walking surveys of existing sensitive plant populations and potential habitat primarily to assess habitat condition, numbers of individuals and reproductive status at known locations, and to opportunistically census presence of new individuals in potential habitat. If status of any of these species was elevated to federally listed, the park would work with the Natural Heritage Program of New Mexico and the USFWS to conduct more formal assessments of suitable habitat and population status.

4.07.7 Sources of expertise

Brian Jacobs, Megan Swan

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Web Sites

NatureServe: http://www.natureserve.org (accessed July-August 2013)

Cypripedium parviflorum var. *pubescens*

http://plants.usda.gov/core/profile?symbol=cypap3

Botrychium virginianum

http://eol.org/pages/597549/overview

http://www.natureserve.org/explorer/servlet/NatureServe?searchName=Botrychium%20virginianum

Lilium philadelphicum var. *andinum*

http://nmrareplants.unm.edu/droplist/liland.htm:

Crataegus erythropoda

http://pick4.pick.uga.edu/mp/20q?search=Crataegus+erythropoda&guide=Trees&flags=not_no:#http://nmrareplants.unm.edu/droplist/craery.htm

Sclerocactus papyracanthus

http://plants.usda.gov/core/profile?symbol=SCPA10

http://www.efloras.org/florataxon.aspx?flora_id=1&taxon_id=242415297

4.08 Nonnative Plants

4.08.1 Description

Exotic or non-native plant species are those that are not naturally a component of pre-Euro-American settlement ecosystems. However not all exotic species necessarily pose a threat to existing communities. For management purposes exotic species can be categorized by their degree of invasiveness— ecologically how well they out-compete native species— and/or how well they can be controlled. For example, while non-native ornamental species are often easily controlled if needed, they usually do not compete well with native species, and so pose little threat to native populations. Such species consequently may be of lower management concern than those that are more aggressive and survive unaided under natural conditions.

In Bandelier NM, exotic species comprise approximately 15% of all the plant species found within the park. However, <5% (e.g. ~40 of ~800 taxa) are considered invasive (NPS 2006). Of those, only about 10-12 species are both high risk and easy to treat. Several species are considered nuisance species in that they are common and fairly wide-ranging, usually in disturbed areas, but they are either weak competitors or treatment on a large scale is not practical. The most challenging management species are those that are so invasive that they are ubiquitous across the landscape and very difficult to constrain. The common and widespread exotic plant species found in the monument are listed in Table 4-2 along with basic information about their ecology and status, if known.

Post-disturbance conditions, in particular those that result from large wildfires, often facilitate the expansion of exotic grasses and other invasive species (Keeley 2006; Balch et al. 2013). Additional factors, such as pre-fire landcover, fire intensity, elevation and climate, also affect post-fire abundance of exotic species (Keeley and McGinnis 2007; Fornwalt et al. 2010; Sherrill and Romme 2012; Shive et al. 2013). *Bromus* species have invaded and now dominate many arid-ecosystems across the west, further altering natural fire regimes (D'Antonio and Vitousek 1992). *Bromus tectorum* (cheatgrass), *B. inermis* and *B. japonicas* are present in the monument.

4.08.2 Reference condition:

No nonnative plants present.

4.08.3 Data and methods

At present, information on exotic species distribution and abundance comes from 1) the Exotic Plant Management Team (EPMT) work that is conducted annually, 2) anecdotally from park staff and cooperators, and 3) incidentally from upland vegetation monitoring. Post-Las Conchas Fire data on vegetation have been collected but are not yet available.

4.08.4 Condition and trend

Bromus tectorum (cheatgrass), *B. inermis* and B. *japonicas* are present in Bandelier NM, and appear to be increasing in distribution. Cheatgrass has expanded across the lower elevations of the park, likely facilitated by both drought and fire conditions of recent decades (B. Jacobs). The other two species are less abundant but possibly increasing as well. Given the enormous impacts of recent fires, these species should be of particular concern to managers.

Several additional weed populations (e.g., *Cirsium vulgare*, *C. arvense, Carduus nutans, Cardaria draba*), around the main headquarters and visitor service areas have been largely extirpated by EPMT efforts (B. Jacobs). Fire impacts have prevented most work from being conducted in the riparian corridors, so very little is known about invasive plant changes in those areas.

Fires in the southwest are probably going to increase with changing climate. Along with predicted increasing temperatures and continued drought conditions in this region, the potential for additional invasions and persistence of exotic species is high (Hurteau et al. 2014).

4.08.5 Level of confidence:

The level of confidence for this analysis is low to moderate.

4.08.6 Data gaps/Research needs/ Management recommendations

- Additional surveys for noxious species in riparian and burn areas should be conducted.
- The use of remote sensing data, now available at relatively finer scales than used previously, could be considered for detecting new invasions (He et al. 2011).
- All prescribed fire plans and wildfire management and recovery actions should consider the potential for post-fire invasive species impacts, as well as the ability to couple fire impacts to invasives with re-introduction and/or facilitation of native propagation (Keeley 2006; Keeley and McGinnis 2007).

4.08.7 Sources of expertise

Brian Jacobs

4.08.8 Literature cited

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Table 4-2. Common and widespread nonnative plant species found in Bandelier National Monument, their location, and status post-Las Conchas Fire (LCF), if known; website provided if available. Organized according to type of nonnative species.

Scientific name common name	Type	Locations in Bandelier NM	Comments/post-LCF status, if known
Invasive non-native plants for which there is an effective suppression method			
Ailanthus altissima[a] Tree of Heaven	woody	Lost Canyon (approx. 2 acres), and previously extirpated from historic orchard	often reproduces by root sprouting so is often difficult to treat; highly adapted to disturbance; early successional species following fire but can persist in shade once established; treated in Lost Canyon area by EPMT in 2013
http://www.fs.fed.us/database/feis/plants/tree/ailalt/all.html			
Elaeagnus angustifolia[b] Russian olive	woody	Rio Grande Corridor and scattered swales; lower tributary canyons and mouths (and associated willow patches) including Frijoles, Lummis, and Alamo; Developed District	common and spreading throughout the southwest
http://www.fs.fed.us/database/feis/plants/tree/elaang/all.html			
Ulmus pumila[b] Siberian elm	woody	RGC and roadsides; lower tributary canyons and mouths (and associated willow patches) including Frijoles, Lummis, and Alamo; Developed District	reproduces by seed and root sprouting
http://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5410128.pdf			
Tamarix ramosissima[c] salt cedar/tamarisk	woody	RGC; lower tributary canyons and mouths (and associated willow patches) including Frijoles, Lummis, and Alamo.	resprouts from root meristem after fire (Drus 2013);
http://www.fs.fed.us/database/feis/plants/tree/tamspp/all.html			
Linaria vulgaris[a] butter and eggs (yellow toadflax)	perennial forb	RGC, and Capulin mesa; lower tributary canyons and mouths (and associated willow patches) including Frijoles, Lummis, and Alamo	infestations are expanding in New Mexico (USFS 2012); treated on Capulin Mesa by EPMT in 2013
http://www.fs.fed.us/database/feis/plants/forb/linspp/all.html http://www.invasivespeciesinfo.gov/plants/toadflax.shtml			
Cirsium arvense[a] Canada thistle	perennial forb	RGC; lower tributary canyons and mouths (and associated willow patches) including Frijoles, Lummis, and Alamo.	can produce new shoots from roots; treated along the Highway 4 corridor by EPMT in 2013
http://www.fs.fed.us/database/feis/plants/forb/cirarv/all.html			
Lepidium latifolium[a] peppergrass	perennial forb	RGC; lower tributary canyons and mouths (and associated willow patches) including Frijoles, Lummis, and Alamo.	
Centaurea repens Russian knapweed	perennial forb	RGC; lower tributary canyons and mouths (and associated willow patches) including Frijoles, Lummis, and Alamo	reproduces from seeds and vegetative buds on roots
http://www.colostate.edu/Depts/CoopExt/TRA/knapweed.html			
Invasive non-native plants for which there is currently no effective suppression method			
Bromus tectorum[b] cheatgrass	annual grass	widely distributed at lower elevations, expanding into canyons and possibly burned areas post-LCF	winter annual, so abundance dependent on winter-spring rainfall; expanding, especially in post-fire areas
http://www.fs.fed.us/database/feis/plants/graminoid/brotec/all.html			
B. japonicas Japanese brome	annual grass	widely distributed in disturbed areas near horse corral, developed park areas, Rio corridor, and lower canyon slopes	
http://www.fs.fed.us/database/feis/plants/graminoid/brojap/all.html			

Table 4-2. Common and widespread nonnative plant species found in Bandelier National Monument, their location, and status post-Las Conchas Fire (LCF), if known; website provided if available. Organized according to type of nonnative species (*continued).*

Scientific name **common name**	**Type**	**Locations in Bandelier NM**	**Comments/post-LCF status, if known**
B. inermis smooth brome	perennial grass	roadsides, Dome Fire area	spreading vegetatively
http://www.npwrc.usgs.gov/resource/plants/exoticab/pipebrom.htm			
Widespread nuisance species			
Cirsium vulgare[b] bull thistle	biennial forb	local patches in Cerro burn area; disturbed or burned meadow areas along Hwy 4	treated along the Highway 4 corridor by EPMT in 2013
Carduus nutans[a] musk thistle	biennial forb	roadside, Hwy 4	post-fire invader (Floyd-Hanna et al. 1993); reproduces only from seed; treated along the Highway 4 corridor by EPMT in 2013
Kochia scoparia kochia	annual forb	RGC and disturbed sites	
Kali tragus Russian thistle/ tumbleweed	annual forb	disturbed sites	
Taraxacum officinale dandelion	perennial forb	riparian, canyons, and lawns	
Verbascum thapsus mullein	biennial forb	disturbed and recently burned areas	treated on Capulin Mesa by EPMT in 2013
Chenopodium glaucum goosefoot	annual forb	disturbed and recently burned areas	
Poa pratensis blue grass	perennial grass	riparian, lawns, developed canyon area; persisting where planted	
Non-native plants (including ornamentals) that do not currently pose a risk to ecosystem health			
Euphorbia esula[a] leafy spurge	perennial forb	near Corral Hill (~0.10 acre)	substantial root systems
Cardaria draba[a] hoary cress	perennial forb	horse corral	treated within the horse corral area and along the Highway 4 corridor by EPMT in 2013;
Melilotus officinalis yellow sweet clover	annual/biennial forb (legume)	roadsides	
Juglans nigra black walnut	ornamental; woody	historic orchard and Frijoles Creek	
Coronilla [securigera] varia crown vetch	ornamental; perennial forb (legume)	residential area	
Lathyrus latifolius sweet pea	ornamental; perennial forb (legume)	residential area	
Salvia pratensis meadow clary	ornamental; perennial forb	roadside, Hwy 4	
Mentha arvensis spearmint	ornamental; perennial forb	lower Frijoles Creek	
Althea rosea hollyhock	ornamental; perennial forb	roadside, Hwy 4	

[a] New Mexico (2009) Class A or B noxious weed species (limited or regional distribution)

[b] NPS Exotic Plant Management Team (EPMT)

[c] New Mexico Class C noxious weed species (widespread)

4 Rio Grande Corridor (RGC)

Water • Hydrology;

Geology and Soils • Geomorphology;

Biological Integrity • Communities of Concern

4.09 Rio Grande Corridor and Associated Riparian Vegetation

4.09.1 Description

The Rio Grande forms the eastern boundary of Bandelier NM from the mouth of Frijoles Canyon on the north to a point midway between Alamo and Capulin Canyons to the south, approximately six river miles. This section of the river is bordered to the east by the Caja del Rio, a basaltic upland managed by the Espanola District of the U.S. Forest Service (USFS) primarily for grazing and ORV recreation. The park does not manage the river system proper since the monument's boundary is defined as being the western-side of the active river channel. However, the park does manage its portion of the terrestrial riparian river corridor and participates as a principal stakeholder to influence how river flow and water holding events are managed by the upstream and downstream water managers, specifically the U.S. Army Corps of Engineers (ACE).

About 12 miles upstream of Frijoles Canyon, the Rio Grande enters White Rock Canyon, a deeply cut gorge of relatively recent geologic origin, and the river remains within this canyon system until it emerges downstream at the Cochiti Reservoir Dam some six miles below the park's southern boundary. White Rock Canyon is a scenic river corridor administered by several different federal, state, and local agencies including the Department of Energy's Los Alamos National Lab (LANL, White Rock Canyon Preserve), Espanola District-Santa Fe National Forest (SFNF), Los Alamos County Open Space, Bureau of Indian Affairs San Ildefonso Pueblo, and the Bureau of Land Management.

The river section within White Rock Canyon is generally well watered, although major reductions in spring runoff flow peaks have been imposed by upstream dam regulation, primarily on the Rio Chama, a major tributary which joins the Rio Grande just north of Espanola, NM. And although White Rock Canyon is not managed cooperatively, and several of the entities are not land management agencies per se, the unit is ecologically coherent and the habitat was identified by the New Mexico State Department of Game and Fish as having high potential for supporting reintroduction of extirpated desert bighorn sheep.

<u>Human Impacts to Rio Grande</u>

As a natural community, the riparian ecosystem along the Rio Grande has experienced enormous change during the last century, largely due to water impoundments and flow management, both upstream and downstream of the park, but also as a consequence of domestic grazing and exotic vegetation. Regional water usage has increased dramatically since the 1950s and water rights within the Rio Grande watershed are fully appropriated. Indeed portions of the Rio Grande south of Albuquerque routinely go dry during the summer months as water is variously diverted, consumed, and evaporated. However the major impacts to Bandelier NM have resulted from construction and management of flow and sediment impoundment structures, in particular the Cochiti Dam and Reservoir system.

In addition, within the last couple of years the city of Santa Fe established a water uptake and filtration plant several miles upstream of the park at Buckmans Crossing, where river water is pumped overland for domestic use by the capital city. In the summer of 2013, the town of Los Alamos announced plans to drill several new wells along the west rim of White Rock Canyon near the community of White Rock to extract shallow aquifer river water (technically San Juan Basin-Chama water) to supplement municipal water supplies and replace old deep aquifer well fields. Finally, feral cattle, probably escaped from the pueblo upstream or crossing from the Caja del Rio, have been a perennial problem and contribute to a degraded riparian zone. Periodically the park attempts to trap and/ or shoot feral cattle in collaboration with the NM state livestock board and under legal guidance of the 1995 feral cattle EA (NPS 1995); recent actions include those in 1995 and 2009.

Cochiti Dam and Reservoir

The Cochiti Dam, completed in 1977, created a flood and sediment control reservoir system which encompasses the entire river corridor within Bandelier NM. Operationally, this means that the reservoir is used to hold back spring runoff in excess of certain thresholds, but excess water is supposed to be evacuated within a 3-month period; however this authority was exceeded during the mid-1980s resulting in several extended water holding events at the maximum flood pool level (~5,460ft). These extended water holding events severely impacted the riparian corridor, initially killing all vegetation both native and exotic, depositing many feet of nitrogen-rich silt, extinguishing springs and seeps and their unique floras, and releasing a succession of exotic plant invasions post-flooding.

Prior to this water holding event the river corridor reportedly supported extensive patches of woody exotics, along with encroachment stands of one-seed juniper on river flats. The flooding eradicated these populations, along with the native understory vegetation. It was primarily the exotic species which recovered and expanded strongly in the aftermath of flooding with a host of new invasive species colonizing the nutrient-rich sediments. In the mid-2000s a major invasive plant control effort focused on three woody exotic species (e.g. Salt Cedar, Siberian elm, Russian Olive) allowed native willow and cottonwood a chance to regain ground along the reach that borders the monument.

Since the mid-1980s water holding, the park has been productively engaged with the ACE (responsible for river operations upstream and downstream, including Cochiti Dam) to promote the benefits of mimicking historic flow regimes (e.g., on channel morphology and cottonwood recruitment) by allowing short duration, high volume spring peak runoff, as well as to point out the negative impacts of long-term water holding events (e.g., on native vegetation,particularly recently colonized sediment bars and willow fly catcher nesting habitat).

Sedimentation

In addition to holding water, the Cochiti Dam and Reservoir traps large quantities of sediment within the affected portions of the river system—as the sediment builds upstream, the river becomes braided, creating a complex mosaic of bars and backwaters, with the bars commonly colonized by native willows. This has created some unique (although artificial) and high quality riparian habitat which, if not altered by future river operations, could support a host of sensitive species including southwestern willow fly catcher. A darker-side to the reservoir sediment storage is the buried contaminant plume originating from Los Alamos National Laboratory (LANL) just upstream of the monument park. Recently the Cochiti Pueblo and ACE have begun coring these sediments to assess levels of various contaminants in the sediment. This same environmental program is beginning to collect baseline vegetation data as well.

Recent fire events have contributed large amounts of sediment to a reservoir system already aggrading much faster than originally forecast due to the sediment inputs from degraded woodland systems. Whether the lifespan of the systems can be extended is an open question. Ironically the absence of sediment in flows below Cochiti Dam has promoted scouring and down-cutting of those reaches which degrades the riparian corridor, as well as habitat for the endangered silvery minnow.

4.09.2 Reference conditions

Reference conditions of the river system are not well documented but we can infer a relatively pristine system prior to 1850, when the hydrologic regime characterized by spring snow melt runoff would generate peaks exceeding 10,000cfs in the absence of upstream dams. Summer monsoonal events would have created temporary pulses in flow and sediment-turbidity, variability that is now greatly moderated by the presence of the dam. The riparian corridor would have supported stands of cottonwood at tributary canyon mouths where sediment fans accumulate and willow thickets along the banks, while scattered seeps and springs emerging at the base of steep canyon slopes would have supported unique floras, including hackberry overstory and sensitive flowering species (e.g., rattlesnake orchid and cardinal flower).

4.09.3 Data and methods

Available data are limited, but include hydrologic flow records from Otowi Gauging Station about 12 miles upstream from the park boundary. This information and a historical perspective are provided in the Cochiti Baseline Biological Report (Allen et al. 1993).

Almost no terrestrial vegetation data are available, aside from floristic reports and an early assessment of post-Cochiti Dam sediment bar formation. Time-series remote sensing reconstructions may offer the best option for assessing landscape scale changes. Baseline documentation of the emerging wetlands and deltas within the monument was provided by mapping efforts in 1996 and 1997 (Promislow 1996; Wechsler and Fettig 1997); more recent changes could be reconstructed using time-series orthoimagery and lidar.

4.09.4 Condition and trend

River flow regimes have been significantly altered from historical conditions. A combination of upstream water impoundments, loss of channel capacity, encroachments into the floodplain, and insufficient snowpack in most years constrain the ability to generate high spring runoff peaks, which are considered beneficial to river dynamics and riparian community health.

Sediment continues to accumulate within the reservoir system, and rates of deposition have spiked in recent post-fire years. While the life-span of this reservoir system is limited by sediment storage, passing sediment through the system is problematic, given the presence of contaminants from LANL. The sediment deltas, braided channels, backwaters, and lush riparian vegetation associated with reservoir deposits have created valuable riparian habitat which supports winter migratory birds, including water fowl and raptors.

Frijoles Spring, which was extinguished by sediment deposition for more than a decade has now re-emerged as a spring fed pond and marsh, although much of the supported wetland flora is exotic.

Recently there are fresh signs of trespass cattle in the park (creating trails, droppings, and impacting Frijoles Spring) despite an intact boundary fence. This situation suggests that the animals were able to walk around the fence in the river during low river levels during the winter of 2012-2013.

Although the magnitude of mid-1980s water holding events has not been repeated (due to a combination of dryer-warmer conditions, earlier snow melt runoff, increased water usage, reduced channel capacity, and more responsible reservoir management), and the system is progressing through a successional recovery phase, visual impacts will continue to be evident for many decades, while the impacts on springs and associated riparian floras and the promotion of exotic weed floras is a longer-term legacy.

Human demand for water in the region continues to increase, and with predicted drying and warming climatic trends, the prospects for additional resource impacts loom while ecological management of the river system become ever more constrained.

4.09.5 Level of confidence

Confidence in the changes of water flow is high due to the presence of gauging station records. Confidence in the assessment of the overall ecological integrity of the system is low to moderate given the absence of resource sampling or monitoring. Historical reference conditions are inferred from a number of lines of evidence, but these are mostly indirect and confidence is generally fair to poor.

4.09.6 Data gaps/Research needs/ Management recommendations

Acquisition of additional remote sensing products (e.g., ortho-imagery and LiDAR), and time-series change analysis to determine status and trends for riparian habitat would be helpful.

The park should continue to participate with other agencies and stakeholders (such as Cochiti Pueblo, ACE, LANL, SFNF) in the management of the river corridor to maintain as natural a flow regime as possible.

4.09.7 Sources of expertise

Brian Jacobs

4.09.8 Literature cited

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Water • Hydrology;

Geology and Soils • Geomorphology;

Biological Integrity • Communities of Concern

4.10 Rito de los Frijoles and Capulin Creek and Associated Riparian Vegetation

4.10.1 Description

Maintaining and protecting proper hydrologic function and natural riparian and aquatic communities of park streams is a priority management goal for Bandelier NM (Mott 1999, Weeks 2007). To function holistically, riparian ecosystems—the biological and structural elements of areas adjacent to streams and rivers—require both physical and ecological integrity. Physical characteristics of healthy riparian zones include intact floodplains, good water quality, sufficient flows, and in-stream structural components (e.g. sediment) that provide habitat for aquatic species and streamside communities (Poff et al. 1997, Bendix and Hupp 2000). Biological elements include native vegetation communities adapted to periodic flooding and appropriately diverse communities of aquatic-adapted and reliant species (Karr 1991, Dwire and Kauffman 2003).

Vegetation types occurring along Rito de los Frijoles (perennial) and Capulin Creek (perennial) are classified as Rocky Mountain Subalpine-Montane Riparian Forest and Woodland, with specific types of box elder-alder-cottonwood, white-fir-box-elder mixed, ponderosa pine-broadleaf mixed, ponderosa pine dry wash, and oneseed juniper dry wash (Muldavin et al. 2011). In Bandelier NM these are fairly narrow zones that include dominant species of ponderosa pine, thinleaf alder, box elder, narrowleaf cottonwood, mountain maple, gambel oak, beech, cherry, and New Mexico olive (Muldavin et al. 2011).

Riparian systems contribute to the overall diversity of regional landscapes, often supporting relatively higher numbers of rare or regionally restricted species (Sabo et al. 2005). In arid systems where water is naturally scarce, riparian systems also provide particularly critical connections between upland, riverine, and downstream communities (Wilcox et al. 2003, Osterkamp and Hupp 2010), but may be relatively more vulnerable to disturbance and climate change effects (Friedman and Lee 2002, Perry et al. 2012).

Disturbance and recovery

Several large wildfires and subsequent flood events since 1977, (most recently the massive Las Conchas Fire (LCF) of 2011; fires are summarized in section 2.2.2), have had substantial impacts, at multiple scales, on the biological communities and hydrologic processes in Capulin and Frijoles canyons in Bandelier NM (Mott 1999, Veenhuis 2002, Stumpf and Monroe 2012). Disturbance is a natural process in all ecosystems, including riparian zones (Lake 2000, Shafroth et al. 2002, Stanley et al. 2010), and periodic flooding is required for the persistence of many stream-associated species (Bornette et al. 2008). However, the events occurring in the monument over the last 30 years, while not always directly human-caused, have had increasingly cumulative negative impacts on natural and cultural park resources. Consequently, there has been concern that park ecosystems may have lost resilience and are now moving toward threshold states from which they may not recover (Dwire and Kauffman 2003, Jackson and Sullivan 2009, Bowker and Smith 2014). Moreover, ongoing and anticipated impacts of prolonged drought and climate change may hinder the recovery and successional processes in riparian systems following disturbance more than would otherwise be expected (Perry et al. 2012).

The primary impacts of the LCF on Capulin Creek and Rito de los Frijoles were the following:

1. Large portions of riparian vegetation communities suffered direct mortality from the fire, so much so that in some areas 100% of the riparian canopy was

lost (Stumpf and Monroe 2012). In Capulin Canyon, approximately 60% of the riparian zone was severely burned, and 40% was slightly to moderately burned. Linearly this translates to about 7 km severely burned and 3 km less so. For Frijoles Canyon, approximately 7.6 km of high severity impacts and 11.4 km of low to moderate severity affected the riparian zone (unpublished GIS analysis). Aerial photography shows near complete loss of vegetation along much of both streams.

2. The extreme reduction in live upland vegetation combined with reduced soil integrity substantially increased runoff during post-fire precipitation events. For example the post LCF peak flow in Frijoles Canyon on September 13, 2013 was three times as large as any previously recorded for that stream (Veenhuis 2002, NPS 2011). Flow in southwestern streams is highly variable and riparian ecosystems in the region are adapted to disturbance caused by large flood events. However, extreme flood events following landscape scale fires can be particularly destructive to riparian ecosystems because they typically transport large volumes of sediment and woody debris (Wilcox et al. 2003, Neary et al. 2005). Post-fire floods frequently result in deepening and widening of stream channels, conditions which in turn alter the attributes of the existing floodplain (Mott 1999, Friedman and Lee 2002, Veenhuis 2002, Neary et al. 2005). Flooding and peak flow events, particularly when large debris are transported, not only cause direct mortality to understory plants that may have survived the original fire, but may also bury or remove the existing seed bank (Yount and Niemi 1990).

Riparian ecosystems are resilient, and after fires, riparian plant communities often recover more rapidly than upland communities due to the moist conditions and because fire severity is often lessened in riparian zones (Dwire and Kaufman 2003, Pettit and Naiman 2007, Ellis 2001). However, ruderal species often establish in heavily disturbed sites, such as flood zones, more rapidly than native species (Bornette et al. 2008, Ringold et al. 2008). Most studies conducted in riparian ecosystems have determined that successional responses to large-scale disturbance can take from years to decades, and are dependent on multiple factors of geography, climate, and human activities (Fisher et al. 1982, Lake 2000, Lite et al. 2005). In addition to the direct loss of riparian forest stands due to fire and flooding, the absence of herbaceous and woody plants removes significant food and habitat resources for numerous vertebrate and invertebrate species (Bess et al. 2002, Lecerf et al. 2005).

Landforms and geomorphology

Stream channel morphology is defined by a complex series of interactions between hydrologic regimes and watershed geologic characteristics. Metrics used to describe channel morphology include substrate size, channel cross-sectional form, and planform of the channel and floodplain. These all influence a stream's habitat abundance and diversity, sensitivity and resilience to natural and human-induced disturbances, retention of surface and sub-surface water, sediment transport and retention, nutrient capacity, and connectivity both within the aquatic system and between the stream and adjacent uplands. Large floods such as those occurring in the streams at Bandelier NM, post LCF, periodically restructure stream channel morphology. Prior to the LCF, Rito de los Frijoles had been relatively unimpacted by disturbances for nearly thirty years, and the stream was in a relatively stable condition, supporting a healthy riparian ecosystem and aquatic communities with high abundance and diversity. Large flood events occurred in Capulin Canyon following the 1996 Dome Fire and at the time of the LCF, Capulin Canyon was in a state of transition. It is probable that the LCF and subsequent floods may have exacerbated geomorphic and vegetation changes in Capulin intiated by the Dome Fire (B. Jacobs pers. comm.).

Resource managers at Bandelier NM have observed noteworthy sediment displacement and landform change since the LCF, particularly in canyon settings that are characterized by steep slopes and fractured bedrock. Limited analysis of LiDAR datasets collected over Frijoles Canyon between 2010 and 2013 identified areas of canyon bottom sediment deposition of +/- 20–40m in severely burned portions of the canyon (Jacobs 2014). The unburned lower canyon also exhibited landform change, particularly near the Lower Falls of Frijoles Canyon that has seen major landslide and rockfall episodes since the 1930s. During the August 2011 major post-fire flood event, the toe of the slope below the falls was scoured by the stream and thin vertical bedrock blocks fell into and broke up in the stream channel. The site is poised to experience another major landslide and/or rock slope failure that could very easily obliterate the remaining remnants of the Falls Trail in this area.

Vegetation recovery

Succession processes after floods should facilitate the transition from essentially bare ground and buried vegetation to herbaceous emergence, woody species recovery (often from resprouting, Engelhardt et al. 2011), and culminating again in a riparian woodland. Riparian vegetation can be particularly resilient to long-term fire impacts, though response and recovery trajectories appear to vary depending on fire severity, availability of source populations, and climatic conditions in the immediate post-fire years (Ellis 2001, Dwire and Kaugman 2003). Characteristics that would be relevant in assessing vegetation community structure could include woody species diversity (Jackson and Sullivan 2009), tree cover (canopy area), woody species recruitment rates (either from seed or re-sprouting, Jackson and Sullivan 2008), population dynamics of dominant tree species, herbaceous species diversity, and exotic species abundance.

However, given how much the riparian zone has been altered by direct fire and flood impacts, and the likelihood that drought conditions may continue for the foreseeable future, consideration should be given to the validity of using past community conditions as a reference (Stoddard et al. 2006). For example, species that are more adapted to flooding and periodic inundation may become less common than more drought-tolerant taxa. Also, the potential for exotic species to proliferate in post-disturbance conditions is likely an undesired outcome with which managers should be concerned (Rindgold 2008).

4.10.2 Reference conditions

Landforms and geomorphology

In the context of the last 10,000 years, the recent flooding observed post-Las Conchas Fire is not unprecedented. The last hundred years, however, have been very quiet in terms of geomorphic stability, as a result of vegetation conditions due to fire suppression. For example, Reneau (2000) identified periods of rapid downcutting in Frijoles Canyon during the early Holocene (8000-10000 years ago).

Vegetation

The fairly narrow montane riparian zones along interior streams support well-developed and diverse forest or woodlands, including dominant species of ponderosa pine, thinleaf alder, box elder, narrowleaf cottonwood, mountain maple, Gambel oak, beech, cherry, and New Mexico olive. Periodic fires and floods result in patchy loss of riparian vegetation followed by succession.

4.10.3 Data and methods

Landforms and geomorphology

Staff observations, review of historic photographs, repeat photo points and repeat LiDAR all contribute to understanding of ongoing landform changes.

Measures of stream morphology, including substrate particle size, habitat type, and geomorphic channel unit, are all useful indicators of channel condition and are collected as part of the SCPN's aquatic macroinvertebrate monitoring program. These data have been collected from two sites on both Rito de los Frijoles and Capulin Creek for more than five years and provide a picture of these streams functional condition (Stumpf and Monroe (2009, 2010, 2011, 2012a, 2012b).

Streamflow

At present, there is one active streamflow gauge in Rito de los Frijoles and no gauges on other streams in Bandelier NM. The streamflow gauge on Rito de los Frijoles was destroyed by the flood on August 21, 2011. A new gauge was installed by the USGS at the same location and data from this station are available online at: (http://nwis.waterdata.usgs.gov/nm/nwis/uv/?site_no=08313350). Since the 2011 flood the accuracy of base flow data from this gauge has been poor due to high volumes of sediment moving through the system and numerous moderate to large flood events.

While only one streamflow gauge is currently active in the monument, there are data available from gauges that had been previously deployed in Rito de los Frijoles and Capulin Creek. Table 4-3 provides information about the gauges and the data available.

Vegetation

Very few data exist that quantitatively describe the historic riparian vegetation community structure of Capulin or Frijoles creeks. A small number of plots were established in riparian areas during the vegetation mapping project (Muldavin et al.2011) and there appear to be three within the Capulin riparian zone but none in the Frijoles area. SCPN estimates of riparian vegetation canopy cover at four sites, two on Rito de los Frijoles and two on Capulin Creek, as part of the network's aquatic macroinvertebrate monitoring program. The riparian canopy was dense at sites along both streams prior to the LCF, ranging from a low of 63% at Rito de los Frijoles to 99% at Capulin Creek. In the years since the LCF, no live riparian vegetation has been recorded at 3 of the 4 SCPN monitoring sites. SCPN's site on Rito de los Frijoles near the monument visitor center site did not burn during the LCF, but has seen a reduction in riparian cover most likely due to scouring. Cover at that site declined from an average of 75% prefire to 27% postfire.

Landforms

Staff observations, review of historic photographs, repeat photo points and repeat LiDAR all contribute to understanding of ongoing landform changes.

4.10.4 Condition and trend

Geomorphology and streamflow

Stream geomorphology data collected by SCPN since 2009 provided the following information:

- Substrate particle size. Gravel, silt, and cobbles were abundant channel substrate types prior to the LCF in both Capulin and Frijoles creeks. In the years since the fire and subsequent flooding sand and small gravel have become the dominant substrates.
- Habitat type. Prior to the LCF, aquatic macroinvertebrate habitat data collected at sites on Rito de los Frijoles and Capulin Creek showed a wide diversity of habitat types, including both substrate bottom materials and primary production materials (leaf packs, woody debris, vegetation and root wads). Post fire data show that the majority of those habitat types are no longer present. Since 2011 available habitat in these streams appropriate for aquatic macroinvertebrates ranged from 5% to 35%.
- Geomorphic channel units. Riffles, runs, and cascades were the most abundant geomorphic channel units found along Rito de los Frijoles and Capulin Creek prior to the LCF. The influx of large volumes of fine sediments since the fire has resulted in the burial or loss of gravel and cobble substrates and has greatly reduced the number of riffles and cascades in the streams. Long sandy runs are now the dominant geomorphic channel units present.

The LCF had substantial physical impacts on both Capulin Creek and the Rito de los

Frijoles (Stumpf and Monroe 2012), greatly altering stream morphology, at least for the short-term. Continued erosion and flood events can be expected for the next several years. Numerous flood events have occurred since the fire, the largest of these in July 2013, suggesting watershed conditions remain impaired (S. Monroe pers. comm.). However, assuming that upland conditions improve in coming years, peak flows and associated debris and high-flow impacts will decline. However, it is possible a prolonged reduction in streamflow resulting from climate changed-induced drought will have significant impacts on Capulin and Frijoles riparian systems (Lake 2000).

Vegetation

If the assessment was for present and standing conditions only, the current condition of riparian vegetation communities in Capulin and Frijoles canyons affected by recent fire and flood events is poor. However, what is of ecological importance is how well these sites are poised to recover. Capulin Canyon is in much worse shape following the LCF than Frijoles. This is probably due to the relatively short time since the Dome Fire, and because a greater percent of the Capulin watershed, including upper Capulin, burned at high severity, while upper Frijoles burned less severely.

Under historic regime conditions, fires and floods alone in these systems would likely support future healthy riparian forests following secondary succession (Scott and Reynolds 2007). But the severity of the impacts, along with the potentially stronger influences of prolonged drought and upland and landscape (watershed) responses to climate change (Bendix and Hupp 2000, Engelhardt et al. 2011), alter that scenario substantially. The potential for invasive species to quickly establish and affect successional trajectories is also a concern.

4.10.5 Level of confidence

The level of confidence in this assessment is moderate.

4.10.6 Data gaps/Research needs/ Management recommendations

- Monitoring of riparian zones to document recovery processes and detect introductions of invasive/exotic species.
- Aerial photography could greatly assist in documenting vegetation recovery.
- Monitor the rate of landform change using LiDAR.

Table 4-3. Data available from streamflow gauges on Capulin Creek and Rito de los Frijoles in Bandelier National Monument.

Stream	Station IDs	Location	History	Data available
Capulin Creek	USGS # 083133655	Capulin Canyon at Ranger Cabin	NPS station destroyed by the first post-Dome Fire flood 6/26/96; new gauge installed by USGS near the previous station so are considered one record.	4/85-6/96; 6/97-11/98; 7/97 upgraded to satellite system (DCP)
	USGS # 08313365	Capulin Canyon above Ranger Cabin	Crest-stage gauges to monitor streamflow after the Dome fire	7/96-11/98
Capulin Creek	USGS # 08313366	Capulin Canyon below Ranger Cabin	Crest-stage gauges to monitor streamflow after the Dome fire	7/96-11/98
Capulin Creek	USGS # 08313368	Capulin Canyon below Painted Cave	Crest-stage gauges to monitor streamflow after the Dome fire	7/96-11/98
*Rito de los Frijoles	USGS # 08313350	Rito de los Frijoles near VC	7/63-9/69 and 7/77-9/79: USGS, location upstream of VC. 10/79-present: USGS and LANL, site downstream of VC.	7/63-9/69; 7/77-present
Rito de los Frijoles	USGS # 08313300	Rito de los Frijoles near Los Alamos		1960-1963

*Data available at:
http://nwis.waterdata.usgs.gov/nm/nwis/uv/?site_no=08313350; http://wdr.water.usgs.gov/wy2012/pdfs/08313350.2012.pdf

- Continue operation of a discharge gauge on the Rito de los Frijoles.
- The known flood history within Frijoles Canyon and projected increase in extreme flood events should be fully valued by managers in future infrastructure development efforts.
- Repeat LiDAR over multiple years, with analysis, to assess how long accelerated landform changes continue once episodes of mass movement are underway.

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Brian Jacobs, Stephen Monroe

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Water • Water Quality

Biological Integrity • Communities of Concern

4.11 Water Quality and Aquatic Macroinvertebrates

4.11.1 Description

4.11.1.1 Background

Maintaining and improving surface water quality at Bandelier NM has been an important natural resource goal since the establishment of the park (Mott 1999, Weeks 2007). Under the National Park Service Organic Act (1916, 16 USC1), national park waters should remain unimpaired for future generations, and, in addition, Bandelier NM has identified surface waters as an important natural and cultural resource (Weeks 2007). Riparian vegetation, stream flow, channel morphology, stream water quality, and aquatic macroinvertebrate communities have all been identified as priority vital signs for monitoring in the monument by the Southern Colorado Plateau Network (SCPN) of the NPS (Thomas et al. 2006).

Aquatic macroinvertebrates serve as the primary food base for many aquatic vertebrates and they function as the primary processors of energetic inputs in low order streams, like the streams found in Bandelier NM. Consequently, they are important indicators of ecosystem health. Communities of aquatic macroinvertebrates in streams are commonly monitored alongside physical and chemical properties of water because of their potential to provide indications of water quality and overall hydrologic condition (Brasher et al. 2011).

This assessment will focus on the water quality and macroinvertebrate communities of the Capulin Creek and the Rito de los Frijoles ('Frijoles'), and associated upland catchment areas. These sites each have a substantial data record, and were selected for long-term monitoring by SCPN because of their history of impairment; the substantial impacts that have occurred to each as a result of recent large-scale disturbances; and because Rito de los Frijoles is the stream most accessible to visitors. A separate section of this report will address vegetation and physical properties of the two streams, and an overview of the two canyon systems is provided in Section 4.10.

4.11.1.2 Impacts on aquatic systems at Bandelier National Monument

Historic and continuing impacts to aquatic systems have made maintaining good water quality and protecting stream ecosystems a particular challenge for park managers (Mott 1999). Significant past impairments to surface water quality in the park include the use of organochlorides (e.g. DDT; Mott 1999); high nutrient loads resulting from human and livestock use in the watershed (Thomas et al. 2006); and the introductions of trace and heavy elements into the groundwater system by Department of Defense activities at the adjacent Los Alamos National Laboratory (LANL; Mott 1999, Weeks 2007).

In addition, a series of extreme wildfires that began in the 1970s, and associated post-fire floods have increased sedimentation, altered existing aquatic invertebrate communities, and damaged or destroyed streamside forest vegetation (Veenhuis 2002, Vieira et al. 2004, Stumpf and Monroe 2012b). Fire can have both direct and indirect impacts on riparian communities. Direct effects occur during the fire and consequent major runoff events (Neary et al. 2005), while indirect effects are a consequence of subsequent vegetative and geomorphologic changes (Minshall 2003, Oliver et al. 2012). Aquatic communities in Bandelier NM have repeatedly experienced both types of effects. Interactions between persistent drought conditions and continued introductions of contaminants (from both natural and anthropogenic sources), with repeated disturbance, may further limit the potential for long-term aquatic and riparian ecosystem recovery (Allen 2007, Weeks 2007).

4.11.1.3 Water quality standards

The state of New Mexico has developed statewide water quality standards and criteria for surface waters and specific additional criteria relevant to streams to meet requirements of the Federal Clean Water Act (NMED 2013). A water quality standard refers to a specific parameter and its associated designated use, in conjunction with a criterion, which is the numeric component against which a result is compared. For example, the standard for chronic mercury (the specific parameter) for aquatic life (designated use) is 0.77 µg/L (numeric criterion). For this assessment results of water quality core parameter measurement as well as other frequently sampled analytes are summarized and evaluated in Appendix E. The National Park Service does not have regulatory authority over waters in the U.S., or even the authority to make assessments for designated use. For a complete summary and explanation of water quality standards see Dyer and Monroe (2013).

4.11.1.4 Research and monitoring history

Water quality

Ongoing efforts to understand and protect aquatic ecosystem integrity in Bandelier NM have resulted in a substantial body of work describing data collection and summary efforts for hydrologic resources (Weeks 2007, Brown 2008). Specifically, the NPS Water Resources Division completed a comprehensive summary of existing surface water quality data for the monument (NPS 1997); Brown (2008) summarized and evaluated available water quality data for Bandelier NM and four other park units in the SCPN through 2004; and Macy and Monroe (2006) collected and presented water quality data for multiple SCPN parks, including Bandelier NM, in 2005 and 2006. See these existing data syntheses for detailed descriptions of data collection methods and data sources. For additional background information, the reader is directed to the above-mentioned references as well as Mott (1999), Appendix C in Thomas et al. (2006), Weeks (2007), Brasher et al. (2011), and Monroe et al. (in preparation). Water quality data for streams and springs in the monument and nearby areas are available at www.intellusnm.com.

During the period since 2005 water quality data have been collected at the park by various agencies and contractors focused on issues that include an assessment of surface water quality in relation to state of New Mexico water quality standards, an NPS contracted study updating the status of previously detected high levels of DDT in Rito de los Frijoles, a sampling program designed by EPA and SCPN to establish baseline levels for potential contaminants that include pharmaceuticals, personal care products, waste indicators, and pesticides in Rito de los Frijoles. Brief summaries of each of these efforts are provided below.

In 2010 SCPN began monitoring water quality in two perennial streams in Bandelier NM —the Rito de los Frijoles and Capulin Creek. Reports and water quality data collected at the monument since 2005, are summarized in Table 4-4.

Aquatic macroinvertebrates

Previous studies investigating the aquatic macroinvertebrate community of streams at the monument have focused on relating community structure to water quality, researching biomonitoring techniques (Hopkins 1992, Stevens 1996), or examining the short-term (Pippin and Pippin 1980, Pippin and Pippin 1981, MacRury and Clements 2002) and long-term (Vieria et al. 2004) effects of large wildfires on community structure and recovery. In 2005 SCPN and USGS conducted a pilot study to determine the most appropriate methods for the long-term monitoring of aquatic macroinvertebrates at Bandelier NM (Brasher et al. 2010). SCPN began monitoring aquatic macroinvertebrates in Capulin Creek in 2007 and in the Rito de los Frijoles in 2009.

4.11.2 Reference conditions

The preferred status for water quality is a condition that minimizes the presence of contaminants associated with 20th century human activities.

Prior to the 20th century, abundant and diverse macroinvertebrate communities were likely present in monument streams, with local loss and recolonization following fire or flood events.

4.11.3 Data and methods

SCPN has been monitoring water quality at Bandelier NM since 2010. The objectives are to determine the status and trends of NPS core water quality parameters (water temperature, pH, specific conductance dissolved oxygen, turbidity, and discharge) and selected water quality constituents, including bacteria, nutrients, major elements, and trace metals (Appendix E) in relation to flow, season, and climatic conditions. There are currently five SCPN water quality monitoring sites in the monument, two on Capulin Creek and three on Rito de los Frijoles, each of which is sampled three to four times per year (Dyer and Monroe 2013).

Long-term annual monitoring of aquatic macroinvertebrates at Bandelier NM began in Capulin Creek in 2007 and in Rito de los Frijoles in 2009. SCPN is currently monitoring four sites, two each on Capulin and Frijoles creeks. Two sites on Rito de los Frijoles and one site on Capulin Creek are co-located with a water quality monitoring site. Detailed sampling methods are described in Brasher et al. 2011 and are based on the USGS NAWQA and the EPA EMAP programs. Sampling is conducted once each year in either September or October. Biological and physical habitat data are collected at each of the four sites (Brasher et al. 2011). Sampling includes both qualitative and quantitative methods; qualitative data provide a species list for each site, while quantitative samples result in estimates of abundance that can be spatially and temporally compared (Brasher et al. 2011, Stumpf and Monroe 2012b). The SCPN is currently using a broad suite of metrics to summarize key aquatic macroinvertebrate community characteristics (Brasher et al. 2011). These metrics describe:

- abundance/richness/diversity: total abundance, number of taxa, and Simpson's Diversity Index
- tolerance: percent of individuals and taxa sensitive to perturbation
- functional – feeding: actual and relative abundance of individuals and richness of taxa.
- composition: abundance and relative abundance of Ephemeroptera, Plecoptera and Tricoptera (EPT).
- habitat factors: (stream width, stream depth, discharge, water quality parameters

As of this writing the SCPN aquatic invertebrate monitoring program has collected six years of data for Capulin Creek, and four years of data for the Rito de los Frijoles. Six years is considered a minimum period needed to begin to assess the range of natural variability in community dynamics (Brasher et al. 2011). However, given the substantial disturbance caused by the Las Conchas Fire (LCF) and subsequent hydrologic events, Capulin and Frijoles creeks have experienced instability during the years since the fire. Several more years of sampling will be necessary to determine whether future communities will be similar to those that existed prior to 2011 (e.g. during the pilot study in 2005-2006 when abundance and diversity in Capulin creek appear to have been relatively high), or be markedly different.

4.11.4 Resource condition and trend

4.11.4.1 Water quality standards

Rito de los Frijoles in Bandelier NM is on New Mexico's 2012–2014 303(d) list of impaired waters for aluminum and DDT in fish tissue. Past sources of impairment have also included fecal coliform bacteria, temperature, turbidity, and radium 226 and 228. Some aspects of water quality in Bandelier NM had improved prior to the LCF in 2011; Capulin Creek was listed as impaired due to sedimentation and turbidity from 2002–2006 and is no longer listed. However, recent water quality data collected after the LCF indicate that the status of both streams will likely change due to large-scale inputs of sediment into the streams and increased temperatures caused by the complete loss of

Table 4-4. Reports of water quality data collected from streams at Bandelier National Monument since 2004, and aquatic macroinvertebrate data collected since 2005.

Collection years	Reference	Source
Water Quality		
	Macy and Monroe 2006	SCPN
2010	Dyer, Monroe and Lawrence 2012	SCPN
2011	Dyer and Monroe 2013	SCPN
2012–2013	In prep., data available from SCPN	SCPN
2004–2008	New Mexico Environment Department 2010	NMED
Macroinvertebrates		
2005-2006	Brasher et al. 2010	SCPN
2007	Stumpf and Monroe 2009	SCPN
2008	Stumpf and Monroe 2010	SCPN
2009	Stumpf and Monroe 2011	SCPN
2010	Stumpf and Monroe 2012	SCPN
2011	Stumpf and Monroe 2012	SCPN
2012–2013	In prep., data available from SCPN	SCPN

riparian vegetation.

New Mexico Environment Department Pajarito Plateau Water Quality Assessment

In 2006 the New Mexico Environment Department (NMED) conducted an assessment of surface water quality focused on streams flowing across the Pajarito Plateau, including three sites on Rito de los Frijoles and one in Lummis Canyon in Bandelier NM. The assessment covered the period 2004–2008 and included collection of water quality samples analyzed for trace metals, radionuclides, bacteria, ions, and PCBs. Among these analytes, aluminum was found to be in exceedance of state of New Mexico water quality standards in samples collected at Rito de los Frijoles (New Mexico Environment Department [NMEDNMED 2010).

DDT

During the 1950s and 1960s NPS applied DDT and other chlorinated hydrocarbons for pest control in lower sections of Frijoles Canyon. Chemicals also entered canyon aquatic and terrestrial ecosystems through the NPS drainage system or via leakage from storage containers (NPS 1995). Samples collected from Rito de los Frijoles in 1975 by the State of New Mexico showed high concentrations of DDT and subsequent studies also found evidence of contamination in sediment and fish tissue. A risk assessment and Environmental Analysis resulted in a ban on fishing in the affected reach of Rito de los Frijoles. Resampling for DDT in 2012 found concentrations similar to those identified in the 1990s (Baker 2013).

Bacteria

Water samples collected by NPS and others in the 1970s, 1980s, and 1990s from multiple sites on the Rito de los Frijoles identified high levels of fecal coliform at various times (NPS 1995). Some of the samples had fecal coliform levels in exceedance of established state of New Mexico water quality standards (New Mexico Water Quality Control Commission 1995). Generally coliform levels are higher during summer months when visitor use and water temperatures are both high, and at sites in the more heavily used portion of Frijoles Canyon (Mott 1999). Various researchers have suggested that the source of coliform could be point sources or ground water. Possible sources identified included spills at the lift station; spillovers at the horse corral due to heavy rain; a continuously leaking sewage pipe; leachate from unlined pit toilets at Ceremonial Cave; natural sources, such as the turkey vulture roost in the Frijoles riparian area; or washing of naturally occurring bacteria, and backcountry human and horse waste, into the stream by storm flows (Mott 1999).

In 2009 SCPN initiated regular collection of water samples for *E. coli* analysis at sites on Rito de los Frijoles and Capulin Creek in Bandelier NM. From 2009–2014, summer *E. coli* levels frequently exceeded New Mexico State water quality standards in lower Frijoles, while sites upstream and on Capulin Creek only occasionally experienced exceedances (Figure 4-20).

Water temperature

Water temperature data collected at 15-minute intervals with temperature loggers deployed in Rito de los Frijoles (unpublished data) by SCPN near the monument's visitor center during the period 8/1/2009 –8/13/2013 indicated regular exceedances of the New Mexico water quality criteria for the high-quality coldwater aquatic life designation currently applied to this stream (Figure 4-21). The number of water temperature exceedances per year has increased significantly since the LCF, probably due to greater solar radiation input to the stream caused by loss of vegetation resulting from the fire and subsequent flooding. While similar datasets do not exist for the Upper Crossing of Rito de los Frijoles or Capulin Creek, temperature measurements during periodic site visits indicate that temperature exceedances occur at these sites as well.

Contaminants of emerging concern

Beginning in 2011, SCPN has partnered with the United State Environmental Protection Agency (USEPA) to assess the occurrence of "contaminants of emerging concern" (CEC) in water sources in SCPN parks. Included in this project are approximately 250 analytes, consisting of pesticides, personal care products, pharmaceuticals and waste water indicators. For many of these contaminants, the risks of long-term, low-level exposure to human and ecosystem health are not known and the goal was to develop a baseline dataset documenting the occurrence of these contaminants. At Bandelier NM SCPN collected samples for CEC analysis from two sites in Rito de los Frijoles during 2011 through 2013. Eighteen contaminants were detected in samples collected near the visitor center, and 15 contaminants were detected in samples collected near the Wilderness Boundary. Results between the two sites were similar, and the most common detections were caffeine, flame retardants, and bisphenol A (BPA). Though the sources of these contaminants are unknown, they may be introduced into park surface water through wastewater treatment, recreational use of surface waters, atmospheric deposition, watershed runoff, or management actions, such as invasive plant treatments, park maintenance and landscaping.

Radionuclides

Runoff from areas burned by wildfire mobilizes high concentrations of ash and sediment, which potentially contain contaminants concentrated by the fire. Contaminants may be present naturally in the forest overstory, understory, and litter, or may have been transported by smoke generated by the fire. These contaminants could be a complex mix of particles, liquids and gaseous compounds, including polynuclear aromatic hydrocarbons, organic acids, particulate matter (PM), semi-volatile and volatile organic compounds and the inorganic fraction of particles. They could also be elevated levels of radionuclides (gross alpha emitters) or other radioactive contaminants of concern.

In order to assess radionuclide levels in Bandelier NM streams after the LCF, SCPN collected surface water and streambed sediment samples at two sites on the Rito de los Frijoles, at a site on Alamo Creek, and at a site on Capulin Creek in 2011 and 2012. These samples were analyzed for gross alpha, beta, gamma, isotopic uranium, and radium 226/228.

Of the analytes, only gross alpha and radium have New Mexico State drinking water criteria. None of the results from the four sites exceeded these criteria. Results for analysis

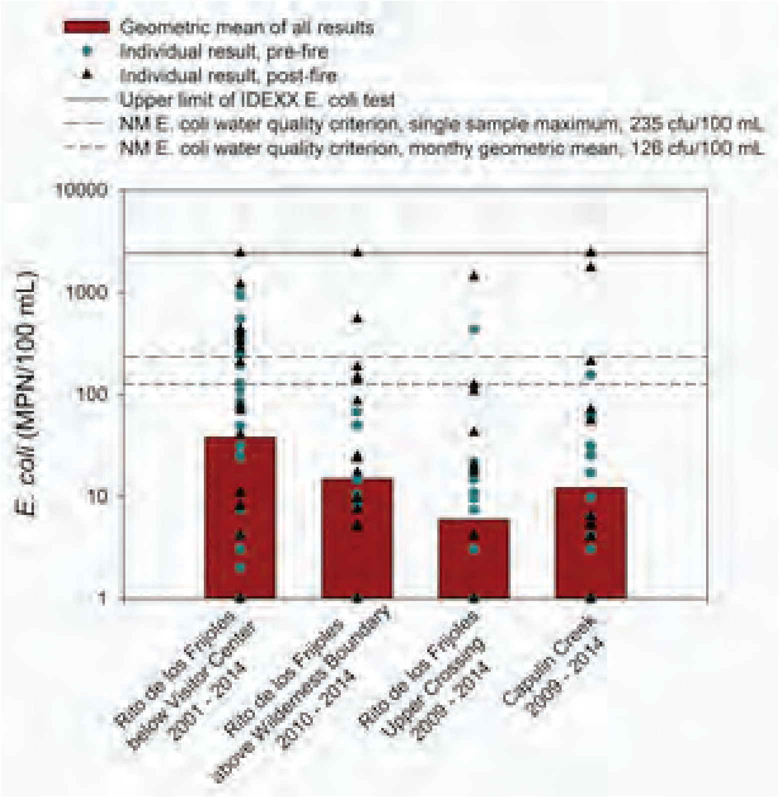

Figure 4-20. *E. coli*. Results of bacteria samples collected from Rito de los Frijoles and Capulin Creek in Bandelier, 2001–2014, showing *E. coli* counts and State of New Mexico water quality criteria.

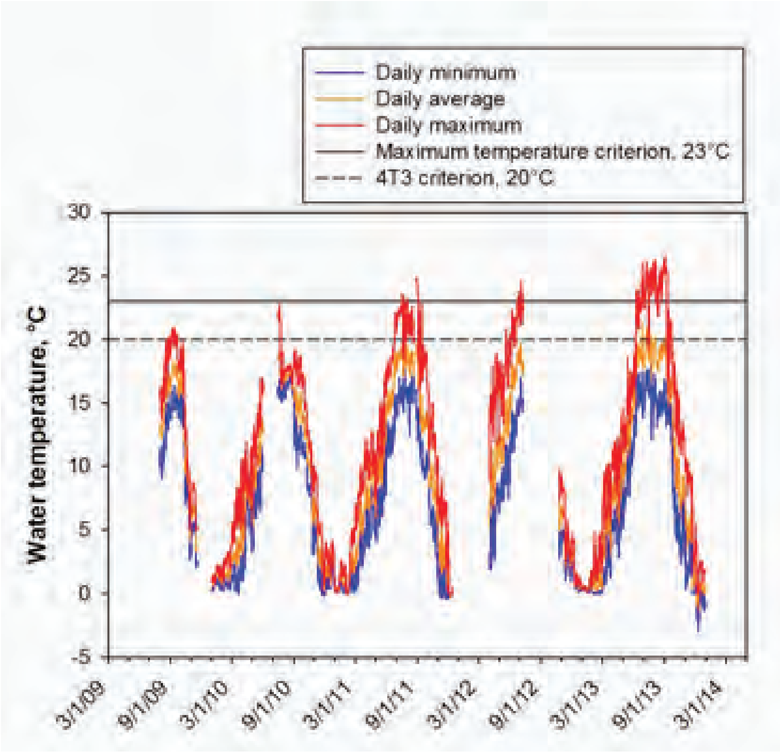

Figure 4-21. Water temperatures in Rito de los Frijoles at Bandelier National Monument from August 1, 2009 to March 1, 2014.

of water samples for gross alpha, beta, gamma, and isotopic uranium results were lower in the 2012 samples than in the samples collected during 2011. Conversely, levels of gross alpha, beta, and isotopic uranium in streambed sediment samples were higher in 2012 than they were in 2011.

4.11.4.2 Aquatic macroinvertebrates

Given the increase in catastrophic wildfire over the last several decades, we recognize that the SCPN monitoring record is not long enough to describe reference conditions for Bandelier NM streams. However, this six-year monitoring effort can be used to describe the condition of aquatic macroinvertebrate communities prior to and after the LCF. SCPN has collected 4 years of prefire data at Capulin Creek and 2 years of prefire data at Frijoles. Post-fire quantitative aquatic macroinvertebrate data exist for 2 years at Frijoles and for 1 year at 1 site on Capulin Creek. Data were not collected at CAP02 in 2011 or either site in 2012 due to a lack of appropriate habitat. The data presented in this section refer to quantitative samples collected at targeted riffle habitats.

Pre-fire condition

Prior to the LCF and subsequent floods, samples collected at SCPN monitoring sites were species rich and high in abundance. Average sample abundances from Rito de los Frijoles in 2009 and 2010 ranged from 688.0 individuals per sample to 760.6. In the 4 years preceding the LCF, 2007–2010, Capulin Creek sample abundance averaged from a low of 229.2 individuals per sample to a high of 757.0. Pre-fire taxa richness at SCPN sites on Frijoles Creek ranged from 27 to 30 taxa per sample. At sites on Capulin Creek, pre-fire taxa richness ranged from 13.4 to 31.6 taxa.

Pre-fire samples collected at sites on both streams possessed rich and diverse aquatic macroinvertebrate communities. At monitoring sites along Frijoles, the Simpson's Diversity Index averaged 0.87. At Capulin Creek pre-fire diversity was slightly lower, with an average Simpson's Diversity index of

0.80. Coleopterans (beetles) were the most abundant orders sampled at the Frijoles prior to the fire, averaging 244.10 individuals per sample. Sensitive taxa in the orders Ephemeroptera (mayflies) (159.25), Plecoptera (stoneflies) (52.85), and Trichoptera (110.30) were also found in high abundances at our sites on the Frijoles. Pre-fire samples collected from the SCPN sites at Capulin Creek were dominated by both Chironomids (midges), averaging 352.46 individuals per sample, ephemeropterans at 290.60, and plecopterans at 92.95 individuals per sample. Coleopterans (91.45), and trichopterans (72.92), while still present, were found in lower abundances at Capulin Creek.

Because of their sensitivity to disturbance, aquatic macroinvertebrate taxa are often monitored as barometers for change or stress in aquatic ecosystems. Prior to the 2011 disturbances, individuals categorized as intolerant to disturbance dominated samples collected at of SCPN's monitoring sites. The average abundance of intolerant individuals collected in samples at Frijoles Creek prior to the fire was 491.23. Moderately tolerant individuals averaged 209.23, followed by tolerant individuals at 1.30 individuals. A similar pattern of dominance was found in samples collected from Capulin Creek. Intolerant individuals dominated samples, averaging 387.98 individuals per sample. Moderately tolerant individuals averaged 156.65 individuals and tolerant abundance averaged 8.55 individuals.

Aquatic macroinvertebrates are grouped into functional groups based on specialized feeding appendages. Prior to the fire, samples collected from monitoring sites along Frijoles Creek were dominated by collector-gatherers, which averaged 540.68 individuals per sample. Scrapers and collector-filterers were also abundant, averaging 214.21 and 148.71 individuals per sample, respectively. Few predators (40.64) or shredders (37.45) were found at sites on Frijoles. Similar to Frijoles, samples from Capulin Creek were dominated by collector-gatherers, which averaged 180.63 individuals per sample. Mean total abundance for shredders (76.55), collector-filterers (72.09), scrapers (65.31), and predators (61.45) were considerably lower.

Post-fire condition

Impacts to aquatic macroinvertebrate communities have been recorded in the park from every fire/flood disturbance event of the previous three decades. Pippin and Pippin (1980) reported a 98% reduction in macroinvertebrates in Rito de los Frijoles following flooding after the La Mesa Fire, a condition Mott (1999) attributed, at least partially, to increased sedimentation. Vieira et al. (2004) found significantly greater change between time periods in the burned (Capulin) than in the unburned (Frijoles) stream after the Dome Fire, and likewise attributed this largely to habitat changes, such as the destruction of pool and riffle habitat (Mott 1999).

The Rito de los Frijoles and Capulin Creek aquatic macroinvertebrate communities appear at present to be in a period of considerable instability following the LCF and the hydrologic events that followed. Samples collected in 2011 and 2012 show that the aquatic communities of both streams have seen massive declines in abundance and shifts in community structure. Along Capulin Creek the loss of appropriate habitat has prevented SCPN from collecting quantitative samples at the lower monitoring site (CAP02) since 2010. Average abundance and richness values at SCPN monitoring sites plummeted to near zero in 2011 and 2012 (Figures 4-22 and 4-23).

While very few individuals of any taxa were found after the LCF, post-fire samples from monitoring sites on both streams show a change in aquatic macroinvertebrate community structure, with a shift in dominance to chironomids (midges) and non-chrionomid dipterans (flies). Post-fire sampling on both streams indicated that moderately tolerant taxa predominated. The dominant post-fire functional group in both streams was collector-gatherers.

Functional groups describe how energy is processed and turned into biomass, based on the structure of the aquatic macroinvertebrate community. Prior to the LCF, sites on both streams had healthy and diverse functional communities capable of processing much of the instream primary production and riparian inputs that flowed through the streams. Scrapers and collector-gatherers were among the most abundant individuals at all sites in the years leading up to the LCF. Scrapers are important because they can process primary production in the form of algae, bacteria, and fungi, while collector gatherers process fine particulate matter found on sediments and detritus. Another ecologically important group found in high abundance along Capulin Creek before the LCF was the shredders. Shredders are responsible for instream processing of live and dead plant materials. Like scrapers, the processing of macrophytes in the water column results in additional resources for both collector gatherer and filterer taxa. Scrapers have been absent from all samples collected at Capulin and Frijoles in the years since the LCF. Shredders have been found at RIT01, but in very low abundances since the fire. The loss of these taxa is ecologically significant because they are responsible for breaking down large pieces of organic matter instream. As a result of their absence, much of the organic matter created or deposited in these streams via riparian vegetation goes unused. An additional consequence of the loss of these groups is less resource availability for the generalist taxa in the collector filterer and gatherer groups. These results suggests that post-fire aquatic communities at SCPN monitoring sites are ill-equipped to process energetic inputs to the stream, a primary function of lower order or headwater streams. Additionally, these results align with those of Vieira et al. (2004), who found that specialized feeders, such as grazers and shredders, that appeared prior to the disturbance were absent in the years immediately following.

One factor that undoubtedly led to the depauperate conditions of the aquatic community at SCPN sites in Bandelier NM was the lack of bottom substrates available for foraging and habitat. Prior to the fire, gravel and cobble size particles dominated at Frijoles Creek (Figure 4-24) and gravels dominated at Capulin (Figure 4-25). Larger substrates provide surfaces that shelter organisms, as well as providing surfaces to cling to while feeding. After the repeated hydrologic events that have occurred along both streams, SCPN data show a shift from larger grained particles to finer grained particles, particularly sand. The overwhelming dominance of sand may be restricting the aquatic community to those organisms adapted to burrowing into softer surfaces. The loss of riparian vegetation in the moderately and severely burned sections of the streams also influences the stream conditions. Riparian vegetation contributes allochthonous inputs to the stream's trophic structure and also shades the streams, thus reducing water temperature.

A third factor influencing both substrate availability and riparian vegetation recovery is hydrologic condition. Previous studies of the aquatic macroinvertebrate communities at Bandelier NM following the La Mesa and Dome fires showed the vulnerability these populations have to large-scale hydrologic events that commonly occur as a result of high severity wildfires (Pippin and Pippin 1980, Vieira et al. 2004). However these studies also show how quickly richness, abundance, and density numbers can rebound in the months following disturbance. Vieira et al. (2004) found density values comparable to their pre-fire collections within in a year of the Dome Fire. These numbers were attributed to taxa whose adult stages have high rates of dispersal. Data collected from SCPN monitoring sites have not yet yielded signs of such a recovery following the LCF. Dipterans, which primarily accounted for the recovery in the Vieira study, have been found in extremely low numbers in samples collected since the LCF.

Differences in post-fire aquatic macroinvertebrate community recovery may be the result of greater hydrologic instability following the LCF in comparison to earlier fires. Large flow events carrying debris flows and depositing fine sediments continued to occur in 2013. These events can impede the recolonization of riparian vegetation and bury or move medium gravel and cobble substrates that are considered vital habitat (Cannon and Reneau 2000, Brasher et al. 2011). This lack of habitat and riparian vegetation undoubtedly affects community structure and diversity. Previous studies suggest that frequency and magnitude of hydrologic events may not diminish until vegetation recovery begins to occur in the watersheds (Veenhuis 2002).

The data collected by SCPN in the years following the LCF suggest that instream conditions are currently unfavorable for most aquatic macroinvertebrate taxa. Vieira et al. (2004) point out that studies from other watersheds around the country experiencing similar disturbance patterns indicate that pre-fire community structure may not be regained for 10 years or more following the fire. Given the scale and severity of the LCF, and the post-fire monitoring data collected thus far, it seems reasonable that recovery of the aquatic macroinvertebrate communities at Bandelier may take 10 years or more. The rate of recovery of the aquatic community will likely be dependent on how quickly hydrologic conditions stabilize and large flow events subside.

Condition and trend

The time to recovery (resilience) or to a new community state will depend on many factors, including future hydrologic conditions and the level of habitat degradation (Neary et al. 2005), and the availability of source populations (Fisher et al. 1982, Hall and Lombardozzi 2008, Vieira et al. 2011). For example, several short-term studies found rapid recovery of both abundance and diversity of riparian invertebrates, attributing this response in part to an influx of nutrients resulting from burned and downed material transported to stream habitats (Malinson and Baxter 2010). However, multiple studies also found that riparian systems within

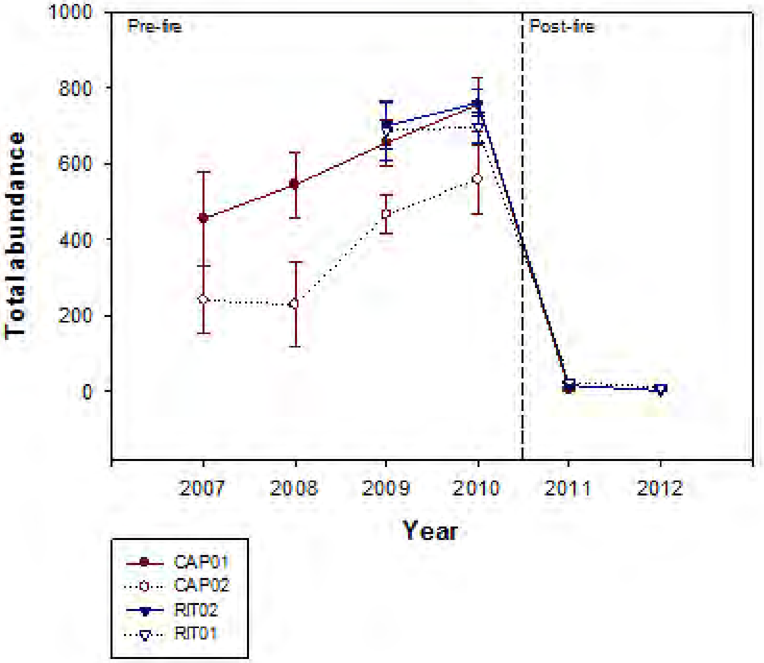

Figure 4-22. Site abundance. Average quantitative sample abundance of individuals collected from SCPN sites in Bandelier National Monument. No quantitative samples were collected from CAP01 in 2011, or CAP01 and CAP02 in 2012 due to a lack of appropriate habitat. Error bars represent +/- 1SE.

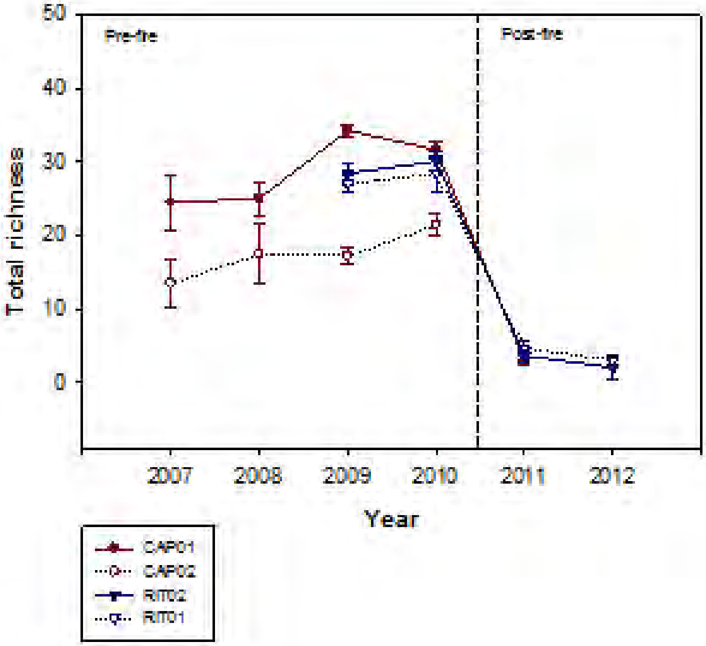

Figure 4-23. Taxa richness. Taxa richness of quantitative samples collected from SCPN sites in Bandelier National Monument. No qualitative samples were collected from CAP01 in 2011, or CAP01 and CAP02 in 2012 due to a lack of appropriate habitat. Error bars represent +/- 1 SE.

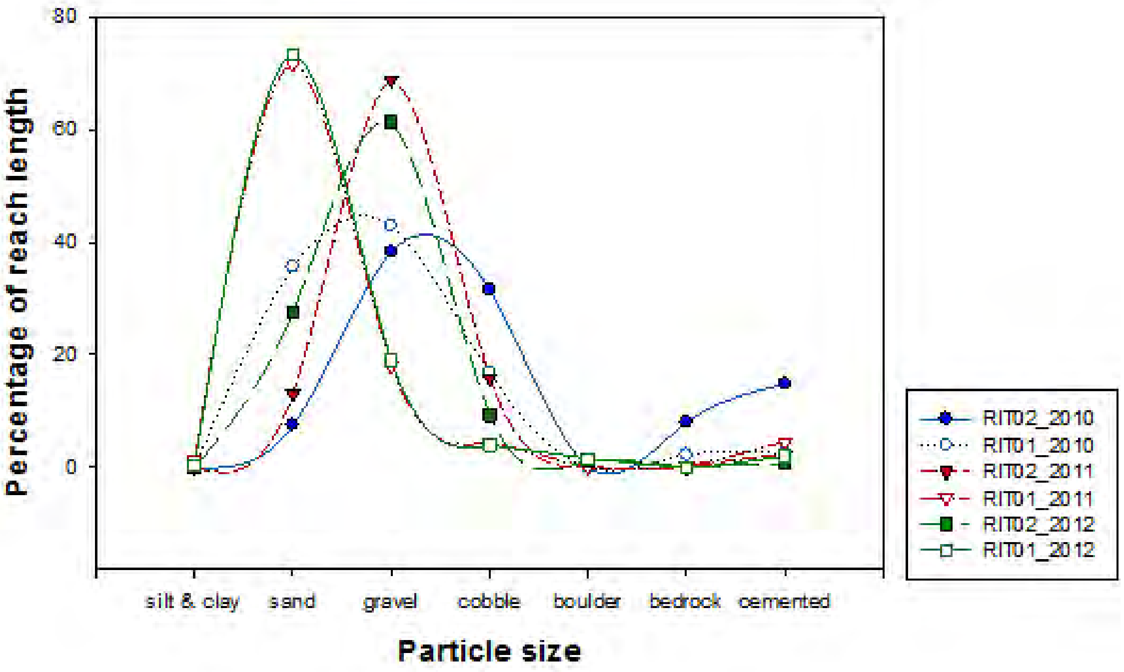

Figure 4-24. Physical habitat. Pre-Las Conchas Fire (2010) and Post- Las Conchas Fire (2011–2012) percentage of particle grain sizes found at RIT01 and RIT02 at the Rito de los Frijoles in Bandelier National Monument.

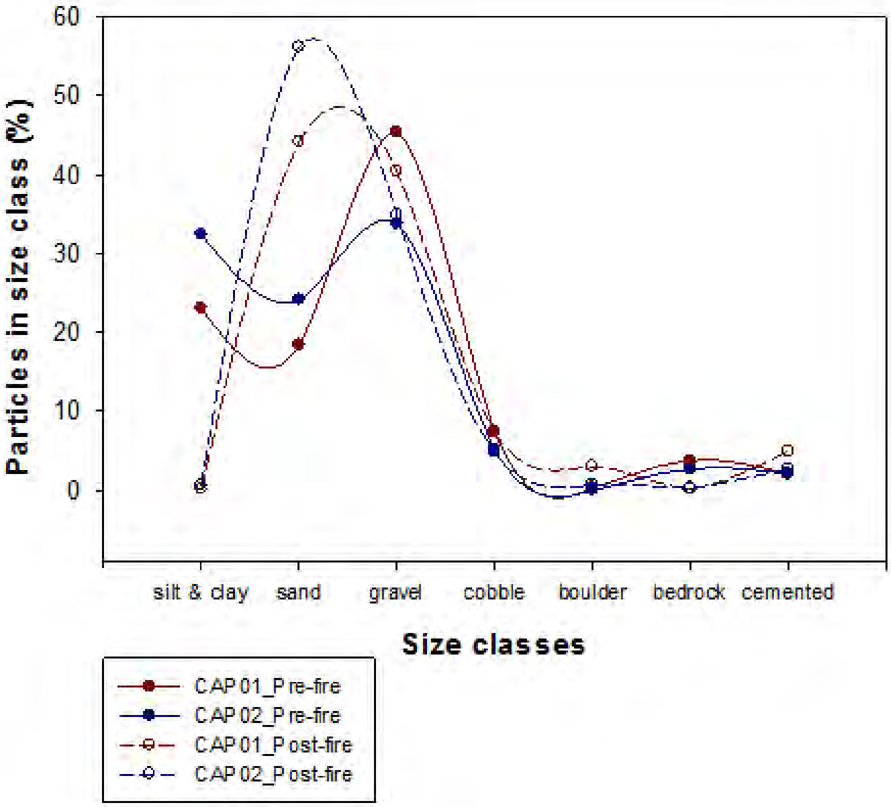

Figure 4-25. Physical habitat. Pre-Las Conchas Fire (2010) and Post- Las Conchas Fire (2011–2012) percentage of particle grain sizes found at CAP01 (red) and CAP02 (blue) at the Capulin Creek in Bandelier National Monument.

catchments that experienced high fire severity exhibited delayed or impaired ecosystem response (Minshall 2003). For example, the absolute abundance of invertebrates may increase long-term following disturbance (Malinson and Baxter 2010), though taxon diversity overall decreases and some functional groups are lost (Neary et al. 2005).

Many studies have found that generalists and tolerant species are relatively more common the first few years post-disturbance (Hall and Lombardozzi 2008, Oliver et al. 2012). Numerous studies have found that fine sediments, as those now predominant in both streams, provide poor quality habitat compared to pebble and cobble sediments, and that the pace at which habitat improves will be reflected in more rapid increases in both abundance and diversity (Vieira et al. 2004, Henne and Buckley 2005, Oliver et al. 2012).

Because of the ephemeral nature of stream invertebrates, (e.g. entire communities can be lost from a site in a matter of hours during catastrophic flooding (Neary et al. 2005), current conditions may have little or no relationship with past community structure and composition (Fisher et al. 1982). Succession as a valid concept for streams has been debated (Fisher et al. 1982), but clearly the recent fires—La Mesa (1978), Dome (1996), and Las Conchas (2011)—and subsequent floods at Bandelier NM have changed the macroinvertebrate communities in both streams substantially, at least in the near-term if not permanently.

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4.12 Native Fish

4.12.1 Description

Fish habitats occur in montane streams at Bandelier NM. While park managers don't manage the Rio Grande, they are interested in habitat status of the river as it informs management considerations within the monument, (see section 4.09, 4.10 and 4.11) for related background information).

The New Mexico portion of the Rio Grande is thought to originally have hosted between 16 and 27 native species (Hatch 1985, Smith and Miller 1986, and Propst et al. 1987), three of which were endemic to the upper Rio Grande as far north as the Chama River—Rio Grande shiner (*Notropis jemezanus*), bluntnose shiner (*Notropis simus simus*), and Rio Grande silvery minnow (*Hybognathus amarus*). Noteworthy is the fact that the type locality for the Rio Grande shiner and the Rio Grande bluntnose shiner is the Rio Grande at Otowi Bridge (currently where New Mexico Highway 502 crosses the Rio Grande in Santa Fe County). Both species have been extirpated from the Rio Grande (Platania 1991). Today, there are no endemic fish species found in the Rio Grande in the reach along the Bandelier NM (Platania 1992). The silvery minnow is the only endemic fish species surviving in New Mexico, and is restricted to below Albuquerque and above Elephant Butte Reservoir (Bestgen and Platania 1991).

4.12.2 History of fish planting in Bandelier National Monument streams (Allen 1989)

New Mexico Department of Game and Fish records on file at Bandelier NM show that 36,750 brook trout (*Salvelinus fontinalis*), 82,740 rainbow trout (*Salmo gairdneri*), and 368,404 cutthroat trout (*Oncorhynchus clarkii virginalis*) of Yellowstone origin were planted in Frijoles Creek between 1912 and 1955. Alamo Creek received 13,000 brook trout, 4,000 rainbow trout, and 6,000 cutthroat trout between 1919 and 1931, while Capulin Creek received 10,500 brook trout, 17,000 rainbow trout, and 1,500 cutthroat trout between 1922 and 1931. Undocumented introductions of brown trout (*Salmo trutta*) have also occurred in these monument streams.

More recently, based on fish collecting (Platania 1992) at Bandelier NM on August 23, 1990 and August 13, 1991, brook, rainbow, and brown trout were found in Rito de los Frijoles.

In the spring of 1996, the Dome fire burnt 16,516 acres of Bandelier NM and the adjacent Dome Wilderness of the Santa Fe National Forest (Veenhuis 2002). Much of the fire was stand-replacing crown fire with debris flows which removed all fish from Capulin Creek. This was unprecedented in the watershed in the previous 330 years (Cannon and Reneau 2000),

On March 30, 2006, after watching the creek and macroinvertebrate populations slowly recover over ten years, a crew moved 100 Rio Grande cutthroat trout from Cañones Creek in the northern part of the Jemez Mountains on the Coyote Ranger District of the Santa Fe National Forest to Capulin Creek in Bandelier NM and the Santa Fe National Forest (Ferrell et al. 2006). Cutthroat trout from these waters had been tested for genetic purity and overall health and were determined to be the best donor population (personal communications between Stephen Fettig and Kirk Patten, 2006).

By 2009, Rio Grande cutthroat trout had reached a self-sustaining population, exceeding the population viability criteria of >500 adult fish or a total population of 2,500 individuals (Patten and Cook 2009). In 2011, post-Las Conchas Fire flooding in Capulin removed all fish from the creek (Stumpf and Monroe 2012).

Rio Grande cutthroat trout are one of 14 recognized subspecies of cutthroat trout native to western North America. Cutthroat trout are distinguished by the red-orange slashes in the gular folds below the jaw. Cutthroat trout once inhabited most cold-water streams throughout western North America. The Rio Grande cutthroat trout is considered the southernmost subspecies of cutthroat trout, and is differentiated from other cutthroat trout by the large spots that are concentrated towards the tail and colorful pink or orange hues on its belly and sides.

Most subspecies of cutthroat trout have suffered large declines within their native ranges. These declines have occurred primarily since the early 1900s due to exotic species introduction, habitat degradation, and over-harvesting (Duff 1996). Cutthroat trout thrive in clear mountain streams that provide clean spawning gravel, feeding and resting sites, and food in the form of aquatic and terrestrial invertebrates (Sublette et al. 1990). Ideal habitat conditions have been altered in many locations by human activities, including grazing, mining, logging, road building, and agriculture. Since the late 19th century, stocking of nonnative trout has been a common practice throughout the western states. Brook trout and brown trout out-compete the native cutthroats for prime habitat areas (Griffith 1988). Behnke (1992) describes a population of greenback cutthroat trout that was virtually replaced by brook trout in five years.

Rio Grande cutthroat trout populations currently occupy less than 10% of their original range (Stumpff and Cooper 1996). However, in October 2014, the U.S. Fish and Wildlife Service issued a 12-Month Finding on a Petition to List Rio Grande Cutthroat Trout as an Endangered or Threatened Species as Not Warranted (http://ecos.fws.gov/speciesProfile/profile/speciesProfile.action?spcode=E05D).

4.12.2 Reference conditions

It is possible, but not certain, that native cutthroat trout existed prehistorically in Frijoles Creek (Allen 1989). Historical information on native fish occurring within creeks internal to the park is lacking. Henderson and Harrington (1914) reported that fish were present in Rito de los Frijoles before 1900, and that fish were gone by 1914. Henderson and Harrington (1914) also referenced flooding and drying, as well as large volumes of debris coming into the creek, as possible reasons for fish vanishing from Rito de los Frijoles between 1900 and 1914.

4.12.3 Data and methods

Sampling was by electrofishing (Smith-Root Type VII, 12-volt backpack unit) and by small mesh seines and preserving (using 10% formalin) in the field for identification and analysis in the laboratory (Platania 1992). Specimens were subsequently transferred to 70% ethyl alcohol and catalogued into the Museum of Southwestern Biology's research collection of fish.

4.12.4 Condition and trend

Internal streams: condition good (non-native fish naturally removed), trend improving (habitat improving). As of the summer of 2014, the stream habitats had not yet stabilized to the point where there was sufficient macroinvertebrate presence to support native fish (See section 4.11 for more information).

4.12.5 Level of confidence

Knowledge of the historical condition of internal creeks is low; knowledge of current condition is high; understanding of whether Rio Grande cutthroat populations could survive long-term in monument creeks is moderately low.

4.12.6 Data gaps/Research needs/ Management recommendations

Creeks internal to the park lack historical baseline data.

Native fish of New Mexico montane streams are at high risk due to expected climate warming and drying in northern New Mexico, It is thought that the near perennial flowing waters of Rito de los Frijoles and Capulin Creek may be an important asset that can aid in regional conservation efforts. However, given the recent loss of large portions of the riparian overstory vegetation in Frijoles, Alamo, and Capulin Canyons, and projected climate warming, it is unclear whether any streams will be cold enough to support Rio Grande cutthroat populations into the future.

Viewing fish management in a regional context is especially important given the likelihood that large fires in the future will result in undesirable extirpations of native fish. Having well distributed populations is the best way of insuring the long-term survival of native fish. Bandelier NM can play an important role in these conservations efforts with the following steps:

1. Develop a fisheries management plan to support regional efforts to manage metapopulations of native fish species of conservation concern across land management agencies, with the goal of establishing multiple native fish species within the internal creeks of the park.
2. In support of this recommendation, park management should (a) work with SCPN to determine when there is sufficient food for native fish species, as network staff continue to monitor the recovery of aquatic macroinvertebrate populations, (b) work with the New Mexico Department of Game and Fish to conduct additional electrofishing surveys, prior to any potential transplants of native fish species, to establish a confidence level of essentially 100% that non-native fish will not interbreed with the transplanted native fish, (c) strictly follow NPS policies on fishing because cutthroat trout are 20 times easier to catch than brown trout (Behnke 1992), making them more susceptible to over harvesting, and (d) assess reintroduction of other non-trout native fish species (e.g., long-nose dace, Rio Grande chub, Rio Grande sucker dace), which may be more resilient to projected climate warming.

4.12.7 Sources of expertise

Stephen M. Fettig

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4.13 Landbirds

4.13.1 Description

Between 1967 and 2007, the average population of common birds across the United States declined steeply, falling 70 percent, from 17.6 million to 5.35 million individuals (Butcher and Niven 2007). These alarming findings document landbird declines which are persistent and based on 40 years of data across 550 bird species (Butcher and Niven 2007a). The analysis of DeSante and Kaschube (2009) further suggests that the observed declines are not something of the past, but an immediate and on-going cause of concern.

Furthermore, DeSante and Kaschube (2009) suggest that the declines are geographically extensive, based on data from 653 monitoring stations across North America. With 192 landbird species contributing to the observed average decrease of 1.77% in adult birds per year, this decline translates to a loss of more than 23% of the adult birds over the 15-year study. With overall landbird populations declining by nearly 2% per year, the prospect of losing another 23% of our birds in 15 years means the situation is critical. With more than 25% of the bird species for which we have reliable data showing significant and worrisome population trends, it is likely that further and irreversible declines of landbirds will occur within many national parks. This is especially worrisome because national parks and other public lands play a disproportionately large role in providing habitat for North American birds.

Partners in Flight (PIF) was founded in 1990 with broad public support as a result of widespread and alarming declines in North American birds (National Fish and Wildlife Foundation 1990). Birds stand out above other wildlife groups as perhaps the most highly valued and actively appreciated component of North America's biological diversity (Rich et al. 2004). In 2001 in the U.S. alone, 46 million birders spent $32 billion to observe, photograph or feed wildlife. The overall economic output of this activity was $85 billion (La Rouche 2003). More importantly, through consumption of pest insects, pollination of plants, dispersal of native seeds, and other services, birds contribute to the maintenance of ecosystems that also support human life.

In 2004, PIF published the North American Landbird Conservation Plan (Rich et al. 2004) that provided a continental synthesis of priorities and objectives to guide landbird conservation actions for 448 native landbirds that breed in the U.S. and Canada. The plan identified the priority species, research, monitoring, and management within Bird Conservation Region (BCR) 16, which includes Bandelier NM. The list of the priority species found at the park are listed in Table 4-5).

In 2009, the first State of the Birds report made it clear that birds are bellwethers of our natural and cultural health as a nation—they are indicators of the integrity of the environments that provide us with clean air and water, fertile soils, abundant wildlife, and the natural resources on which our economic development depends (U.S. Committee of the North American Bird Conservation Initiative [NABCI] 2009). At the same time the report made clear that in western forests, 38 obligate forest species collectively are showing a declining trend (NABCI 2009).

In 2010, the second State of the Birds report focused on climate change impacts. Climate change is producing new threats to birds and vegetation, which may be particularly severe in arctic and alpine regions. Birds are a vital element of every terrestrial habitat in North America. Conserving habitat for birds will therefore contribute to meeting the needs of other wildlife species and entire ecosystems (NABCI 2010).

The 2014 State of the Birds report identified western forest birds that need conservation efforts, as well as common birds in steep

Table 4-5. Status in Bandelier National Monument of bird species identified in the North American Landbird Conservation Plan (Rich et al. 2004) for Bird Conservation Region 16 and species that are found in the park.

Common name	Status at Bandelier National Monument
Dusky grouse	regular breeding species
Flammulated owl	breeding numbers have likely decreased greatly since 1980s
Spotted owl	likely no longer breeds at the park
Black swift	accidental
White-throated swift	characteristic species of the park's dry canyons
Calliope hummingbird	south-bound migrant
Rufous hummingbird	south-bound migrant
Lewis' woodpecker	mostly a very rare visitor to the park, usually during migration
Williamson's sapsucker	regular breeding species; likely to decline with the loss of snags and cavities for nesting
Red-naped sapsucker	regular breeding species; likely to decline with the loss of snags and cavities for nesting
Olive-sided flycatcher	rare breeding species
Willow flycatcher	very rare visitor to the park, usually during migration
Gray flycatcher	regular breeding species
Dusky flycatcher	regular breeding species
Pinyon jay	rare breeding species, may not be present every year, nomadic nature makes monitoring difficult
Clark's nutcracker	rare breeding species
Mountain bluebird	rare breeding species
Sage thrasher	accidental
Virginia's warbler	regular breeding species
Grace's warbler	regular breeding species
Green-tailed towhee	regular breeding species
Brewer's sparrow	migrant species
Cassin's finch	rare breeding species

Rich, T. D., C. J. Beardmore, H. Berlanga, P. J. Blancher, M. S. W. Bradstreet, G. S. Butcher, D. W. Demarest, E. H. Dunn, W. C. Hunter, E. E. Iñigo-Elias, J. A. Kennedy, A. M. Martell, A. O. Panjabi, D. N. Pashley, K. V. Rosenberg, C. M. Rustay, J. S. Wendt, T. C. Will. 2004. Partners in Flight North American Landbird Conservation Plan. Cornell Laboratory of Ornithology. Ithaca, NY.

decline (NABCI 2014). The report identifies Yellow-Watch-List species which are range restricted (small range and population), or are more widespread, but with troubling declines and high threats. For Bandelier NM these species include (see Table 4-9 for scientific names): band-tailed pigeon, flammulated owl, Mexican whip-poor-will, rufous hummingbird, Lewis's woodpecker, olive-sided flycatcher, pinyon jay, Virginia's warbler, Cassin's finch, and evening grosbeak. On the list of common birds in steep decline that regularly occur at Bandelier NM are common nighthawk, Wilson's warbler, and pine siskin.

4.13.1.1 Bird monitoring at Bandelier National Monument

The Southern Colorado Plateau Inventory and Monitoring Network (SCPN) completed bird community monitoring within piñon-juniper habitats in 2008, and within mixed conifer habitats in 2008, 2009, and 2012 (Holmes and Johnson 2012a, 2012b, 2014). The approach used variable circular plot (VCP) counts each sampling year, with 100 plots originally established in the mixed conifer forest. (See Appendix F for a discussion of the point count method for collecting avian data). In 2012, after the Las Conchas Fire, which burned approximately half of the mixed conifer bird sampling plots at high or moderate severity, the remaining habitat was assessed and sampling was conducted only in monitoring plots with no evidence of fire (unburned), or with low canopy burn severity—plots with little to no changes in vegetation structure.

A preliminary comparison of relative abundance between pre-fire (100) and post-fire unburned and low-severity burn (48) plots found that of the 9 most commonly detected species that increased after the fire, 7 were either long distance migrants (chipping sparrow, Hammond's flycatcher, warling vireo, western tanager, and western wood peewee) or species that move to lower elevations in the winter (hermit thrush and Williamson's sapsucker). This increase may have been due to the movement of individuals, upon arrival from their wintering grounds, from unsuitable burned territories to already occupied post-Las Conchas forest areas where foliage for nesting or tree sap was available. Just two of the species that increased postfire were resident species—the hairy woodpecker and pine siskin (Holmes and Johnson 2014).

Of the 9 most commonly detected species that decreased in mixed conifer habitat post-Las Conchas Fire, 6 were resident species (brown creeper, dark-eyed junco, mountain chickadee, northern flicker) or species that move to lower elevations and different habitats in winter (ruby crowned kinglet, yellow-rumped warbler). Three migrant species also decreased in abundance (Cordilleran flycatcher, house wren, violet-green swallow, Holmes and Johnson 2014). It is important to note, however, that these data are from only one year postfire and are likely not indicative of long-term patterns.

Within piñon-juniper habitats, bird banding data specific to Bandelier NM are consistent with broader trends pointing to large-scale declining bird populations. The data are from a juniper-woodland mesa-top sites south of the park's headquarters with methods following the national Monitoring Avian Productivity and Survivorship (MAPS) protocol (DeSante et al. 2007). The site was not treated in the restoration work of the first decade of the 2000s. The data show a decline in both the number and subspecies captured over six years. The decline is robust for scaling, based on effort measured in birds per net-hour (Table 4-6).

In 2010, the park began operating three MAPS stations following continent-wide protocols (DeSante et al. 2007). Data for 2010 through 2013 show low adult return rates (data not presented) and low reproduction rates (Table 4-7) (Fettig unpublished data).

4.13.1.2 Presidential directive to protect migratory birds

Executive Order 13186 of January 10, 2001, identified the responsibilities of federal agencies to protect migratory birds and required the NPS to sign a Memorandum of Understanding (MOU) with the U.S. Fish and Wildlife Service (USFWS) for the protection of migratory birds. Such an MOU was signed in 2010 and committed the NPS to evaluating migratory birds species identified in one or more of the following sources: (1) the USFWS periodic report Birds of Conservation Concern (www.fws.gov/migratorybirds); (2) federally-listed threatened and endangered bird species; (3) priority bird species documented in conservation plans completed under NABCI (Partners in Flight North American Landbird Conservation Plan, U.S. Shorebird Conservation Plan, North American Waterbird Conservation Plan, and North American Waterfowl Management Plan); and (4) listed as priority species or species of highest conservation concern in State Wildlife Action Plans. For Bandelier NM these references include Butcher et al. (2007), NMDGF (2006), NMPIF (2007), Norris et al. 2005), Rich et al. (2004), and USFWS (2008).

Table 4-8 lists bird species of concern which are regular breeding species with moderate or high abundances at Bandelier NM. In contrast, Table 4-9 lists bird species of concern which are accidental, rare, or migrants at the park. Because of the species abundances, investigations and management

Table 4-6. Bird capture rates by year at a Juniper-woodland mesa-top site at Bandelier National Monument (S. Fettig unpublished data). Note that the rate has been declining each year.

	2008	2009	2010	2011	2012	2013
Total birds banded	80	162	55	60	40	37
Total banding mornings	8	11	10	11	11	10
Total net hours (9–12 nets)	324.64	477.77	369.19	417.20	575.87	525.12
Birds banded per net hour	0.24	0.34	0.15	0.14	0.07	0.07
Total subspecies banded	20	24	19	16	18	15

Table 4-7. Number of hatch-year (HY) birds banded per 600 net-hours, for three sites (Alamo Drainage (ALDR), Frijoles East (FREA), and Frijoles West (FRWE)) at Bandelier National Monument from 2010 through 2013. Percent hatch-year birds is equal to hatch year birds banded divided by total birds banded.

Site/Year	2010	2011	2012	2013
Hatch-year birds banded				
ALDR	33	5	50	18
FREA	4	0	10	3
FRWE	24	0	58	9
Totals	**61**	**5**	**118**	**30**
Hatch-year birds/600 net hours				
ALDR	38.60	8.61	64.66	23.39
FREA	4.99	0.00	12.77	4.03
FRWE	27.68	0.00	76.18	11.36
Mean	**24.17**	**2.89**	**50.90**	**13.01**
All age birds banded				
ALDR	171	45	89	72
FREA	58	29	31	32
FRWE	132	48	129	249
Totals	**361**	**122**	**249**	**181**
Percent hatch-year birds				
ALDR	19.30	11.11	56.18	25.00
FREA	6.90	0.00	32.26	9.38
FRWE	18.18	0.00	44.96	11.69
Mean	**16.90**	**4.10**	**47.39**	**16.57**

actions directed at species in Table 4-9 are less likely to contribute to landscape-level conservation efforts than actions directed at species in Table 4-8.

4.13.2 Reference conditions

Populations of selected landbirds in prominent monument habitats maintain viable population levels.

4.13.3 Data and methods

Johnson and Wauer (1996) documented changes in bird communities using a line-transect method that mapped bird observations along one-mile routes in the footprint of the 1977 La Mesa Fire. Additional data for five of the transects extend into the 1990s (Wauer unpublished). Point count data with variable circular plot (VCP) distance sam-

Table 4-8. Birds identified as species of concern by existing published conservation plans, USFWS (2008), Rich et al. (2004), Butcher et al. (2007), Norris et al. 2005), NMDGF (2006), NMPIF (2007), that are regular breeding species with moderate or high abundances at Bandelier National Monument.

Standard English name	Scientific name	Habitats
White-throated swift	*(Aeronautes saxatalis)*	Canyon and cliffs
Broad-tailed hummingbird	*(Selasphorus platycercus)*	Higher elevation shrublands
Williamson's sapsucker	*(Sphyrapicus thyroideus)*	Higher elevation forest with snags
Hammond's flycatcher	*(Empidonax hammondii)*	Closed canopy forests
Dusky flycatcher	*(Empidonax oberholseri)*	Open canopy forests and shrublands
Plumbeous vireo	*(Vireo plumbeus)*	Forests across the park
Western scrub-jay	*(Aphelocoma californica)*	Open canopy forests and shrublands
Juniper titmouse	*(Baeolophus ridgwayi)*	Juniper woodlands
Western bluebird	*(Sialia mexicana)*	Open canopy forests and shrublands
Virginia's warbler	*(Oreothlypis virginiae)*	Shrublands
Grace's warbler	*(Setophaga graciae)*	Ponderosa forests

pling for some piñon-juniper habitats are available from 2008 (Holmes and Johnson 2012a). Additional point count data with VCP are available for mixed conifer forest for 2008, 2009, and 2012 with limited comparison of post Las Conchas Fire changes (Holmes and Johnson 2012a, 2012b; 2014). Park-wide data for landbirds are from Fettig (2004), who utilized Breeding Bird Atlas (BBA) sampling methods (Laughlin 1981, Kingery 1998, Fettig et al. 2002). Briefly, BBA observations are conducted within a 5 km x 5 km area, called a block. Observers develop a comprehensive species list by spending approximately 20-40 hours actively surveying a block during 5–10 visits throughout the breeding season. For each observation of a potential breeding bird, the species, breeding behavior, date, and cover type are recorded onto standard forms. MAPS constant-effort-bird banding data (DeSante et al. 2007) exist for six years (2008–2013) at one site within piñon-juniper woodlands and at three sites for five years (2010–2014).

4.13.4 Condition and trend

Reliable trend data are not yet available but reproductive rate information pooled across all species from MAPS stations suggests low reproduction in both piñon-juniper habitats and mixed conifer habitats.

4.13.5 Level of confidence

The level of confidence for the condition

Table 4-9. Birds, identified as species of concern by existing published conservation plans, USFWS (2008), Rich et al. (2004), Butcher et al. (2007), Norris et al. (2005), NMDGF (2006), NMPIF (2007), that are accidental, rare, migrant, etc. at Bandelier NM, and their status at the park.

Standard English name	**Scientific name**	**Status at Bandelier NM**
Dusky grouse	*(Dendragapus obscurus)*	Rare
Turkey vulture	*(Cathartes aura)*	Common, breeding is very difficult to confirm
Bald eagle	*(Haliaeetus leucocephalus)*	Rare, winter only
Northern harrier	*(Circus cyaneus)*	Accidental
Northern goshawk	*(Accipiter gentilis)*	Accidental
Swainson's hawk	*(Buteo swainsoni)*	Accidental
Zone-tailed hawk	*(Buteo albonotatus)*	Rare, probably no longer a breeding species
Ferruginous hawk	*(Buteo regalis)*	Accidental
Golden eagle	*(Aquila chrysaetos)*	Accidental
Peregrine falcon	*(Falco peregrinus)*	Rare
Prairie falcon	*(Falco mexicanus)*	Accidental
Western sandpiper	*(Calidris mauri)*	Accidental
Stilt sandpiper	*(Calidris himantopus)*	Accidental
Wilson's phalarope	*(Phalaropus tricolor)*	Migrant
Band-tailed pigeon	*(Patagioenas fasciata)*	Rare, breeding is very difficult to confirm
Yellow-billed cuckoo	*(Coccyzus americanus)*	Never has been documented at Bandelier
Flammulated owl	*(Psiloscops flammeolus)*	Rare
Northern pygmy-owl	*(Glaucidium gnoma)*	Rare
[Mexican] spotted owl	*(Strix occidentalis)*	Extripated
Black swift	*(Cypseloides niger)*	Accidental
Calliope hummingbird	*(Selasphorus calliope)*	Migrant, Common
Rufous hummingbird	*(Selasphorus rufus)*	Migrant, Common
Olive-sided flycatcher	*(Contopus cooperi)*	Rare breeding species
Willow flycatcher	*(Empidonax traillii extimus)*	Migrant, no breeding records
Gray vireo	*(Vireo vicinior)*	Never has been documented at Bandelier
Pinyon jay	*(Gymnorhinus cyanocephalus)*	Rare, may breed at Bandelier in some years
Sage thrasher	*(Oreoscoptes montanus)*	Migrant
Yellow warbler	*(Setophaga petechia)*	Migrant
Brewer’s sparrow	*(Spizella breweri)*	Migrant
Black-chinned sparrow	*(Spizella atrogularis)*	Accidental
Black-throated sparrow	*(Amphispiza bilineata)*	Accidental
Lincoln's sparrow	*(Melospiza lincolnii)*	Migrant
Lazuli bunting	*(Passerina amoena)*	Accidental
Bullock's oriole	*(Icterus bullockii)*	Rare breeding species
Cassin's finch	*(Haemorhous cassinii)*	Uncommon

and trend of landbirds at Bandelier NM is low, due to the lack of rigorous analysis of existing MAPS data. Large-scale vegetation changes associated with the Las Conchas Fire will affect population levels over coming decades. Definitive information will require targeted field work to answer management questions.

4.13.6 Data gaps/Research needs/ Management recommendations

The lack of local information on Grace's warblers is the most significant data gap. Breeding Bird Survey (BBS) data show sharp annual declines of 5.5% in New Mexico for Grace's warblers (a relatively easy to detect species) and an annual decline of 2.3% throughout western states. An analysis by Butcher and Niven (2007) suggests the species has declined an average of 1.93% per year over 40 years (1965–2005) in the U.S., a 54% decrease. Thus, the decline is persistent and appears to be range-wide in the U.S. SCPN bird community monitoring and MAPS stations at the monument should continue with integrated analyses linked to weather to provide information to management on likely causes of population changes. Unpublished data by Ro Wauer should be computerized and analyzed to provide additional historical information on bird numbers. Repeat breeding bird atlas work would provide park-wide spatial data with relative abundance information as well as specific information on level of breeding and links to habitats.

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4.14 Mexican Spotted Owl

4.14.1 Description

4.14.1.1 Ecology

The Mexican spotted owl (MSO; *Strix occidentalis lucida*) occupies primarily mixed conifer forests dominated by Douglas-fir, true fir and pine, or pine with an oak or other broad-leafed understory component. Their favored habitat is usually steep, forested canyons, often with rocky cliffs, perennial water and riparian vegetation (Gutierrez et al. 1995, U.S. Fish and Wildlife Service [USFWS] 1995, Willey 1993). The species prefers old growth forests but also occupies uneven-aged stands with complex vertical structure (Ganey and Balda 1989a).

Adult MSOs are highly faithful to breeding sites, with territory sizes in Arizona and New Mexico ranging from 7–11 square kilometers (Kroel 1991). In mixed conifer communities, breeding MSOs select sites with more mature Douglas-fir and pine canopy closure of 75% or more, and the presence of an oak understory (Seamans and Gutierrez 1995, Peery et al. 1999). In pine-oak habitat, territories may be located on more moderate slopes with 60% or greater canopy cover, and are less concentrated in canyon bottoms (Ganey et al. 2000). In steep-walled canyons, owls may also nest in cliff crevices (Gutierrez et al. 1995). In winter, lower-elevation piñon-juniper habitat may be used.

MSOs may forage and roost in a wider range of habitats than are used for nesting, but generally prefer sites with high canopy closure, live-tree basal area, available snags, and fallen logs (Ganey and Balda 1989b). Fledglings may depend on oak thickets for roosting and to avoid predator detection (Gutierrez et al. 1995). The majority of dispersing birds are juveniles (Arsenault et al. 1997), and nearly all isolated patches of mixed conifer or ponderosa pine in New Mexico and the southwest can be reached by dispersing owls (USFWS 1995). Dispersers have also established home ranges in piñon-juniper habitat (Ganey et al. 1998).

In Bandelier NM, Mexican spotted owls nest in canyons having a cool micro-environment and vegetation dominated by cool-moist species typical of mixed conifer forests. The majority of MSO habitat is located within the Bandelier Wilderness. Nesting-roosting zones cover approximately 2,800 hectares (6,900 acres or 20% of the park), have areas with steep slopes (>40%) in mixed conifer habitat types, and have been consistently monitored prior to any planned management action.

The prey base of the species in New Mexico is strongly affected by climatic variation. A recent study shows annual survival and reproduction of MSOs is positively correlated with previous year's precipitation (Seamans et al. 2002).

4.14.1.2 Status

MSOs were first reported at Bandelier NM in 1910. Management-related surveys began in 1985. As part of the park's 1998 Fire Management Plan, nine nesting-roosting zones (NRZs) and suitable nesting areas (SNAs) were created and mapped (Johnson 1998). Through formal consultation, the U.S. Fish and Wildlife Service approved the use of such designations. The park continues to manage all potential MSO habitat based on the 1998 NRZ and SNA definitions. Analysis of observations from 1977 to 1997 within the park and surrounding areas suggested that fire benefits MSOs, provided it occurs at intensities low enough to maintain essential characteristics of nesting habitat. Thus, in the 1998 Fire Management Plan, the park proposed treating NRZs with low intensity fire in an effort to reduce the threat of crown fires. Since then, 170 hectares (430 acres) of NRZ habitat have been treated with prescribed fire.

4.14.2 Reference conditions

The maximum number of occupied sites within Bandelier NM known prior to 2002 was three.

4.14.3 Data and methods

Mexican spotted owls are typically located by visiting potential habitat and looking for fresh signs (feathers and droppings). Nocturnal calling surveys are also used to elicit calls from owls, at least three times each season, if owls are not found in other ways.

4.14.4 Condition and trend

The last breeding MSOs in the monument were observed in 2002. No MSOs were detected by standard survey methods from 2003–2013. In 2011, the Las Conchas Fire destroyed most potential MSO breeding habitat in the park.

4.14.5 Level of confidence

The level of confidence for the condition and trend of MSO in Bandelier NM is high. The physical structure provided by trees in cool canyon settings is now gone because of the Las Conchas Fire of 2011. Our confidence is high that the owls are currently not breeding in the park.

4.14.6 Data gaps/Research needs/ Management recommendations

Recommendations to consider are (1) maintain large stands of mature and old-growth ponderosa pine and mixed conifer forest, particularly in areas with steep slopes; (2) protect and restrict activity in 243-hectare (600-acre) areas around known nest sites, or around roost sites if nest sites are unknown (U.S. Fish and Wildlife Service 1995, Peery et al. 1999) and (3) restore riparian habitats and restrict human use of appropriate riparian areas.

4.14.7 Sources of expertise

Stephen Fettig

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4.15 Jemez Mountain Salamander

4.15.1 Description

The Jemez Mountain (JM) salamander (*Plethodon neomexicanus*) is a federally endangered species restricted to the Jemez Mountains of northern New Mexico in Los Alamos, Rio Arriba, and Sandoval Counties, and around the rim of the collapsed Valles Caldera, with some occurrences within the caldera. The majority of salamander habitat is located on federally-managed lands, including the Santa Fe National Forest, Bandelier NM, Valles Caldera National Preserve, and Los Alamos National Laboratory, with some habitat located on tribal land and private lands (New Mexico Endemic Salamander Team [NMEST] 2000).

Relatively warm and wet environmental conditions suitable for salamander aboveground activity are likely influenced by snow infiltration and rain from summer thunderstorms (U.S. Fish and Wildlife Service [USFWS] 2012). When active above ground, the species is usually found under decaying logs, rocks, bark, moss mats, or inside decaying logs or stumps. The salamander's subterranean habitat appears to be deep, fractured, subterranean rock in areas with high soil moisture (NMEST 2000). Compacting soil makes the habitats unusable for JM salamanders. Thus, parking areas, hardened walking areas, and trails make habitat unusable. JM salamanders are normally found in mixed conifer communities, including areas dominated by Douglas-fir (*Pseudotsuga menziesii*), blue spruce (*Picea pungens*), Engelmann spruce (*Picea engelmannii*), ponderosa pine (*Pinus ponderosa*), and white fir (*Abies concolor*) with occasional aspen (*Populus tremuloides*), Rocky Mountain maple (*Acer glabrum*), New Mexico locust (*Robinia neomexicana*), ocean spray (*Holodiscus* sp.), and various shrubby oaks (*Quercus* sp.) (Degenhardt et al. 1996, Hathcock 2008, Whitford 1976, Williams 1973).

The JM salamander is strictly terrestrial, does not possess lungs, and does not use standing surface water for any life stage. Respiration occurs through the skin, which requires a moist microclimate for gas exchange. The absorption and conservation of substrate moisture is probably the most important factor in the ecology of this terrestrial salamander, as it is in other strictly terrestrial salamander species (Heatwole and Lim 1961). The species is rarely observed aboveground, and can only been found on the surface when environmental conditions are moist and temperatures warm (typically July through September; but occasional salamander observations have been made in May, June, and October (USFWS 2012). Snow infiltration and summer monsoon-generated thunderstorms provide needed environmental moisture (USFWS 2012).

The overall range-wide population size of the JM salamander is unknown because surveys tend to cover small areas (approximately 256 ft by 256 ft (200 m by 200 m). Like most plethodontid salamanders, monitoring population size or trends of the Jemez Mountains salamander is inherently difficult because of the natural variation associated with the species' behavior (Hyde and Simons 2001). For example, when the species is underground, individuals cannot be detected. The available data cannot be used to estimate population size or trends in the range-wide abundance of the salamander. Even though we are not able to estimate population trends, the number of surveys resulting in no salamander detections is increasing.

4.15.2 Reference conditions

As explained above, determining population sizes for JM salamanders is problematic because most of their lives are spent below ground. Instead, the amount of available habitat can be used as a proxy for population status. Critical habitat for an endangered species is defined by the USFWS as the specific areas within the geographical area occupied by the species which (a) are essential to the conservation of the species and (b) may require special management considerations or protection. For the JM salamander, the

Primary Constituent Elements (PCE; areas of potential habitat a species is known to use; USFWS 2013) include:

- elevations from 6,988 to 11,254 feet (2,130 to 3,430 meters)
- moderate to high tree canopy cover with greater than 50 % canopy closure that provides shade and maintains moisture and high relative humidity at the ground surface, and that
    - consists of the following tree species alone or in any combination: Douglas-fir; blue spruce; Engelmann spruce; white fir; limber pine; ponderosa pine; and aspen
    - has an understory containing New Mexico locust or oak
- ground surface in forest areas with structural features, such as rocks, bark, or moss mats that provide the species with food and cover
- underground habitat in forest or meadow areas containing interstitial spaces provided by
    - igneous rock with fractures or loose rocky soils
    - rotted tree root channels
    - burrows of rodents or large invertebrates

4.15.3 Data and methods

A complete overview of the available survey data and protocols for the Jemez Mountain salamander is reported in the USFWS 12-month finding (USFWS 2010). In summary, there are approximately 20 years of salamander survey data that provide detection information at specific survey sites for given points in time. However, like most plethodontid salamanders, monitoring population size or trends of the JM salamander is problematic due to the life history and behavior of the species (Hyde and Simons 2001). For example, when the species is underground, individuals cannot be detected, therefore, the probability of detecting a salamander is highly variable and dependent upon the environmental and biological parameters that drive above- and below-ground activities (Hyde and Simons 2001). Consequently population size estimates using existing data cannot be made accurately, and the available data cannot be used to estimate population size or trends in the range-wide abundance of the salamander.

Despite an inability to quantify population size or trends for the salamander, we can use information provided by qualitative data to make potential inferences. For example the number of surveys completed where no salamanders are detected is increasing. Disease has been implicated in the decline of many amphibians, and it is an unknown, but credible threat to the JM salamander. Cummer et al. (2005) reported chytridiomycete fungus (*Batrachochytrium dendrobatidis*) in an individual *P. neomexicanus* from Sierra Toledo on the Valles Caldera in Sandoval County. Since then at least one other individual has tested positive for *B. dendrobatidis*. Based on these inferences, the status of salamanders is likely variable across their range, and persistence may be at risk in some locations. For example, in places where the salamander was once considered abundant or common, it is now rarely detected or not detected at all (unpublished data, New Mexico Heritage Program). There also appears to be an increase in the number of areas with the above Primary Constituent Elements (PCE) where salamanders once were present, but have not been observed during more recent surveys (New Mexico Heritage Program 2010 unpublished data).

There are, however, two localities in the Valles Caldera National Preserve (VCNP) where JM salamanders are relatively abundant compared to other locations—Redondo Border located in the central portion of the VCNP, and on a slope in the northeast portion of the VCNP. Still, the number of individuals found at these two localities is at present far below the number noted in historical surveys (e.g. 659 individuals were captured in a single year in 1970 and 394 of those individuals were captured in a single month (Williams 1976). Currently, there is no known location where the number of salamanders observed is similar to that observed in 1970.

4.15.4 Condition and trend

The overall range-wide population size of the JM salamander is unknown, however, all qualitative information and observations suggest declining trends. Salamander populations are known to be susceptible to fire-related effects, including decreased forest humidity, desiccation of habitat, loss of microhabitat (such as downed logs and litter), erosion, and filling in (by runoff) of subterranean habitat utilized by salamanders (USFWS 2012). Post-fire management actions that have negatively impacted JM salamanders and their habitat include the mulching and reseeding of occupied habitat with soil-binding, non-native grasses.

4.15.5 Level of confidence

We are highly confident that JMS habitat is declining and that the overall population is likely following a similar trend. There are many aspects of the species' biology that are unknown, which limits the development of management recommendations.

4.15.6 Data gaps/Research needs/ Management recommendations

A great deal of JM salamander biology is unknown. Specific information on all subterranean aspects of the species' biology remains uninvestigated and would be valuable for developing monitoring protocols or recovery strategies. Currently there are no reliable monitoring methods for the JM salamander that do not disrupt or destroy their aboveground (under logs, rocks, moss mats) or subterranean habitats. Developing and implementing a monitoring protocol that can be used consistently across land management units would be valuable. If such a protocol was also robust with climate change forecasts, forest restoration treatments, and wildfire impacts, management options for promoting salamander resiliency would likely be greatly expanded. As summers become warmer and drier, and winter precipitation becomes more variable, it will be critical to maintain high percent cover of forest canopy in habitats most valuable to the salamander. High percent canopy cover provides shade and maintains moisture and relative humidity at the ground surface. Thus, significant fire management effort should be devoted to protecting forest canopy in areas of the park with north facing slopes and fractured-rock habitats or other rocky habitats that may contain small underground air pockets for salamander movement and overwintering. Surface habitats with bark, or moss mats, in addition to fractured-rock habitat, may be especially important for providing this species with food and cover.

4.15.7 Sources of expertise

Stephen Fettig

4.15.8 Literature cited

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4.16 Mountain Lion

4.16.1 Description

Mountain lions (*Puma concolor*; synonym: cougar) are ambush predators, unlike wolves which are short-distance pursuit predators. This hunting style conforms well to mountain lions being essentially solitary, with individual adults deliberately avoiding one another, except during the brief period of courtship (Nowak 1991, p.1205). These behaviors permit mountain lions to thrive in many habitats—at one time, mountain lions had the most extensive natural distribution of any mammal in the new world, except humans. However, mountain lions have been extirpated from much of eastern North America (Dixon 1982). By the early twentieth century, mountain lions appeared to have been eliminated everywhere in the United States north of Mexico except the mountainous part of the West and southern Texas and in Florida (Nowak 1991, p1206).

Their territorial nature means that mountain lions can be well-spaced across the landscape. When prey densities are low, mountain lion populations can survive at low densities. Individual territory sizes depend on terrain, vegetation, abundance of prey, gender, and whether a female has kittens or not. While mountain lions are relatively large in size, the species is not always the apex predator in its range. Mountain lions are known to yield to other predators, such as jaguar (*Panthera onca*), gray wolf (*Canis lupus*), black bear (*Ursus americanus*), and grizzly bear (*Ursus arctos*), a behavior that comports well with its solitary nature.

Solitary mountain lions typically are able to ambush a variety of ungulate prey including deer (*Odocoileus* sp., the most common food item for mountain lions in North America), elk (*Cervus canadensis*), moose (*Alces alces*), and bighorn sheep (*Ovis canadensis*). Estimates of deer kill frequency vary from about one every three days for a female with large kittens to one every 16 days for a lone adult (Lindzey 1987). Mountain lions are reclusive, usually avoiding people, with fatal attacks on humans (*Homo sapiens*) being rare. Attacks have trended upward in recent years as more people build homes and spend time in mountain lion habitats (Beier 1991, Chester 2006). New Mexico reported an attack in 2008, the first there since 1974 (New Mexico Department of Game and Fish [NMDGF] 2008).

4.16.2 Reference conditions

Mountain lions are present in the monument at unknown densities.

4.16.3 Data and methods

Approximately 64% of New Mexico's 315,194 km^2 (121,589 mi2) is considered mountain lion habitat (NMDGF 2010). NMDGF classifies mountain lion habitat as fair, moderate, good, and excellent based on the density of mountain lions. Excellent habitat has an adult cougar density of 3.0–4.0/100 km^2; good habitat has an adult cougar density of 1.2–1.7/100 km^2; moderate habitat has an adult cougar density of 0.6–0.9/100km^2; fair habitat has an adult cougar density of 0.4–0.5/100 km^2.

4.16.4 Condition and trend

New Mexico Game Management Unit 6, which includes Bandelier NM and includes all of Los Alamos County, parts of northern Sandoval County, northwestern Santa Fe county, and southern Rio Arriba County is roughly delineated by a circle that intersects the cities of Bernalillo, Cuba, Española, and Santa Fe (NMDGF 2011). Unit 6 contains approximately, 6,659 km2 of mountain lion habitat with an estimated population of 156–209 animals in 2011. Thus, Unit 6 is considered excellent mountain lion habitat (NMDGF 2011). If Bandelier NM has a mountain lion population in proportion to its area and is typical of unit 6, we would expect 4-6 cougars in the park. Given that mountain lion territories may cross park boundaries from neighboring agencies, the park may occasionally see additional animals.

4.16.5 Level of confidence

This assessment is based on currently available information (NMDGF 2011) and the level of confidence is high.

4.16.6 Data gaps/Research needs/ Management recommendations

Communication with the public should focus on the importance of predators for long-term healthy ecosystem function (Estes et al. 2011) and on the importance of predators in limiting or reducing grazing, as a climate change mitigation strategy (Welch 2005).

Managers should be aware that NMDGF may seek to protect bighorn sheep introduced west of the park in August 2013 by reducing mountain lion populations in some areas. NPS management policies do not support such reductions within the Bandelier NM. Park managers should undertake monitoring to establish population densities and be vigilant to possible reductions in mountain lion use of the park, if reductions are conducted on adjacent public lands.

4.16.6.1 Management considerations in the face of climate change

With wolves gone from the landscape, it is essential to maintain mountain lion populations as an essential ecosystem element. Any decrease in mountain lion populations is likely to exacerbate ungulate herbivory (Ripple and Beschta 2006; Estes et al. 2011) and a predator-prey imbalance (Berger and Wehausen 1991) and lead to a less stable situation in the face of increased climate variability (Welch 2005 p. 87; Martin and Maron 2012).

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Biological Integrity • Species of Management Concern - Overabundant Species

4.17 Elk and Deer

4.17.1 Description

Elk (*Cervus canadensis*) are the largest ungulates in New Mexico. Elk were long believed to be a subspecies of the European red deer (*Cervus elaphus*), but evidence from a study of the mitochondrial DNA indicates that the two are distinct species (Ludt et al. 2004). The Merriam's Elk (*Cervus canadensis merriami*) is the extinct subspecies of elk once found in the arid lands of the southwestern United States, including southern New Mexico. Rocky Mountain elk (*Cervus canadensis nelsoni*) were native in the Jemez Mountains until extirpated due to overhunting by around 1900 (Bailey 1931).

Although elk were present prior to 1900, there is no evidence to suggest that elk were plentiful in the region during prehistoric or historic times (Allen 1996). In 1948 and again in 1964–1965, elk were reintroduced to the region and their population grew exponentially, reaching around 7000 individuals by 1991. Data from New Mexico Department of Game and Fish suggest that by the late 1980s or early 1990s, the Jemez Mountains population of reintroduced elk had grown to levels greater than were known since at least 1200 A.D. (Allen 1996).

During the 1990s, grazing and browsing by elk and livestock became a pressing issue of public concern in the Jemez Mountains. An interagency and citizen group called "Seeking Common Ground" struggled with this issue. Current information suggests that undesirable impacts on vegetation are visible in widely distant places across the Jemez Mountains of North Central New Mexico, including in Bandelier NM (Krantz 2001, Fettig 2001, Fettig 2002a, Fettig 2003a, and Fettig 2003b).

The Rocky Mountain National Park and Estes Park environments could also be used as reference areas. This park and adjacent mountain community are physically disrupted by the migration of the elk, ranging in size from calves to full-grown 700-pound adults. Several butterfly and plant species are harmed by over-browsing by elk, especially the aspen groves that the elk herd of perhaps 3,000 animals decimates in its search for food. The elk population, while taxing the common food resources, also adversely affects native species that share the same food supply, such as beaver.

Excerpt from Allen (1996).

Faunal remains in local archeological sites and historic information suggest that elk populations in the Jemez Mountains were low from ca. 1200 A.D. through ca. 1900 A.D., when they were extirpated from this region. Elk were reintroduced to the Jemez country in 1948 and 1964–1965, and their population apparently grew exponentially, reaching 1000 animals in the 1970s and about 7000 by 1991. Elk populations in Bandelier NM and adjoining areas increased rapidly after the 1977 La Mesa Fire. Winter use by elk in the La Mesa Fire area, occurring in the monument grew from about 100 animals in 1978 to around 1500 elk by 1992. The dramatic increase in the monument's elk herd (an annual growth rate of 21.3% and a 3.6 year population doubling time) was due in part to the creation of about 6000 hectares of grassy winter range in and around the park by the La Mesa Fire. Some of this local population increase reflects concentration of elk into this favorable wintering habitat from surrounding portions of the Jemez Mountains. Existing data are inadequate to determine whether elk populations are still growing rapidly in the Jemez Mountains. While annual aerial surveys since 1990 reveal no clear population trend, a variety of observations demonstrate increasing elk use of lower elevation areas. Negative resource impacts from today's high elk populations are beginning to be widely noted across the Jemez Mountains, especially in high-use portions of the Bandelier NM area. Affected resources range from plant communities to soils and even archeological sites. Given the large uncertainties associated with the current data on elk populations, care should be taken to avoid further population increases until the resource impacts of this new phenomenon (large numbers of elk) can be identified, desirable population levels identified (based to a significant degree upon ecological information and resource carrying capacities, as well as social considerations), and appropriate cooperative management strategies implemented.

4.17.2 Reference conditions

Prior to Euro-American arrival, elk and deer populations were low due to Native American hunting and occurrence of apex carnivores on the landscape.

4.17.3 Data and methods

In 2002 and 2005, field workers investigated the influences of browsing by elk on aspen sprouts and shrubs species at randomly selected sites from a universe predominantly located within Bandelier NM's Cerro Grande area, using the Alaska Pack extension to ArcView 3.3 (Alaska Region 2005). The selection universe overlapped slightly onto the neighboring Santa Fe National Forest and Valles Caldera National Preserve. The crew visited additional randomly selected sites outside of Bandelier NM in order to evaluate any potential similarities or differences for the effects of browsing vegetation adjacent to the park. These sites were specifically located on Mesa del Rito and Sawyer Meadow.

In 2006, a study took place in which a crew of 3–4, spaced out approximately 10-75 m apart to cover the target area and identify refuges, visually surveyed the high elevations (areas of approximately 2000 m or more, or approximately 8,800 or more ft). Refuges are defined as places where woody shrubs and aspen saplings are being protected from ungulate browse. These refuges or safe zones (Turner et al. 2003) consisted primarily of fallen trees, rocks, and other natural obstacles which inhibit ungulates from reaching the young trees.

In addition to aspen (*Populus tremuloides*) sprouts, the crew searched for shrubs species within each plot to evaluate browsing by elk. These species included: wild raspberry (*Rubus strigosus*), chokecherry (*Prunus virginiana*), red elderberry (*Sambucus microbotrys*), serviceberry (*Amelanchier utahensis*), mountain maple (*Acer glabrum*), dogwood (*Cornus stolonifera*), thimbleberry (*Rubus parviflorus*), ocean spray (*Holodiscus dumosus*), mountain mahogany (*Cerocarpus montanus*), cliffbush (*Jamesia americana*), ninebark (*Physocarpus monogynus*), barberry (*Berberis fendleri*), Gambel oak (*Quercus gambelii*), currant (*Ribes* sp.), wild rose (*Rosa* sp.), whortleberry (*Vaccinium myrtillus*), mock orange (*Philadelphus microphyllus*), mountain lover (*Paxistima nyrsinites*), New Mexico locust (*Robinia neomexicana*), buckbrush (*Ceanothus fendleri*), snowberry (*Synphoricarpos oreophilus*), and shrubby cinquefoil (*Potentilla fruticosa*). Additional information is available (Krantz 2001, Fettig 2001, Fettig 2002a, Fettig 2003a, and Fettig 2003b).

4.17.4 Condition and trend

Since 2000, fieldwork at Bandelier NM has examined the influence of browsing by elk on aspen sprouts and other woody shrubs within the park's mixed conifer forests (Fettig 2002b, Fettig 2003a, and Fettig 2003b). Results have shown widespread browsing of regenerating aspen sprouts. In 2002, fieldwork on randomly selected burned plots documented a clear decline in the density of regenerating sprouts within multiple aspen clones covering a total area of 8.3 ha (20 ac). The sprout density has declined from an estimated 6,000–8,000 stems/ha (2400-3200 stems/ac) to fewer than 300 stems/ha (120 stems/ac) with the mean stem height less than 0.5 m (1.6 ft) and all sprouts browsed. Such a low sprout density with 100% browsing during the growing season, and no live canopy-height trees means that clone survival is questionable. An additional 10.1 ha (25 ac) of regenerating aspen have similar stem heights but densities are over 6,000 stems/ha (2430 stems/ac). These higher stem densities suggest that near-term clone survival is not in question, even with no sprouts able to reach tree size due to browsing. From these observations we expect that some aspen clones have a relatively high chance of declining in spatial extent over the next few years. In other places the prognosis for clone survival is good, even while aspen sprouts are not able to grow into trees.

4.17.5 Level of confidence

Our level of confidence is high for this analysis—the data are spatially robust and supported by photographs which can be re-examined.

4.17.6 Data gaps/Research needs/ Management recommendations

Additional observations of shrubs not producing seed or fruits in burned and browsed settings leads to a management question, "What is the spatial distribution of woody shrubs able to reproduce under the current level of elk browsing?"

Managers should stay informed on Chronic Wasting Disease (CWD). CWD affects the brain tissue of infected elk and is similar in symptoms to bovine spongiform encephalopathy (BSE), commonly known as mad-cow disease (MCD). There is yet no evidence to conclude that elk CWD is transmutable to humans, and research concerning CWD effect on the ecosystem is underway. As of 2010, environmental and Chronic Wasting Disease problems in Estes Park, Colorado and on a greater scale throughout the Western U.S. and North America have local, state, and federal policy-makers searching for solutions.

4.17.6.1 Management considerations in the face of climate change

At their current densities within Bandelier NM and the Valles Caldera National Preserve, elk are likely to inhibit some vegetation responses to climate change by reducing tree establishment and shrub growth. Although the interaction effects of ungulate browsing with vegetation responses to climate change are not well understood, the removal of vegetative biomass by elk may increase soil temperatures and decrease soil moisture levels (Beschta et al. 2012; also see Appendix G, ecological impacts of grazing by livestock), thus perhaps exacerbating the impacts of warmer summers on stressed vegetation and related animal communities.

4.17.7 Sources of expertise

Craig Allen, Stephen Fettig

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4.18 American Pika

4.18.1 Description

4.18.1.1 Ecology

American pikas (*Ochotona princeps*) are small lagomorphs (rabbit family) that are distributed across most of the high altitude regions of western North America. There are currently five recognized subspecies of *O. princeps*. *O. p. saxatilis*, the Goat Peak pika, is endemic to the Jemez Mountains, and is marginally found at (or closely adjoining) Bandelier NM (Hafner and Smith 2010). A different species, the collared pika, *O. collaris*, ranges from southeast Alaska to northwestern Canada and is not sympatric with *O. princeps*.

Pikas inhabit talus fields and felsenmeers (fields of broken surface rocks) located near low-stature grasses and forbs. Such sites are located near or above tree-line in alpine habitats. However, the Goat Peak pika is found in felsenmeers well below alpine zones and often with trees adjacent to the felsenmeers. Pikas do not dig burrows, but rely on existing spaces between and under rocks for homes. However, they can enlarge their homes by digging. Pikas require rocky habitat near forage plants, and have specific temperature requirements. They are generally restricted to cool, moist habitats at high elevations. However, in the northern portion of their range they may be found near sea level.

Found primarily above 1,600 m, pikas do not hibernate and spend much of the relatively short, high-altitude summer foraging and stockpiling vegetation for winter use in 'haypiles' (Conner 1983, Dearing 1996). Plant types selected for haypiles can be distinctly different from plants immediately consumed—summer foods appear to be chosen in relation to flowering times and nutritional value (Smith 1974, Huntly et al. 1986), while plants collected for storage are often high in phenols which can function as preservatives (Dearing 1996). Overwinter survival of pikas has been shown to be strongly dependent on the amount and quality of food collected and stored for the winter (Dearing 1997, Morrison et al. 2009).

Pikas have an unusually high resting body temperature. This adaptation allows them to survive cold temperatures at high elevations, but also requires that they avoid behaviors and/or environments that could lead to overheating (Smith 1974). Consequently both seasonal and diurnal behaviors are related to temperature. At lower elevation (warmer) sites, collecting and storing food for winter (called haying) begins earlier in the season and lasts later into the fall, (thus avoiding the hottest periods of the summer), and daily activity during warmer mid-day hours is reduced compared to at higher elevations (Smith 1974, Moyer-Horner 2010).

4.18.1.2 Threats and status

While pikas do occupy low elevation and anthropogenic sites, such as mine tailing piles and rock quarries (Beever et al. 2008, Simpson 2009, Rodhouse et al. 2010, Manning and Hagar 2011), it is largely accepted that the availability of appropriate habitat strongly determines pika distribution (Hafner 1994). The contraction of the pika's range with general warming during the Holocene left isolated populations at high elevations separated by uninhabitable areas (Grayson 2005, Galbreath et al. 2009). Though pikas are relatively poor dispersers across non-suitable areas (Smith 1974, Aho et al. 1998, Jeffress et al. 2013), until recently the dynamics of source-sink habitats and metapopulations apparently sustained pikas across their range (Hafner 1994, Molainen et al. 1998, Kreuzer and Huntly 2003).

Interacting, multiple effects of climate change (e.g. declining precipitation, higher temperatures, reduced snowfall), now appear to be reducing the number of sites that support pikas in some places (Smith 1974, Molainen et al. 1998, Kreuzer and Huntly 2003, Beever et al. 2003, Morrison and Hik 2007, Beever et al. 2011, Erb et al. 2011, Stewart and Wright 2012, Jeffress et al. 2013).

Pika distribution is shrinking as a result, (Beever et al. 2003, Wolf et al. 2007, Calkins et al. 2012; though see Millar and Westfall 2010a, Millar and Westfall 2010b, and Wolf 2010 for discussion).

Pikas are found across the high mountain ranges of southern Colorado and Utah and northern New Mexico in appropriate talus habitat (Hafner 1994). In the Jemez Mountains they occur in most rocky areas above about 8,800 ft, including areas of the Valles Caldera National Preserve and Bandelier NM (Valles Caldera National Preserve [VCNP] 2013, Allen 1989, Hathcock et al. 2011). The Southern Colorado Plateau Network (SCPN) mammal inventory noted pika sign in 2003 and observed individual pikas in 2004 at high elevations on Cerro Grande (Bogan et al. 2007).

A petition to list American pikas as endangered or threatened within the Endangered Species Act was submitted to the U.S. Fish and Wildlife Service (USFWS) in 2007 by the Center for Biological Diversity (CBD; Wolf et al. 2007). CBD described rising temperatures resulting from global climate change as the primary threat to pika persistence. Increased summer temperatures as a result of climate change may have the potential to adversely affect some lower and mid-elevation pika populations of *Ochotona princeps princeps, O. p. fenisex, O. p. schisticeps* and *O. p. saxatilis* in the foreseeable future; however, this does not equate to a significant portion of the suitable habitat for any of the five subspecies or the species collectively. American pika can tolerate a wider range of temperatures and precipitation than previously thought (Millar and Westfall 2009, p. 17). The American pika has demonstrated flexibility in its behavior, such as using cooler habitat below the surface to escape hotter summer daytime temperatures, and physiology that can allow it to adapt to increasing temperature (Smith 2009, p. 4). Cooler temperatures below the talus surface can provide favorable thermal conditions for pika survival below relatively warm surface environments. In 2010 USFWS determined that listing at any level was not warranted, and that, based on existing climate and other models applied in their analysis (specifically Ray et al. 2010), there was not a threat to the species or the any subspecies now or in the foreseeable future (USFWS 2010).

As far as is known there are no available population estimates from any pika populations within the Jemez Mountains. Likewise presence/absence monitoring has not been conducted for sites that have previously been identified as occupied.

4.18.2 Reference conditions

Pikas are present at unknown population levels within Bandelier NM.

4.18.3 Data and methods

As far as is known there have been sporadic observations that indicate the presence of pika in the park in the past.

4.18.4 Condition and trend

New Mexico's Comprehensive Wildlife Conservation Strategy (CWCS) lists *O. p. saxatilis* as a subspecies of greatest conservation need, as well as vulnerable and state sensitive (New Mexico Department of Fish and Game [NMDGF] 2006), and Beever and Smith (2008) have assigned *O. p. saxatalis* a status of vulnerable. Largely as a response to climate change threats, NPS has initiated pika monitoring and research as a unique program ('Pikas in Peril') within the NPS Inventory and Monitoring Program (Garret et al. 2011).

Persistence of *O. p saxatilis* populations within New Mexico and across the southern Rocky Mountains appears to be good (Colorado Division of Wildlife [CDOW] 2009, Utah Division of Wildlife [UDW] 2009), though density and trend data are not available for populations in New Mexico (NMDGF 2009, U.S. Forest Service [USFS] 2009). Rock and grassland habitats where this species breeds at Bandelier NM are at low risk in the foreseeable future, even in the face of regional drying and increased fire frequencies. Furthermore, any behavioral adaptions to increased summer temperature, such as reducing activity during the warmest part of summer days, may be compensated for by increased length of the growing season or frost-free days. However, based on other status reviews (Beever and Smith 2008; NatureServe 2009), further monitoring may be wise for *O. p. saxatilis* in the Jemez Mountains of New Mexico to obtain information on the current status of this population of the subspecies.

4.18.5 Level of confidence

The level of confidence for this analysis is high.

4.18.6 Data gaps/Research needs/ Management recommendations

During SCPN's vital signs development, the monitoring of high elevation or "boreal" mammals was considered, but not selected. No demographic rate information or population size information is available for pikas at Bandelier NM. Currently there is no information that would suggest local populations of pikas are at risk; however studies in other areas have shown that populations can disappear without physical changes to habitat. Monitoring of the Goat Peak pika (*O. p. saxatilis*) at Bandelier NM could provide helpful information to regional land managers.

4.18.7 Sources of expertise

Stephen Fettig

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Note: In contrast to Species of Management Concern, Species of Management Interest are not directly managed by monument managers, but have been of recurring interest by internal or external conservation partners in recent decades. They are reviewed here to summarize early or more recent discussions.

4.19 River Otters

River otters occur in the Rio Grande, but are not managed by Bandelier staff. They do not inhabit the internal streams of Bandelier NM and were not observed in the most recent mammal survey of the monument (Bogan et al. 2007).

4.19.1 Description

4.19.1.1 Biology

The river otter (*Lontra canadensis*) is a relatively large member of the weasel family (Mustelidae), with a combined body and tail length of between 90 and 140 cm (3–4.5 ft). River otters are much larger than American mink (*Neovison vison*), the only other aquatic carnivore in Colorado, and much smaller than beaver (*Castor Canadensis*; Rodentiae) with whom they are often sympatric. An adult river otter weighs between 5–14 kg (11–31 lb). Males have been known to range up to 50 miles. Females typically stay within three to 10 miles of their den.

River otters are well-adapted to freshwater habitats and cold climates—webbed feet with non-retractable claws facilitate swimming, capturing live prey and living in dens, a long, tapered tail propels the animal in the water, and two layers of fur provide critical insulation. All otters are carnivorous. River otters are primarily fish-eaters, but will also consume a variety of other small prey, such as crayfish, birds and small mammals.

Otters occasionally construct dens, but more commonly inhabit existing cavities in areas adjacent to calm waters, particularly beaver dens. Otters forage mostly during twilight and in the early morning hours and rest in their dens during the day. They appear to be adept at conserving energy, often sliding along snowbanks and mudflats in a fashion that appears playful but is most likely simply an efficient means of travel. Otters do not hibernate and are active throughout the winter, when they feed and travel under the ice. Breeding occurs throughout the winter beginning in December, and small litters of 1-3 pups are born between February and April.

4.19.1.2 Distribution

River otters historically ranged across North America, including Canada and all U.S. states (except Hawaii). Otter populations were decimated by the cumulative effects of fur trapping, habitat loss (stream and river impoundments) and water pollution, and were extirpated from Colorado and most of the Rocky Mountain and plains states by the early 20th century (Boyle 2006).

4.19.1.3 Status

Otters are considered a Sensitive Species by the USFS throughout Region 2 (includes Colorado). River otters have no other federal status, though NPS considers them a species of concern. The principle threats to river otters in Colorado are habitat destruction, particularly water development efforts that result in stream flow and channel morphology alteration, water pollution, and human settlement and recreational use along rivers and lakes (Boyle 2006).

According to the International Union of Conservation of Nature (IUCN, 2009) populations of otters currently occur in 1) the Rio San Pedro of Chihuahua, a tributary of the Rio Conchos entering the Rio Grande from the southeast, 2) the upper Rio Grande near the Colorado/New Mexico border, and 3) the middle Pecos River in southeastern New Mexico, which enters the Rio Grande from the west. These observations are corroborated by multiple observations by competent observers, and in the case of the first population, otter photos and sign. These populations are centered in areas with macro-habitats characterized by a river flowing through deep canyons or ancillary wetlands. Considerably more detailed survey work is needed to determine the full extent of the distribution of otters in the Rio Grande drainage. A genetic study is critically needed to determine the true taxonomic affiliation of these recently discovered populations. A moratorium on translocations should be put in place for the Rio Grande to conserve the native populations already existing.

4.19.1.4 River otter re-introduction efforts in New Mexico

Excerpt from D. Williams. Restoration begins in Rio Grande Basin in New Mexico Wildlife Vol. 53, No. 4, Winter 2008-2009.

"On Oct. 14, the magic of six federal and state agencies, one Indian Pueblo, several conservation organizations and many individuals came together with New Mexico's first release of wild river otters. It was the first time anyone had seen a wild otter in New Mexico waters since 1953, when one was caught in a Gila River beaver trap"….

River otters are highly social, playful, semi-aquatic members of the weasel family. They are believed to have once inhabited the Gila, upper and middle Rio Grande, Mora, San Juan and Canadian river systems. Early settlers occasionally mentioned otters in their journals, but references were infrequent, leading biologists to speculate that otter populations were small. In 2004, scat and tracks discovered in Navajo Lake indicated a few otters have migrated south from Colorado, one of many states that have reintroduced otters in the past 30 years.

Jim Stuart, the New Mexico Department of Game and Fish mammalogist, said river otters have thrived almost everywhere they have been reintroduced (personal communication).

"Putting otters in the Rio Grande Basin will be a good learning experience while we consider putting more in the Gila, and possibly other rivers," Stuart said. "Right now we don't know how many otters the Rio Grande system can support. The Upper Rio Grande was chosen as the first otter release site because of its reliable water flows, good food sources and relatively undisturbed habitat with little human activity. Adjacent lands are controlled by the U.S. Bureau of Land Management and Taos Pueblo, both supporters of otter restoration. A feasibility study conducted by the New Mexico River Otters Working Group and the Department of Game and Fish identified six suitable release sites: the Upper Rio Grande, the Rio Grande in White Rock Canyon, the Rio Chama from El Vado Dam to Abiquiu Lake, the Upper Gila River, Lower Gila River and the Lower San Francisco River. In 2006, the State Game Commission approved the study and directed the Department to initiate efforts to restore otters in the Upper Rio Grande and the Gila River."

"If future releases around the state are successful, limited trapping of otters may be possible," Stuart said. River otters currently are not on federal or state endangered or threatened species lists. "

"In New Mexico, otters are considered protected furbearers with no take allowed," he said. "It's quite possible that at a future date that we could see a harvest, but we'll have to wait and see."

He said historic records indicate otters were not numerous enough in New Mexico to provide trappers a good income.

In 1931, mammalogist Vernon Bailey, chief naturalist for the U.S. Bureau of Biological Survey, noted in his book, "Mammals of New Mexico," (1931) that otters were too uncommon to be of any marketing importance.

Whether New Mexico's river otter population ever reaches the numbers required to support limited hunting and trapping will depend upon the initial success of the New Mexico Friends of River Otters. The group is one of the state's largest, most diverse and dedicated organizations formed to support a single species. Members include Amigos Bravos, Taos Pueblo, Earth Friends Wild Species Fund, New Mexico Wildlife Federation, Center for Biological Diversity, Defenders of Wildlife, Four Corners Institute, Rio

Grande Chapter of the Sierra Club, Upper Gila Watershed Alliance, and the U.S. Bureau of Land Management. Melissa Savage, a member of the New Mexico Friends of River Otters, said the group is keeping its goals modest for now. “Our target is 60 otters—30 in the Rio Grande and 30 in the Gila,” she said. “We think that’s what it will take to really get them started.” ’

4.19.2 Reference conditions

River otters occur intermittently in the Rio Grande today and forage in Bandelier NM.

Due to the lack of sufficiently large fish populations as a food source. river otters most likely would not persist in the internal streams of the monument.

4.19.3 Data and methods

See Bogan et al. 2007

4.19.4 Condition and trend

River otters are making a remarkable comeback in many parts of their original range in the U.S.

4.19.5 Level of confidence

Level of confidence for this analysis is high.

4.19.6 Data gaps/Research needs/ Management recommendations

No recommendations at this time.

4.19.7 Sources of expertise

Stephen Fettig

4.19.8 Literature cited

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4.20 Lynx

4.20.1 Description

Canada lynx (*Lynx canadensis*) feed predominantly on snowshoe hares (*Lepus americanus*). They will also eat rodents and birds, and sometimes hunt larger prey, such as deer. Like many cats, they will eat carrion when it is available. They hunt both by ambush and by actively seeking out prey, varying their tactics depending on the available prey species. The lack of any documented snowshoe hare observations in the Jemez Mountains means that the presence of any sustained population of Canada lynx would be highly unlikely.

Until one radio collared animal from Colorado wondered onto the Valles Caldera, Canada lynx had never been documented in the Jemez Mountains (Bailey 1931).

4.20.2 Reference conditions

There are no faunal records indicating that lynx occurred in the Bandelier NM landscape during historic or prehispanic times.

4.20.3 Data and methods

n/a

4.20.4 Condition and trend

Canada lynx are not present at Bandelier NM and there are no verified observations for the park.

4.20.5 Level of confidence

The level of confidence for this analysis of the condition and trend of Canada lynx at Bandelier NM is very high.

4.20.6 Data gaps/Research needs/ Management recommendations

No management action is recommended.

4.20.7 Sources of expertise

Stephen Fettig

4.20.8 Literature cited

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4.21 Bighorn Sheep

4.21.1 Description

Rocky Mountain bighorn sheep (*Ovis canadensis canadensis*) occupy open, mountainous habitat, either above timberline, or in open canyons and slopes below forests and woodlands. The species is characterized by low reproductive rates, long life spans, and populations that can be bottom-up regulated by nutritional constraints (i.e., populations are limited by food availability) or top-down limited by predation (i.e., populations are limited primarily by mountain lion predation).

In the past 20 years, the state of New Mexico has made two attempts to start an interagency planning process for reintroduction of bighorn sheep into White Rock Canyon. The New Mexico Department of Game and Fish (NMDGF) assessed the post-Las Conchas Fire landscape in the vicinity of Cochiti Canyon and determined that post-fire vegetation conditions were favorable for reintroduction of bighorn sheep. An introduction location was selected that was reasonably distant from suburban sites and entirely within the boundary of the Santa Fe National Forest (SFNF), thus avoiding the interagency and domestic animal interaction issues that have characterized previous reintroduction efforts. Acting under a permit issued by SFNF, the department released fifty Rocky Mountain bighorn sheep into Cochiti Canyon in August 2014, captured from a high-elevation Sangre de Cristo population. Introduced animals represented a range of ages, both male and female. NMDGF will monitor this new population closely in coming years using their standard protocols.

Rocky Mountain bighorn sheep are thought to have never been widespread in New Mexico, with historical evidence for just four populations in Wheeler Peak, Pecos Wilderness, White Rock Canyon, and Manzano/Los Pinos Mountains (Bailey 1931, Leopold 1933). However, pre-Hispanic era populations are hypothesized to have been more widespread than recent historical accounting. In 2004, there were an estimated total of 950 Rocky Mountain bighorn sheep among three alpine and three low-elevation populations. In 2004 all three alpine populations were estimated to be > 100, and each of the three low-elevation populations were estimated to be < 100. Populations with more than 100 bighorn sheep have an increased probability of long-term persistence (Berger 1990) in New Mexico.

Two of three alpine populations are currently at carrying capacity and require trapping and removal to keep herds below carrying capacity (Hacker et al. 2000). Declines in the three low-elevation populations in New Mexico are associated with habitat loss resulting from fire suppression and livestock grazing (Huddleston-Lorton 2000), increased predation from mountain lions (*Puma concolor*) (Ahlm 2001, Huddleston-Lorton 2000, Rominger and Dunn 2000), train-strike kills (NMDGF files), and disease (Ahlm 2001). Other factors influencing bighorn sheep populations include: recreation use, roads, fences, exotic ungulates, poor range conditions, feral dogs, and illegal harvest.

Bighorn sheep have been extirpated from Bandelier NM. The species is known through scientific and archeological evidence to have existed in the park in historic and prehistoric times. Based on one skull in the Smithsonian Museum, the population of bighorn sheep in White Rock Canyon at the time of collection may have been intermediate in physical form between rocky mountain and desert bighorn sheep.

4.21.2 Reference conditions

Bighorn sheep were rare and concentrated on rocky steep slopes within White Rock Canyon.

4.21.3 Data and methods

See Ahlm (2001).

4.21.4 Condition and trend

Rocky Mountain sheep were extirpated from Bandelier NM around 1890. The NMGF reintroduced them into Cochiti Canyon in August 2014, and continues to monitor the population's status..

4.21.5 Level of confidence

The level of confidence for the condition and trend of Rocky Mountain bighorn sheep in the monument is very high.

4.21.6 Data gaps/Research needs/ Management recommendations

Based on discussions during the 1990s, there are many non-biological considerations to contemplate in any consideration of putting bighorn sheep back into White Rock Canyon.

NMDGF will have the primary responsibility for preventing nose-to-nose contact between wild sheep and domestic sheep and goats, unregulated hunting, and predation management. Bandelier NM may need to establish MOUs with American Indian pueblos to clarify what access tribes would have to animal parts (if any) for religious use.

Recommendations

1. The state of New Mexico needs to take a leading role in any establishment effort because they are in control of all source populations of bighorn sheep in New Mexico.

2. In the 1990s, potential impacts from bighorn sheep browsing were identified as an issue. This topic needs to be examined again in the light of fire-related habitat changes and expected warmer and dryer summers in the future.

4.21.7 Sources of Expertise

Stephen Fettig

4.21.8 Literature cited

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4.22 Natural Sounds

4.22.1 Description

Soundscapes are generally defined as the total amount of ambient noise in an area, measured in terms of frequency and amplitude (decibels; Ambrose and Burson 2004). Because national parks are often (perhaps wistfully) considered 'islands' of quiet (Lynch et al. 2011, Miller 2008), NPS has been working for several decades to establish baseline conditions and develop measuring and monitoring methods for soundscapes in national parks (Miller et al. 2008). Similar to the topic of light pollution, however, soundscapes have primarily been addressed in relation to visitor experiences (Rogers and Sovick 2001, Sovick 2001, Miller 2008, Lynch et al. 2011) with relatively little attention given to ecological and landscape-scale impacts of anthropogenic noise (Barber et al. 2011, Francis and Barber 2013).

'Soundscape ecology' is an emerging field within landscape ecology that attempts to connect ecological processes with human and natural sounds at landscape scales (Dumyahm and Pijanowski 2011b, Pijanowski et al. 2011, Traux and Barrett 2011). When evaluated ecologically, the impacts of anthropogenic sounds are most commonly considered in terms of effects on wildlife. (Marine studies are abundant, but herein only terrestrial systems will be discussed.) For example, studies have demonstrated the negative impacts of human-generated noise on birds (Dooling and Popper 2007, Slabbekoorn and Ripmeester 2008, Francis et al. 2009, Francis et al. 2011a), bats (Schaub et al. 2008), rodents (Shier et al. 2012), frogs (Barber et al. 2010, Bee and Swanson 2007), and invertebrates (Morley et al. 2014). Human noise is perhaps most detrimental to prey species because it can both mimic predator sounds and mask them (Landon et al. 2003, Chan et al. 2010, Brown et al. 2012).

The presence of roads and energy development facilities appear to have the greatest impacts on wildlife (as opposed to inputs such as overflights; Barber et al. 2011, Newman et al. 2014). Road noise can alter animal behavior, movement patterns, ability to find prey, and breeding processes (Reijnen and Foppen 2006, Bee and Swanson 2007, Barber et al. 2011, Siemers and Schaub 2011), while noises associated with energy development are often incessant, causing increased levels of chronic stress for animals near these sites (Bayne et al. 2008, Barber et al. 2009, Francis et al. 2011b, Blickley et al. 2012, Souther et al. 2014). Some species are able to adapt to long-term additions of noise in their environment but others are not (Barber et al. 2010). Research further suggests that due to the complex nature of sounds and the fact that impacts at individual and population scales translate up to ecosystem and process levels, ambient and pulsed noise levels perceived by wildlife should be addressed at multiple spatial and temporal scales (Slabbekoorn and Halfwerk 2009, Barber et al. 2011, Dumyahn and Pijanowski 2011a and b).

4.22.2 Reference Conditions

NPS policies direct that the absence of anthropogenic noise be the baseline against which impacts are measured, though there are places where human-generated noise is appropriate for the given purpose of a park (e.g. battle re-enactments at Gettysburg; Lynch et al. 2011). However, because the primary cultural resource protection mission of Bandelier NM is to preserve a period of human occupation wherein human noise was minimal, aside from occasional performances or demonstrations, natural quiet should be considered the reference condition for visitor experiences. Given the relative absence of information on the effects of noise on wildlife, measurements of anything over natural sounds should be considered ecologically undesirable.

4.22.3 Data and methods

Methods for measuring and monitoring sound are well-established; however, these are objective measures only, without a context of relevance to wildlife impacts (Miller et al. 2008, Villanueva-Rivera et al. 2011). In terms of human perception, a relatively high level of information on the local soundscape has been collected at the park, with methods described in Newman et al. (2014) and White (2014). Data collection methods included on- and off-site instruments as well as visitor surveys. As far as is known there have been no studies investigating the impacts of anthropogenic noise on wildlife at Bandelier NM.

4.22.4 Condition and trend

Bandelier NM is a relatively very quiet place, but the most common sources of noise at the park are aircraft overflights and vehicles (White (2014). Nothing is known regarding the ecological impacts of noise at any location in the park nor, the trend in noise levels across any time period.

4.22.5 Level of onfidence

the level of confidence for this analysis is low to moderate.

4.22.6 Data gaps/Research needs/ Management ecommendations

Data collected by White (2014) in 2012 could serve as a baseline against which to measure future noise levels, though again there is no information on wildlife impacts from conditions measured in 2012.

4.22.7 Sources of Expertise

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4.23 Air Quality

4.23.1 Background

The Clean Air ActT (CAA) requires that the air quality of national parks and '…other areas of special national or regional natural, recreational, scenic or historic value be preserved and protected' (42 U.S.C. §7470(2)). The CAA further designates 48 NPS units, including Bandelier NM, as "Class 1" areas, where special protections are provided for air quality and scenic views. Consequently, managers of Class 1 areas are required by the CAA as well as the Title 54 (54 USC 100101(a) et seq.), commonly known as the NPS Organic Act, and Management Policies to protect these values. The greatest existing threats to maintaining good air quality at Bandelier NM include regional and local sources of air pollution, such as power plants, oil and gas development, industrial facilities, agriculture, urban developments, and wild fire (NPS-ARD 2014).

4.23.2 Reference conditions

The NPS Air Resources Division (ARD) focuses on four indicators to evaluate air quality conditions and trends in national parks: visibility, ozone, nitrogen and sulfur deposition, and mercury and toxics deposition. Benchmarks have been established using regulatory standards and best available scientific evidence.

4.23.2.1 Visibility

Visibility is a human-centered concept that can also be valuable for assessing ecological impacts because it indirectly measures pollutant particles in the atmosphere from both natural and human-caused sources. The CAA established a national goal to return visibility to "natural conditions" in Class I areas, which are those conditions estimated to exist in a given area in the absence of human-caused visibility impairment. Visibility is often measured using deciview (dv) metric measures, and ARD recommends that mid-range visibility days should be < 2 dv above natural conditions (NPS-ARD 2014).

4.23.2.2 Ozone

Ozone is formed when nitrogen oxides from vehicles, power plants, and other combustion sources combine with volatile organic compounds from gasoline, solvents, and vegetation in the presence of sunlight. The National Ambient Air Quality Standard (NAAQS) for ozone is set by the U.S. Environmental Protection Agency (EPA), and is based on human health effects. The NPS ARD recommends a benchmark for good ozone condition of 60 parts per billion (ppb) or less, which is 80% of the human health-based NAAQS (NPS-ARD 2013).

In addition to being a concern for the health of park staff and visitors, long-term exposures to ground-level ozone can cause injury to ozone-sensitive plants . The W126 metric is a biologically relevant measure that focuses on plant response to ozone exposure and is a better predictor of vegetation response than the metric used for the human health standard. The W126 preferentially weighs the higher ozone concentrations most likely to affect plants and sums all of the weighted concentrations during daylight hours. The highest 3-month period that occurs during the growing season is reported in "parts per million-hours" (ppm-hrs). The NPS ARD recommends a W126 of < 7 ppm-hrs to protect most sensitive trees and vegetation.

4.23.2.3 Sulfur and Nitrogen

Sulfur (S) and nitrogen (N) compounds in the air can deposit into ecosystems and cause acidification, excess fertilization (eutrophication), and changes in soil and water chemistry (Allen and Geiser 2011).

Nitrogen, together with sulfur, can also acidify surface waters and soils, which can result in changes in community structure, biodiversity, reproduction, and decomposition (Bobbink et al. 2010; De Schrijver et al. 2011). NPS ARD recommends a nitrogen or sulfur wet deposition of less than 1 kilogram per hectare per year (kg/ha/yr) as condition to protect sensitive ecosystems (Ellis et al. 2013; NPS-ARD 2013).

4.23.2.4 Mercury and toxics

Mercury and other toxic pollutants (e.g., pesticides, dioxins, PCBs) accumulate in the food chain and can affect both wildlife and human health (Eagles-Smith et al. 2014). High mercury concentrations in birds, mammals, and fish can result in reduced foraging efficiency, survival, and reproductive success. Other toxic air contaminants of concern include pesticides (e.g., DDT), industrial by-products like PCBs, and emerging chemicals, such as flame retardants for fabrics (PBDEs). These pollutants enter the environment from historically contaminated soils, current day industrial practices, and air pollution.

The ARD does not currently assess the condition for mercury, as condition thresholds for mercury deposition have not been established. Environmental conditions play a significant role in the potential for mercury to accumulate in the food chain. Therefore, the ARD recommended condition, once developed, will take into account physical, chemical, and biological parameters.

4.23.3 Data and Methods

4.23.3.1 Condition methods

Air quality status assessments are based on current conditions compared to NPS ARD benchmarks for specific measures of visibility, ozone, and atmospheric deposition. NPS ARD uses six specific measures for three indicators to summarize current air quality conditions at the park (Table 4-10).

4.23.3.2 Overview

For each of the specific measures of air quality identified above, data from national air quality monitoring networks were reviewed and 5-year averages (2008–2012) were calculated for monitoring sites with at least 3 years of complete annual data. The Inverse Distance Weighted (IDW) interpolation method was then used to estimate 5-year average values for all locations in the contiguous U.S. The condition for Bandelier NM is the value derived from this national analysis at the geographic center of the park. The estimated 5-year averages (conditions) for the park were compared to established benchmarks to be assigned one of three status categories:

- Warrants Significant Concern,
- Warrants Moderate Concern, or
- Resource is in Good Condition.

4.23.3.3 Visibility

Visibility conditions are measured using Haze Index in deciviews (dv). Annual average measurements for visibility on mid-range days (40th to 60th percentile) were averaged over a 5-year period (2008–2012) and subtracted from the estimated natural visibility condition on mid-range days at each Interagency Monitoring of Protected Visual Environments (IMPROVE) monitoring site. The difference between current visibility and natural visibility conditions on mid-range days were interpolated for all IMPROVE locations with an Inverse Distance Weighted (IDW) estimation method to estimate 5-year average values for the contiguous U.S. The estimated current visibility condition for Bandelier NM is the value derived from this national analysis at the geographic center

Table 4-10. National Park Service indicators of air quality and specific measures. The W126 metric is a biologically relevant measure that focuses on plant response to ozone exposure.

Indicator of air quality	Specific measure
Visibility	Visibility on mid-range days minus natural visibility condition on mid-range days
	Vegetation health: 3-month maximum 12-hour W126
	Nitrogen wet deposition

of the park. A resulting condition greater than 8 dv is assigned a Warrants Significant Concern status. A current visibility condition from 2–8 dv is assigned Warrants Moderate Concern status. Resource is in Good Condition if the current visibility condition is less than 2 dv.

4.23.3.4 Ozone

Current condition for human health risk from ozone is measured using the 4th-highest daily maximum 8-hour ozone concentration in parts per billion (ppb). Annual 4th-highest daily maximum 8-hour ozone concentrations were averaged over a 5-year period (2008–2012) at all Clean Air Status and Trends Network (CASTNET) and Air Quality System (AQS) monitoring sites. For 5-year average calculations, annual ozone data must meet a 75% data completeness criterion, meaning that there must be 115 or more valid days within a year. Five-year averages were interpolated for all ozone monitoring locations with an IDW estimation method to estimate 5-year average values for the contiguous U.S. The estimated current ozone condition for human health risk at Bandelier NM is the value derived from this national analysis at the geographic center of the park. A resulting condition greater than or equal to 76 ppb is assigned a Warrants Significant Concern status. A current ozone condition from 61–75 ppb is assigned Warrants Moderate Concern status. Resource is in Good Condition if the current ozone condition is less than 2 ppb.

Current condition for vegetation health risk from ozone is measured using the maximum 3-month 12-hour W126 in parts per million hours (ppm-hrs). Annual maximum 3-month 12-hour W126 were averaged over a 5-year period (2008–2012) at all Clean Air Status and Trends Network (CASTNET) and Air Quality System (AQS) monitoring sites. Five-year averages were interpolated for all ozone monitoring locations with an IDW estimation method to estimate 5-year average values for the contiguous U.S. The estimated current ozone condition for vegetation health risk at Bandelier NM is the value derived from this national analysis at the geographic center of the park. A resulting condition greater than 13 ppm-hrs is assigned a Warrants Significant Concern status. A current ozone condition from 7–13 ppm-hrs is assigned Warrants Moderate Concern status. Resource is in Good Condition if the current ozone condition is less than 7 ppm-hrs.

4.23.3.5 Atmospheric deposition

Conditions of atmospheric deposition are based on wet deposition in kilograms per hectare per year (kg/ha/yr) only because dry deposition data are not available for most areas. Wet deposition for sites within the contiguous U.S. was calculated by multiplying nitrogen or sulfur concentrations in precipitation by a normalized precipitation amount. Annual nitrogen and sulfur wet deposition measurements were averaged over a 5-year period (2008–2012) at all National Atmospheric Deposition Program – National Trends Network (NADP-NTN) monitoring sites. Five-year averages were interpolated for all atmospheric deposition monitoring locations with an IDW estimation method to estimate 5-year average values for the contiguous U.S. The estimated current nitrogen and sulfur condition for the monument is the value derived from this national analysis at the geographic center of the park. A resulting condition greater than 3 kg/ha/yr is assigned a Warrants Significant Concern status. A current nitrogen or sulfur condition from 1–3 kg/ha/yr is assigned Warrants Moderate Concern status. Resource is in Good Condition if the current nitrogen or sulfur condition is less than less than 1 kg/ha/yr

4.23.3.6 Trend methods

Trends were computed from data collected over a 10-year period at on-site monitors. Trends were calculated for sites that have at least 6 years of annual data and an annual value for the final year of the 10 year period, i.e. 2012.

4.23.3.7 Visibility

Visibility trends were computed from the Haze Index values on the 20% haziest days and the 20% clearest days, consistent with visibility goals in the Clean Air Act, which include improving visibility on the haziest days and allowing no deterioration on the clearest days. If the Haze Index trend on the 20% clearest days was deteriorating, the overall visibility trend was reported as deteriorating. Otherwise, the Haze Index trend on the 20% haziest days was reported as the overall visibility trend. Visibility monitoring data were retrieved from the BAND IMPROVE monitor (BAND1).

4.23.3.8 Ozone

Typically, annual 4th-highest daily maximum 8-hour average ozone concentrations (ppb) and maximum 3-month 12-hour W126 are used to calculate 10-year trends for ozone. However, no trend information was available for Bandelier NM because there were not sufficient on-site or nearby ozone monitoring data.

4.23.3.9 Atmospheric deposition

Wet deposition trends were evaluated using pollutant concentrations in precipitation (micro equivalents/liter) so that yearly variations in precipitation amounts do not influence trend analyses.

For sulfur wet deposition trends, sulfate concentrations measured in precipitation were trended over a 10-year period. For nitrogen wet deposition trends, total nitrogen in precipitation was estimated using molecular weight ratios to calculate the nitrogen portions of nitrate and ammonium. The resulting ratios were summed to estimate total nitrogen concentration in precipitation and trended over a 10-year period. Wet deposition monitoring data were retrieved from BAND NADP monitor (NM07).

4.23.4 Resource condition and trend

4.23.4.1 Visibility

The visibility condition at Bandelier NM does not meet the ARD recommended benchmark for good condition. Based on 2008–2012 estimated visibility data, visibility on mid-range days at the monument was 3.8 dv above estimated natural conditions (3.0 dv), and falls within the moderate concern category. On the haziest days, visual range has been reduced from approximately 120 miles (without the effects of human caused pollution) to 60 miles because of human caused pollution at the park. Severe haze episodes can occasionally reduce visibility to approximately 4 miles (IMPROVE 2013). During the last decade (2003–2012), visibility improved on the clearest days and remained relative unchanged on the haziest days at the BAND IMPROVE monitor (BAND1) (NPS-ARD 2014). However, visibility on the clearest days appears to have deteriorated over the last three years (2010–2012).

4.23.4.2 Ozone

Current condition for human health risk from ozone at Bandelier NM does not meet the ARD recommended benchmark condition. The estimated 4th-highest daily maximum 8-hour ozone concentration from 2008–2012 is at 66.1 ppb and falls within the moderate concern status category (NPS-ARD 2014). Bandelier NM lies within Los Alamos and Sandoval counties that meet the NAAQS ozone standard of an 8-hour average concentration of 75 parts per billion (ppb). For this reason, these counties are EPA-designated “attainment” area for ozone.

Current condition for vegetation health risk from ozone warrants moderate concern status category based on estimated W126 metric of 11.3 ppm-hrs for 2008–2012 (NPS-ARD 2015). A risk assessment that considered ozone exposure, soil moisture, and sensitive plant species concluded that plants in the park were at low risk of foliar ozone injury (Kohut 2007; Kohut 2004); however, estimated ozone concentrations and cumulative doses at the park are high enough to induce foliar injury to sensitive vegetation under certain conditions (Binkley et al. 1997).

4.23.4.3 Sulfur and nitrogen

Sulfur deposition is in good condition and nitrogen deposition condition warrants moderate concern at Bandelier NM in accordance with ARD benchmarks. During the period of study (2008–2013), estimated wet sulfur deposition at the park was 0.8 kg/ha/yr and estimated wet nitrogen deposition was 1.9 kg/ha/yr (NPS-ARD 2014). During the last decade (2003–2012), the trend in wet sulfur and nitrogen deposition in rain and snow remained relatively unchanged (no statistically significant trend) at the BAND NADP monitoring station (NM07) (NPS-ARD 2014).

Nitrogen deposition at Bandelier NM is at levels known to affect diversity of plants and lichens (Pardo et al. 2011), and ecosystems in the park were rated as having high sensitivity to nutrient-enrichment effects relative to other NPS units (Sullivan et al. 2011a; Sullivan et al. 2011b). Plants in arid ecosystems such as Bandelier NM are particularly vulnerable to changes caused by nitrogen deposition; for example, invasive grasses tend to thrive in areas with elevated nitrogen deposition, displacing native vegetation adapted to low nitrogen conditions (Brooks 2003; Schwinning et al. 2005; Allen et al. 2009). The 2011 Las Conchas Fire at Bandelier NM burned over 75% of the Frijoles Canyon watershed, further impacting the park's waterways and susceptibility to acidification and nutrient enrichment. In addition, weed density is known to increase in post-fire environments with higher soil N levels (Floyd-Hanna et al. 2004).

In addition to assessing wet deposition levels, critical loads can also be a useful tool in determining the extent of deposition impacts (i.e., nutrient enrichment) to park resources. A critical load is defined as the level of deposition below which harmful effects to the ecosystem are not expected.

Pardo et al. (2011) suggested a critical load of 3.0–8.4 kilograms nitrogen per hectare per year (kg N/ha/yr) to protect lichen and herbaceous vegetation in the North American Deserts ecoregion; and 2.5–17.0 to protect lichen, forest, and herbaceous vegetation in the Northwestern Forested Mountains ecoregion. To maintain the highest level of protection in the park, the lower end of this range would be an appropriate management goal. The estimated maximum 2010–2012 average for total nitrogen deposition in the North American Deserts ecoregion of Bandelier NM was 2.6 kg/ha/yr and 4.5 kg/ha/yr in the Northwestern Forested Mountains ecoregion (NADP 2014). Therefore, total deposition levels in the park are below ecosystem critical loads in the North American Deserts ecoregion and above ecosystem critical loads in the Northwestern Forested Mountain ecoregion, suggesting that in some parts of the park, lichen, forest, and herbaceous vegetation are at risk for harmful effects.

4.23.4.4 Mercury and other toxins

Landers et al. (2008) detected airborne toxics in passive air samples and vegetation (lichen, conifer needles) at Bandelier NM. Compared with concentrations at all 20 western national parks, concentrations detected in the park's air sampler were near the median. Pesticides detected included both current-use and historic-use. Baker (2013) also detected persistent toxic compounds, such as DDT, in the park soils at levels that may pose a health risk to humans and wildlife, including fish. The predicted concentrations of methylmercury (the toxic, bioavailable form of mercury) in surface waters at the park are moderate, as compared to other NPS units (USGS 2015).

4.23.5 Level of confidence

The degree of confidence in the condition and trends of visibility at Bandelier NM is high because there is an on- site visibility monitor. The degree of confidence in the condition and trends of ozone at the park is medium because there is not an on-site or nearby representative ozone monitor with enough data to calculate a trend. The degree of confidence in the condition and trends of sulfur and nitrogen at the park is high because there is an on-site wet deposition monitor.

4.23.6 Data gaps/Research needs/Management recommendations

Data and planning priorities for improving air quality at Bandelier NM include:

- continued support for existing in-park air quality monitoring;
- increased monitoring of ozone;
- additional support for monitoring air quality and mitigating impacts during wildfire events;
- applied research to better understand air pollution impacts on sensitive park ecosystems, including the potential impact of mercury and other toxics;
- continued management direction that emphasizes efforts to protect air quality and minimize impacts to biota;

4.23.7 Sources of expertise

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Chapter 5: Toward Ecosystem Integrity in a Warmer Drier Future: Don't Panic!

5.1 State change

For centuries, people living and working within the current boundaries of Bandelier National Monument (NM) have both driven, and responded to, landscape change. Since the transfer of the land to National Park Service management, that fact has remained the same. Heavy grazing prior to and during early federal management of Bandelier initiated fire exclusion across most of the monument. Combined with a policy of active fire suppression and the timing of favorable wet climate windows, land managers were able to manage ecosystem processes and temporarily create an artificial semblance of stability across much of the landscape for most of the 20th century.

The fragility of this fabricated state became apparent when historical ecologists reconstructed fire histories from local sources and found that, at least since the 1500s until the 1890s, fires were frequently burning across the Jemez Mountains, including on the Bandelier NM. These frequent, low severity fires, left behind open forests with grassy understories (see section 4.01 Fire History & Ecology).

Fire exclusion has promoted dramatic increases in upland forest density and fuel loading during the early and mid-20th century. Subsequently, large and severe crown fires have burned over large areas, converting many forested areas into savannas, shrublands, or grasslands. Large patch sizes of high-severity burn areas may preclude natural tree seed recruitment for centuries, while shrub and grass understories, which now dominate many sites, will reinforce type conversion through altered disturbance regimes.

Surveys of archaeological site conditions, along with intensive ecohydrological research near the ancestral pueblo of Frijolito, provided evidence of other widespread landscape changes by revealing that late 20th century levels of soil erosion were much higher than in the past, likely a reverberation effect from historic grazing. Following the recognition of pervasive and accelerated erosion rates, managers initiated the piñon-juniper ecological restoration effort, which treated large portions of the low elevation mesa tops within Bandelier NM (see section 4.06-1 Piñon-Juniper Erosion).

In recent decades Bandelier resource managers have shifted management philosophy toward trying to preserve processes and basic ecological capital to foster incremental landscape adaptation to changing conditions; the piñon-juniper woodland restoration project is a strong example of this approach. Because of historical human intervention prior to the park's establishment, many keystone ecosystem processes and species are missing or declining (see section Plant Species of Management Concern). Events like the early 2000s piñon mortality and the 2011 Las Conchas Fire have caused widespread and rapid ecosystem state change that have forced Bandelier land mangers into a reactive stance.

Following an ecosystem state change, many organisms strongly associated with or dependent on particular vegetation communities are at risk, as these systems reorganize into new states, degrade, and/or become more limited in extent, even though new ecosystem conditions will benefit some set of current and/or new species. The post-disturbance ecosystems (e.g., treeless landscapes) may follow successional trajectories that result in characteristics and processes that differ from the pre-disturbance ecosystem (e.g., forested landscapes), ranging from different habitat availabilities and fire regimes to altered hydrologic properties. Management intervention may be required to shift the post-disturbance trajectories away from undesirable landscape conditions. At Bandelier NM, where more than half of the park is congressionally-designated as wilderness, any such management actions would need to be considered within the context of the park's legislated purpose and with regard to wilderness values.

Many ecological systems within the monument are currently at high risk of rapid system shifts away from historical conditions due to cumulative impacts of recent drought-insect-fire-flood disturbance and ongoing climate warming. These risks are not spatially homogenous and the effects of disturbance interaction will likely be patchy. For example, in the aftermath of the 2011 Las Conchas Fire, it appears that upland forests and woodlands within the monument fared much better than those on adjacent national forest lands where large expanses are now treeless. This could be a result of many factors, such as multiple fires, daily fire behavior, or vegetation densities, all of which are associated with a high degree of randomness.

In short, if the Bandelier landscapes of the 20th century were defined by an illusion of artificial stasis brought on by management's aim for *control* of nature, the landscapes of the 21st century will be defined by substantial dynamic changes, with management aiming to foster ecological integrity and adaptive capacities of natural systems.

5.2 Future state

The American Southwest is projected to experience dramatic changes in temperature and precipitation patterns in response to increasing levels of anthropogenic CO_2, which, if realized, would fundamentally alter vegetation composition and structure within the park, probably through a series of severe disturbances not unlike what we have already been experiencing since 2000 (cf. Williams et al. 2013).

Recent climatic conditions for Bandelier NM are already shifting beyond the historical range of variability, with average annual temperature for the decade from 2003 to 2012 exceeding that of any other decade for all 10-year periods from 1901 to 2012 (Monahan and Fisichelli 2014; Appendix A). The ecological responses to recent climate change effects have been so profound that they likely constitute tipping points for many vegetation communities; i.e., the sort of ecological system changes anticipated in response to predicted patterns of warming and drying in the American Southwest, and beyond.

Future climate projections for the Southwest, including Bandelier NM, have been generated from multi-model averaged data by Kunkel et al. (2012). Mean annual temperature, compared with the 1971-1999 average, is projected to increase 3–5 °F by mid-century and 5–8 °F by the end of the century, depending on the greenhouse gas emissions scenario. Current greenhouse gas emissions are on a trajectory similar to the higher emissions scenarios (Peters et al. 2013). Warming by mid-century is projected for all seasons, with the greatest increases likely in summer. Wide agreement exists among individual climate models in the direction and magnitude of warming over the coming decades. Total annual precipitation may decrease slightly by mid-century (Kunkel et al. 2012); however, precipitation variability is likely to remain large over the coming decades, and there is greater uncertainty in precipitation than in temperature projections (Kunkel et al. 2012).

5.3 State of management

Bandelier serves as a potential harbinger of the types of changes projected for the Southwestern U.S. driven by anthropogenic global warming. The recent decadal drought, fires, and insect outbreaks have all created a landscape much different from one that would be familiar to the park's namesake. Under current conditions, agency managers need to fully realize that the window of opportunity for restoration of degraded systems is closing with each passing dry year. The rapid changes unfolding may preclude management actions beyond monitoring in the near term, as many ecological systems are simply overwhelmed and make adjustments to the new conditions. Although drought conditions and destructive fires have seemingly become the norm, it is likely that ocean currents will create conditions that result in a wetter period for the Southwest and Bandelier NM.

Managers should both prepare for a continuation of recent dry conditions AND contingently plan for wet conditions and be prepared to take advantage of what may be the last window to restore ecosystem processes or intervene in current ecosystem trajectories. Such planning will be more likely to succeed if managers can fundamentally change their management emphasis away from a traditional approach that attempts to maintain historical conditions toward one that incorporates promoting ecosystem integrity and resilience (NPS 2010). Generally, managing for ecological integrity (*sensu* Woodley et al. 1993, Woodley 2010) or resilience (*sensu* Holling and Gunderson 2002) means an ecosystem maintains its full complement of native species and the processes to ensure their survival, as well as the means to withstand and recover from disturbances caused by natural environmental or human factors.

Managing for such broadly defined goals as ecosystem integrity or resilience can be challenging, to say the least. Identifying key ecosystem characteristics or processes that lead to integrity or resiliency may provide an initial starting point for a discussion on management in an uncertain, and changing future. For example, managers could choose broad ecological integrity objectives with the intent of 1) relying on historical information for guidance but not as blueprint for future designs, and 2) maintaining landscape-scale adaptive capacity by retaining keystone species and ecosystem processes. Regardless of the exact details of future management strategies, Bandelier NM managers should seek to:

- document the changes occurring across our landscape
- promote constructive and transparent discussions about management strategies
- broaden resource management frameworks by integrating a range of disciplines, scales, and approaches
- engage in broad landscape-scale efforts across jurisdictions

Given the high likelihood of climate change to drive big, fast, and patchy landscape rearrangements (Breshears et al. 2011), managers should anticipate a quickening pace for acquiring and incorporating resource information as a basis for management action. Successful managers will confront and overcome bureaucratic constraints on management action. These are difficult challenges, but if not accomplished we will fall short of our NPS obligations to the public as caretakers of "America's Best Idea."

5.4 Opportunities for success

The authors of this report provide the following guiding recommendations for Resource Managers at Bandelier NM.

5.4.1 Facilitate conservation planning across a broader landscape and develop multi-partner, multi-agency approaches to resource management.

Conversations across jurisdictional boundaries are likely to yield positive outcomes for landscape conservation. As stressors become recognized at landscape scales, action on a similar scale would likely be appropriate. Without local relationships in place, it would be a much more difficult and slower process to react to problems as they arise. The separate agencies in the Jemez Mountains all have obligations that may be different on the edges but are all based around a similar core set of conservation principles. Fostering conversations and/or planning efforts around shared core management principles could go a long way to furthering, or healing, relationships with neighboring land management agencies.

5.4.2 Support a climate change vulnerability assessment for the Jemez Mountains landscape.

Use climate change vulnerability assessment planning efforts to theorize potential management actions. Search for proactive strategy options; identify affirmative steps and options for management action. Retrospectively analyze the previous decades of fire management efforts, BAER programs, and piñon-juniper restoration to learn from past successes and failures.

5.4.3 Initiate NPS Resource Stewardship Strategy planning efforts for Bandelier National Monument.

As a Resource Stewardship Strategy (RSS) is undertaken, we see the following lines of inquiry based on Parks Canada's approach to assessing ecological integrity (Woodley 2010), as helpful for getting that effort started:

- What are the status and trends of species and communities in the park and in the encompassing Jemez Mountains landscape?
- What are the status and trends in ecosystem processes (e.g., disturbance regimes, pollination)?
- Are ecosystem trophic levels intact? For example, is hunting of predators (or predator control) outside the park influencing predator populations and thereby ecosystems within Bandelier NM?
- Do biological communities exhibit an appropriate mix of age classes and spatial arrangements that will support native biodiversity?
- Are productivity and decomposition processes operating within acceptable limits?
- Are ecosystems cycling water and nutrients within acceptable limits?

5.4.4 Evaluate options to integrate the Resource Management and Fire Management programs.

A programmatic transition of fire management from hazard fuel reduction to more ecologically based objectives, using combinations of available restoration techniques, will require 1) focused research to refine restoration targets and methods; 2) coordinated application at landscape scales, and 3) validation by a companion monitoring component that enables an adaptive management approach. Methods, restoration targets, and supporting documentation for mechanical and other restoration treatments to thin mature tree size classes are available regionally, but have not been evaluated or adapted for local conditions. Modeling is a useful tool, especially if used retroactively with existing, local response data to refine expected outputs. At Bandelier, mechanical thinning techniques may be considered for forested systems to produce selected mortality in canopy components, while leaving heavy fuels vertical, at least in short term. Pre-treatment mechanical thinning and litter fuels treatments may also be useful for protecting old growth or high value, fire intolerant individuals. Using prescribed fire as a blunt but intelligent tool that restores appropriate stand structure across variable site conditions through time is a low intervention approach that is desirable for many natural areas. However, it may require longer time periods to reach historical structural densities; leaving the system vulnerable to crown fire during the interim period. Conversely, increasingly more frequent and larger scale occurrences of crown fire and other landscape scale disturbances (e.g., drought and associated disease/ insect outbreaks amplified by high stand densities) should be modeled to determine if longer term, less intrusive, restoration approaches are timely enough to meet management goals.

At Bandelier NM, focused restoration studies have primarily addressed lower elevation woodland communities. A ten-year research program documenting the effects of past land use, as well as testing proposed treatment options, won support for its uniquely interventionist restoration treatments from many environmental and wilderness advocates, including the original three proponents for establishment of the Bandelier Wilderness. Ecological restoration of the balance of Bandelier's vegetation communities, mostly higher-elevation forested areas, has been pursued primarily through the reintroduction of fire disturbance with the goal of providing some stability and ecosystem resistance (see Fulé et al. 2012). While the central role of fire in maintenance of these forested systems is well documented from a historical perspective, there is some question as to whether fire treatments alone will continue to constitute a viable, long-term restoration strategy. For example, during the past 37 years, nearly half of the park's ponderosa pine community has experienced destructive crown fire. Despite an active prescribed fire program for nearly 15 years, restoration of fire process and historic structure/composition to forested systems at Bandelier NM remains a challenging goal.

5.4.5 Be alert to the potential for nonnative plant species already present in the park to become major ecosystem stressors.

Bandelier NM has been extensively invaded by cheat grass, and smooth brome is poised to expand into the monument from the Santa Fe National Forest on the western boundary. If heating and drying trends continue in the region, the presence of these species within the monument may become more impactful. The legislatively-mandated Alamo Watershed Ecosystem Study of three watersheds adjacent to and within Bandelier NM may provide an opportunity to evaluate management actions for smooth brome. The study is intended to provide a unified vision and determine best ecosystem management practices for watersheds that are jointly managed by the Santa Fe National Forest and Bandelier National Monument. Investigation of cheat grass is ongoing at multiple locations throughout the American Southwest, reflecting the difficulty of controlling this damaging plant.

5.4.6 Develop opportunities for land management experiments and take advantage of available "natural experiments."

Given high levels of uncertainty in future landscape conditions, managers should work to design experiments to compare alternative management strategies in order to find potential solutions to challenges brought on by rapidly changing conditions. There is also potential to take advantage of ongoing "natural experiments" (e.g., between NPS and USFS strategies or differences between treated vs. untreated forest). To learn from such "natural experiments", the differing management treatments must be well-documented, and the managers involved must adopt a shared monitoring plan that includes pre-treatment data.

5.4.7 Interpret climate change-driven landscape changes and the human response to them, both past, present, and future.

The dynamic landscape of Bandelier NM provides a core story that is ripe to be shared through the monument's interpretive program, incorporating both current events and earlier climate stress events that reshaped human occupancy of the Jemez Mountains.

5.5 Addressing uncertainty

Ongoing learning about the landscape will be essential for successful resource management. It must include provision for real-time or near real-time information development at multiple scales, ranging from routine backcountry observations, to long-term plot-based monitoring and research, to remote sensing. There should be agency flexibility to ramp up monitoring for locations that exhibit rapid dynamic change.

5.5.1 Data gaps and research needs

Many data gaps in basic inventory, long-term monitoring, and research have been identified during this NRCA process. Closing these information gaps will be an important first step toward informed decision-making.

- **Investigate geomorphic and vegetation changes in Frijoles Canyon.** Analyze recently acquired LiDAR data and acquire/analyze multi-spectral remote sensing imagery to better understand recent changes in Frijoles Canyon. The wealth of LiDAR data collected for this canyon and watershed between 2010 and 2014 has created a promising research opportunity that would inform park managers on the pace and extent of geomorphic and vegetation change.

    - Acquire a complete time-series of remote sensing products from a consistent source (e.g., Landsat) with multi-spectral remote sensing imagery to track changes in physical landform and vegetation cover.
    - Analyze recently acquired LiDAR data to better understand recent changes in Frijoles Canyon where most of the LiDAR was taken.

- **Update the park vegetation map** to reflect changes since 2004 due to widespread drought, insect outbreaks, fire, and flood effects. During a multi-year drought from 2000 to 2004, over 95% of the mature piñon pine within the monument died, including the oldest trees which exceed 300 years in age. Mature piñon was nearly erradicatedfrom the woodland zone, essentially type converting the piñon-juniper zone where the majority of cultural resources are located to juniper woodland.

  In 2011 the Las Conchas Fire burned over 60% of the monument, severely altering monument vegetation. In addition to these natural disturbances, park management implemented a landscape-scale mechanical thinning restoration effort within the woodland zone to mitigate soil erosion issues between 2007 and 2010.

  Thus, a vegetation map of the park based on 2004 photography is now well out-of-date along with soils information based on 2000-era surveys. The timing and pattern of drought and fire induced vegetation and soil-hydrologic changes need to be fully documented and quantified to support effective management of this rapidly changing cultural landscape in order to inform activities to protect the embedded archeological resources.

  Status: development of an updated vegetation map is ongoing through a Cooperative Agreement with the University of New Mexico Natural Heritage Program.

- **Investigate potential for fire management actions to contribute to ecosystem resilience.** Ecological restoration of Bandelier's montane forests has been pursued primarily through the reintroduction of fire disturbance with the goal of providing some stability and ecosystem resistance. While the central role of fire in maintenance of these forested systems is well documented from a historical perspective, there is some question whether fire treatments alone continue to constitute a viable, long-term restoration strategy. For example, during the past 37 years, nearly half of the park's ponderosa pine community has experienced destructive crown fire, with limited prospects for conifer regeneration in some areas. Despite an active prescribed fire program for over 30 years, the restoration of fire as a natural process and of historic vegetation structures and compositions to forested ecosystems at Bandelier continues to be challenging in an increasingly dynamic global-change world.

- **Research poorly known groups** like: soil bacteria, earthworms, fungi, moths, pathogens, insects (e.g., native bees as pollinators), and non-vascular plants (e.g., mosses) and lichens).

- **Quantify predator population sizes** to track this key trophic layer.

- **Replicate breeding bird atlas data collection** from the early 2000s to provide documented spatial changes in breeding species with explicit connections to habitat types. (See Appendix F for information on bird demographic needs).

- **Monitor post-disturbance ecosystem recovery** to
    - Identify areas of potential post-disturbance landscape-scale type-conversion.
    - Assess suitability and timing of native fish reintroduction by monitoring riparian vegetation, channel geomorphology, water chemistry and macroinvertebrates.

    - Identify trajectory of piñon-juniper systems following ecological restoration, and drought mortality.
    - Assess effectiveness of wildfire fuels treatments and prescribed fire.
    - Detect exotic invasions in disturbed and successional areas.
- **Monitor plant phenology** to provide proxy for climate change and incorporate into national phenological networks. Plant-based phenology observations would complement SCPN land surface phenology monitoring using MODIS imagery.

5.6 Conclusions

As Bandelier NM approaches its centennial as a protected natural area and federally-designated wilderness, managers have come to recognize that its protected status neither heals nor hides the wounds of past land use; thus monument landscapes bear stark tribute to past mistakes and mismanagement. Although set aside and protected, it has sustained losses of ecological integrity, biological capital, component species, and connectivity. Even so, the application of restoration treatments in an attempt to repair past damage remains controversial because it challenges basic philosophical and legal tenets for land management in national parks and wilderness areas to let nature be 'free willed' or largely absent the effects of human intervention—including federal agency management. Among many environmental advocates, there is both concern and suspicion that if intensive, broad-scale restoration treatments can be justified and implemented in one natural area, then the door is open for all public land to become more vulnerable to both well-intentioned and potentially damaging management actions.

Understanding that these concerns are founded on previous adverse experience, we advise future managers to rely on the best available science as they weigh alternative management decisions and actions. The recent piñon-juniper woodland restoration project exemplifies the incorporation of decades of research and pilot treatments into developing a restoration strategy. It also demonstrates a philosophical shift toward fostering incremental landscape adaptation to changing conditions. Even if near-term management is tilted toward natural recovery of post-disturbance landscapes, those decisions should be made in an open and active decision-making framework.

With the pace of landscape changes likely to quicken, managers may not have the luxury of decades of research to support future management decisions. However, amidst the uncertainty of modeling system trajectories, the questions remain—what interventions are possible to foster resilience and reduce stressor impacts, and when do we have enough knowledge, even if it is far from complete, to act?

The best management strategies will incorporate all aspects of ecosystem understanding. Such a holistic approach will require integration of solid science, across all disciplines, at a landscape level. The benefit of exercises like a resource stewardship strategy or climate change vulnerability assessment is that they provide opportunities to develop multi-management possibilities and understand the tradeoffs that are inherent in integrated landscape scale management. Important ecosystem components of the Bandelier landscape are presented in Chapter 4 in a compartmentalized manner with the hope of simplifying complex systems. It would be a mistake to view any one component as isolated from the rest, and only by thinking of them as an integrated whole can we hope to avoid deleterious intervention and long-term landscape scars.

Maintaining the stability of core management principles will serve as a strong foundation from which to build successful resource management strategies as priorities shift with changing conditions. The landscape that comprises Bandelier National Monument, through its associated native cultures and post-Hispanic land-based cultures, has exemplified strong linkages between human and "natural" worlds for many centuries.

Moving forward, we must remember that both worlds are, and will remain, closely intertwined as we seek to steward a healthy and sustainable future.

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Appendix A: Climate and Climate Change at Bandelier National Monument

This appendix includes the following two reports on climate and climate change relating to Bandelier National Monument, New Mexico:

Recent Climate Change Exposure of Bandelier National Monument
National Park Service Resource Brief
Natural Resource Stewardship and Science, U.S. National Park Service, Washington, DC
July 24, 2014

Climate Change, Impacts, and Vulnerabilities, Bandelier National Monument, New Mexico
Patrick Gonzalez
Natural Resource Stewardship and Science, U.S. National Park Service, Washington, DC
May 8, 2014

Climate Change

Resource Brief

National Park Service
U.S. Department of the Interior

Natural Resource Stewardship & Scienc

Recent Climate Change Exposure of Bandelier National Monument

Climate change is occurring at especially rapid rates in some areas of the U.S. In national parks, climate change challenges the ability of park managers to preserve natural and cultural resources. To understand the "climate change exposure" of national parks—that is, the magnitude and direction of ongoing changes in climate—we investigated how recent climates compare to historical conditions. This recently published research (Monahan & Fisichelli 2014) updates the basic climate inventories for 289 national park units. Here, we summarize results for Bandelier National Monument, including areas within 30-km (18.6-mi) of the park's boundary.

We evaluated climate-change exposure by asking which of 25 biologically relevant climate variables recently (past 10–30 years) experienced "extreme" values relative to the 1901–2012 historical range of variability. We define "extreme" conditions (e.g., extreme warm, extreme wet) as exceeding 95% of the historical range of conditions.

Methods

To evaluate recent climate values within the context of historical conditions, we used the following methods (also illustrated in Figure 1):

- **For each temperature and precipitation variable, we analyzed data within three progressive time intervals, or "moving windows," of 10, 20, and 30 years to calculate a series of averages over the entire period of analysis (1901–2012).** For example, in progressive 10-year intervals, we calculated averages of temperature and precipitation for 103 blocks of time (1901–1910, 1902–1911 . . . 2003–2012), and repeated this approach for the 20 and 30-year "windows." This type of analysis helps to smooth year-to-year fluctuations in order to identify longer-term trends that characterize the park's historical range of variability (HRV). The three windows encompass both near- and long-term management and planning horizons, as well as important climatic periods and cycles.
- **We compared the average temperature and precipitation values for each of the *most recent* 10, 20, and 30 year intervals (2003–2012; 1993–2012; and 1983–2012) to those of all corresponding intervals across the entire period of 1901–2012. These results (expressed as percentiles) describe "recent" conditions relative to historical conditions.** For example, a 90th percentile for annual average temperature over the most recent 10-year interval (2003–2012) means that the annual average temperature during this time exceeded 90% of annual average temperatures for all 10-year periods from 1901 to 2012.
- **We then averaged the percentiles of the most recent 10, 20, and 30-year time periods and computed the maximum difference in recent percentile.** For each park and climate variable, this resulted in both an overall measure of recent climate change exposure with respect to HRV (dots in Figure 2), and an estimate of sensitivity to moving-window size (length of bars in Figure 2).

See Monahan & Fisichelli (2014) for a detailed explanation of methods, and Figure 1 for an example analysis applied to annual mean temperature.

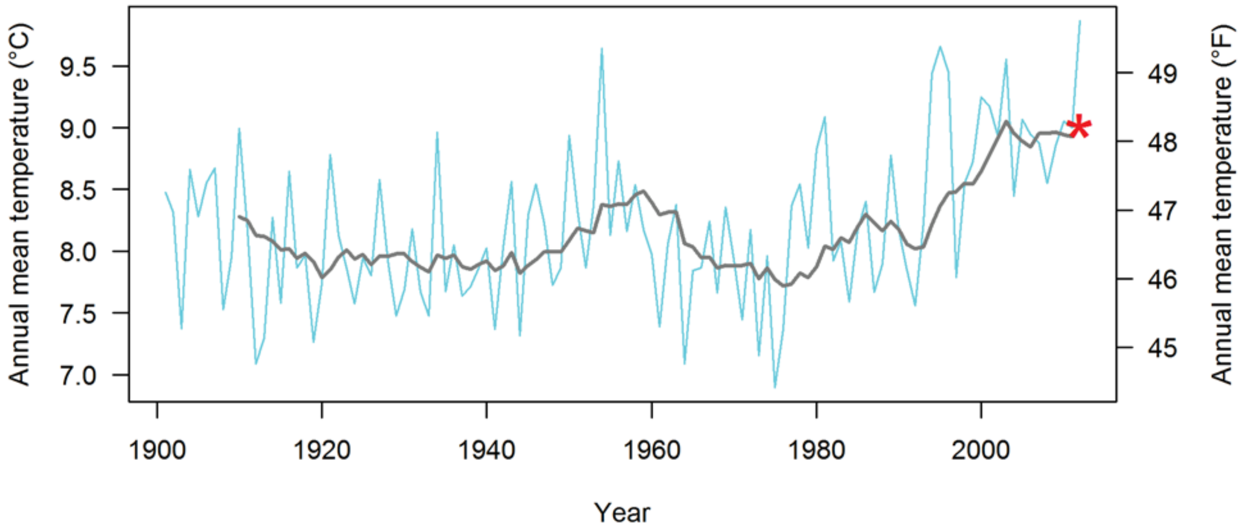

Figure 1. Time series used to characterize the historical range of variability and most recent percentile for annual mean temperature at Bandelier National Monument (including areas within 30-km [18.6-mi] of the park's boundary). The blue line shows temperature for each year, the gray line shows temperature averaged over progressive 10-year intervals (10-year moving windows), and the red asterisk shows the average temperature of the most recent 10-year moving window (2003–2012). The most recent percentile is calculated as the percentage of values on the gray line that fall below the red asterisk (see results of most recent percentiles for all temperature and precipitation variables in Figure 2).

Results

Recent percentiles for 14 temperature and precipitation variables at Bandelier National Monument appear in Figure 2. Results for "extreme" variables at the park were as follows:

- Three temperature variables were "extreme warm" (annual mean temperature, mean temperature of the driest quarter, mean temperature of the warmest quarter).
- No temperature variables were "extreme cold."
- Two precipitation variables were "extreme dry" (precipitation of the wettest month, precipitation of the warmest quarter).
- No precipitation variables were "extreme wet."

Key points for interpreting these results in the context of park resources include:

- Recent climatic conditions are already shifting beyond the historical range of variability.
- Ongoing and future climate change will likely affect all aspects of park management, including natural and cultural resource protection as well as park operations and visitor experience.
- Effective planning and management must be grounded in our comprehension of past dynamics, present conditions, and projected future change.
- Climate change will manifest itself not only as changes in average conditions, as summarized here, but also as changes in particular climate events (e.g., more intense storms, floods, or drought). Extreme climate events can cause widespread and fundamental shifts in conditions of park resources.

Climate Change Adaptation

These findings can inform climate change adaptation at Bandelier National Monument by helping park managers, planners, and interpreters to understand how recent climates compare to past conditions. For example, these findings may be used to:

- Characterize park exposure to recent climate change in a vulnerability assessment.
- Develop plausible and divergent futures for use in a climate-change scenario planning workshop.
- Synthesize desired future conditions (i.e., reference conditions) for use in a Resource Stewardship Strategy or other National Park Service management plan.
- Create interpretive materials for communicating with local communities and park visitors.

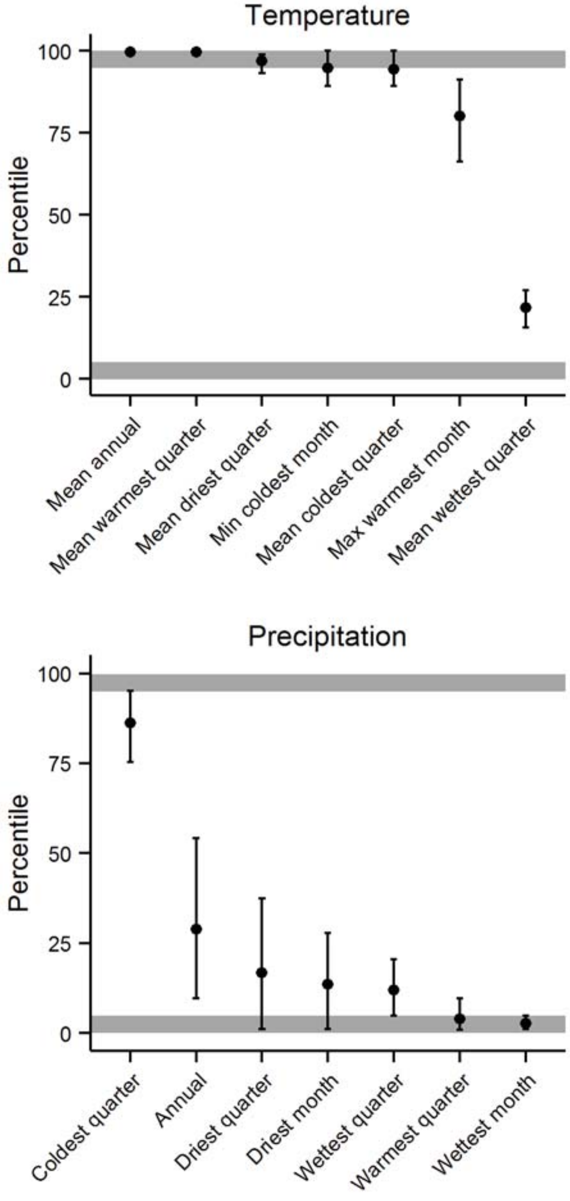

Figure 2. Recent temperature and precipitation percentiles at Bandelier National Monument (including areas within 30-km [18.6-mi] of the park's boundary). Black dots indicate average recent percentiles across the 10, 20, and 30-year intervals (moving windows). Variables are considered "extreme" if the mean percentiles are <5th percentile or >95th percentile (i.e., the gray zones, where recent climate is pushing the limits of all observed climates since the year 1901). Black bars indicate the range of recent percentiles across 10, 20, and 30-year moving windows (larger bars indicate higher sensitivity to moving window size).

More Information

Monahan WB, and NA Fisichelli. 2014. Climate exposure of US national parks in a new era of change. PLoS ONE 9(7): e101302. doi:10.1371/journal.pone.0101302. Available from http://dx.plos.org/10.1371/journal.pone.0101302.

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Climate Change, Impacts, and Vulnerabilities, Bandelier National Monument, New Mexico

Patrick Gonzalez
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May 8, 2014

Climate Trends for the Area within Park Boundaries

- Temperature is increasing at a rate of up to 1.3° C per century, but the rate for the park as a whole is not statistically significant due to less warming in the northwest (Figures 1, 3).
- Precipitation is increasing at a statistically significant rate of 30% per century for the park as a whole, but precipitation has decreased in some areas (Figures 2, 4).
- Analyses of atmospheric measurements and other data show that emissions from cars, power plants, and other human activities are causing climate change (IPCC 2013).
- If we do not reduce our emissions, models project substantial future warming and slight decreases in precipitation (Figures 5-7).

Past Impacts

- Drought in the 1950s caused extensive mortality of ponderosa pine at Bandelier and shifted piñon-juniper woodland upslope into ponderosa pine forest (Allen and Breshears 1998).
- Drought and bark beetle infestations in the early 2000s caused extensive dieback of piñon pine at Bandelier (Breshears et al. 2005). This and numerous other cases of tree dieback around the world are consistent with climate change (Allen et al. 2010).
- Remote sensing data indicate a 38% decrease of crown area in Bandelier between 2002 and 2006, mainly in piñon-juniper woodland (Garrity et al. 2013).
- Across the western U.S., climate controlled the extent of burned area from 1916 to 2003 (Littell et al. 2009).

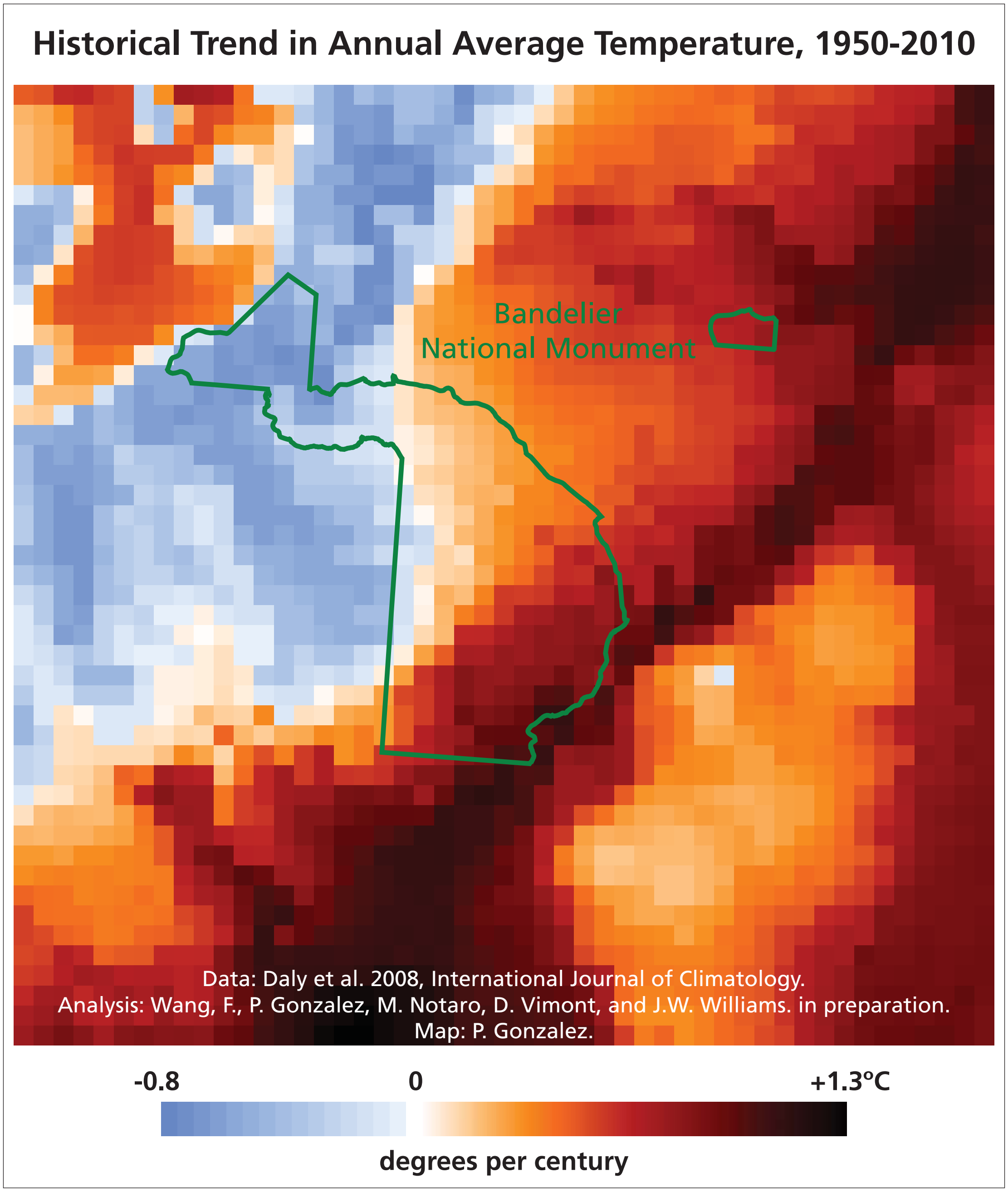
Future Vulnerabilities

- Under continued warming, forest drought stress could continue to cause substantial tree dieback and possible conversion of some forest to grassland (Williams et al. 2013).
- Under high emissions, fire frequencies could increase up to 25% by 2100 (Moritz et al. 2012).
- Past warming has reduced snowpack widely and rainfall in some areas, which may continue to reduce summer streamflow and water supplies (Garfin et al. 2014).
- Bandelier staff and partners are evaluating vulnerabilities of archeological resources to erosion and climate change and possible adaptation measures.

Table. Historical and projected rates of change in annual average temperature (degrees per century) and annual total precipitation (percent per century) (data Daly et al. 2008, IPCC 2013; analysis Wang et al. in preparation). The table only gives central values for the park as a whole. Figures 1, 2, 5, and 6 show spatial variation. Figures 3, 4, and 7 show the uncertainties.

	1950-2010	**2000-2050**	**2000-2100**
Historical			
Temperature	+0.3ºC (0.5ºF.)		
Precipitation	+30%		
Projected (compared to 1971-2000)			
Low emissions (IPCC RCP 4.5)			
Temperature		+2.2ºC (4ºF.)	+2.8ºC (5ºF.)
Precipitation		~0	+1%
High emissions (IPCC RCP 6.0)			
Temperature		+1.8ºC (3ºF.)	+3.3ºC (6ºF.)
Precipitation		-2%	-2%
Highest emissions (IPCC RCP 8.5)			
Temperature		+2.8ºC (5ºF.)	+5.1ºC (9ºF.)
Precipitation		~0	-2%

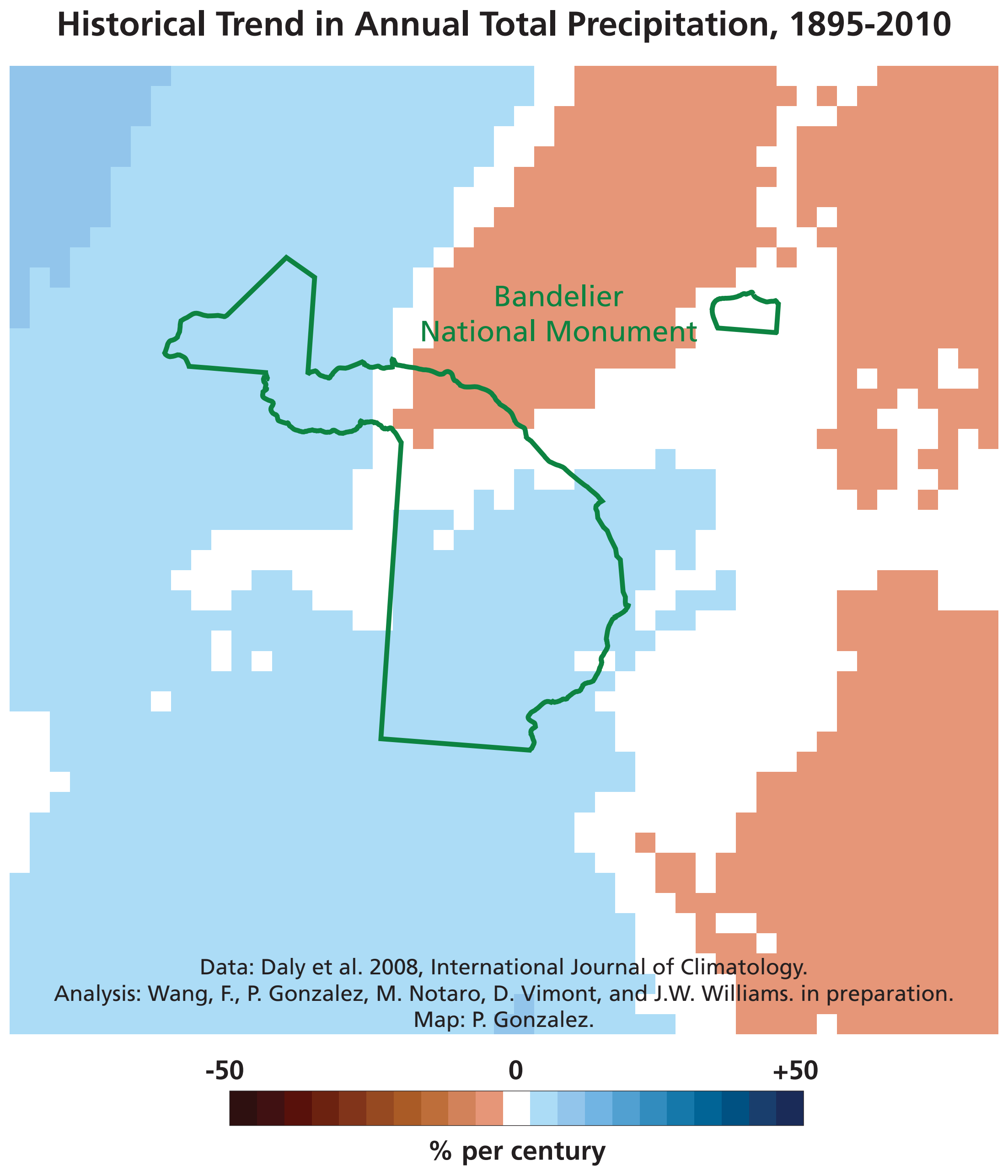
Figure 1.



Historical Trend in Annual Average Temperature, 1950-2010

Data: Daly et al. 2008, International Journal of Climatology.
Analysis: Wang, F., P. Gonzalez, M. Notaro, D. Vimont, and J.W. Williams. in preparation.
Map: P. Gonzalez.

Figure 2.



Data: Daly et al. 2008, International Journal of Climatology.
Analysis: Wang, F., P. Gonzalez, M. Notaro, D. Vimont, and J.W. Williams. in preparation.
Map: P. Gonzalez.

Figure 3. Temperature.
Historical and projected average annual average temperature for the area within park boundaries. For projections, each bar shows one standard deviation above and below the average of up to 33 climate models. (Data: Daly et al. 2008, IPCC 2013. Analysis: Wang et al. in preparation, University of Wisconsin and U.S. National Park Service).

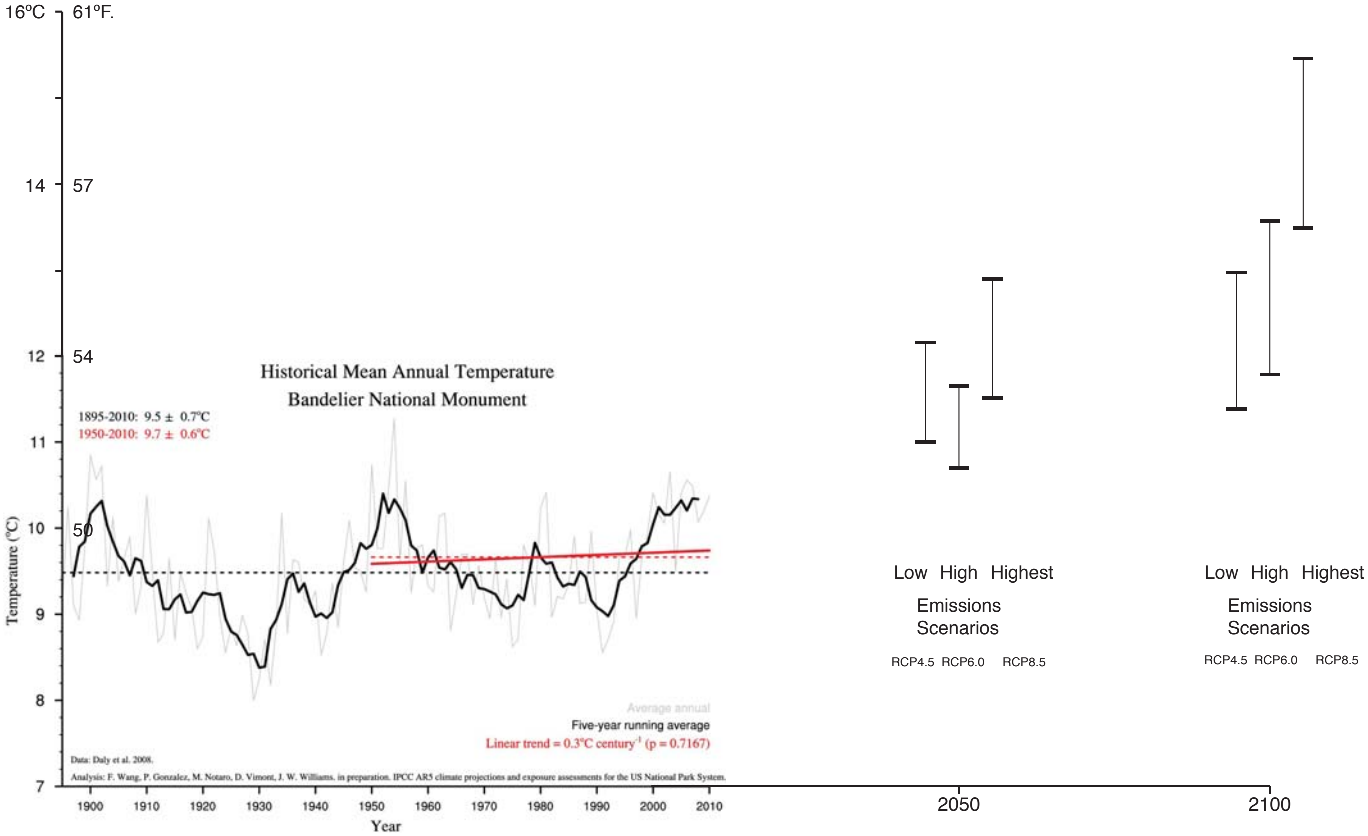


Figure 4. Precipitation.
Historical and projected annual total precipitation for the area within park boundaries.
For projections, each bar shows one standard deviation above and below the average of up to 33 climate models. (Data: Daly et al. 2008, IPCC 2013. Analysis: Wang et al. in preparation, University of Wisconsin and U.S. National Park Service).

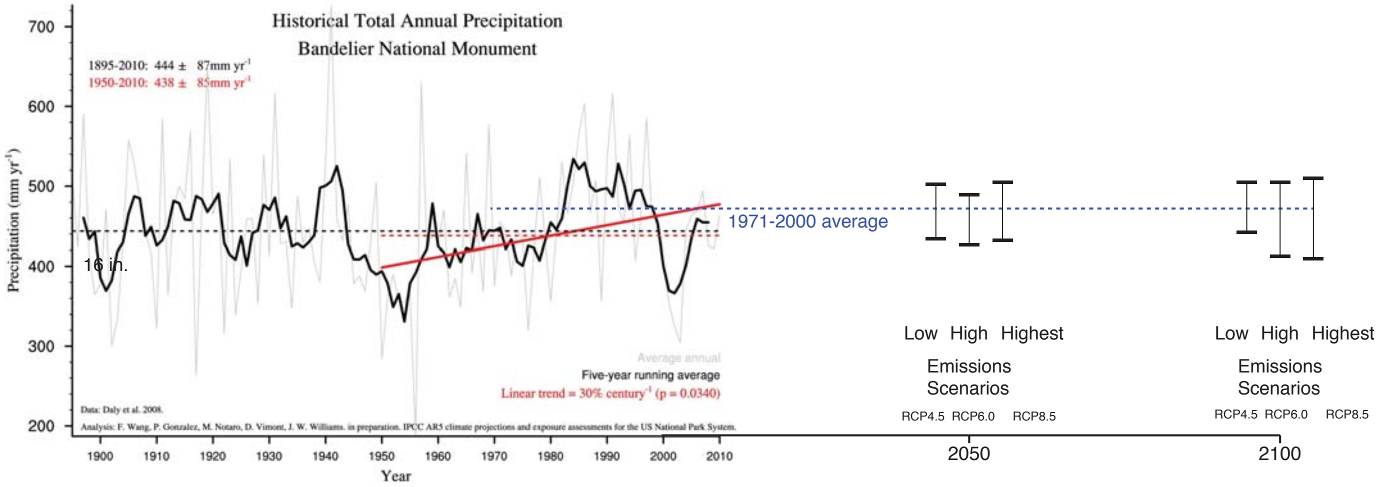
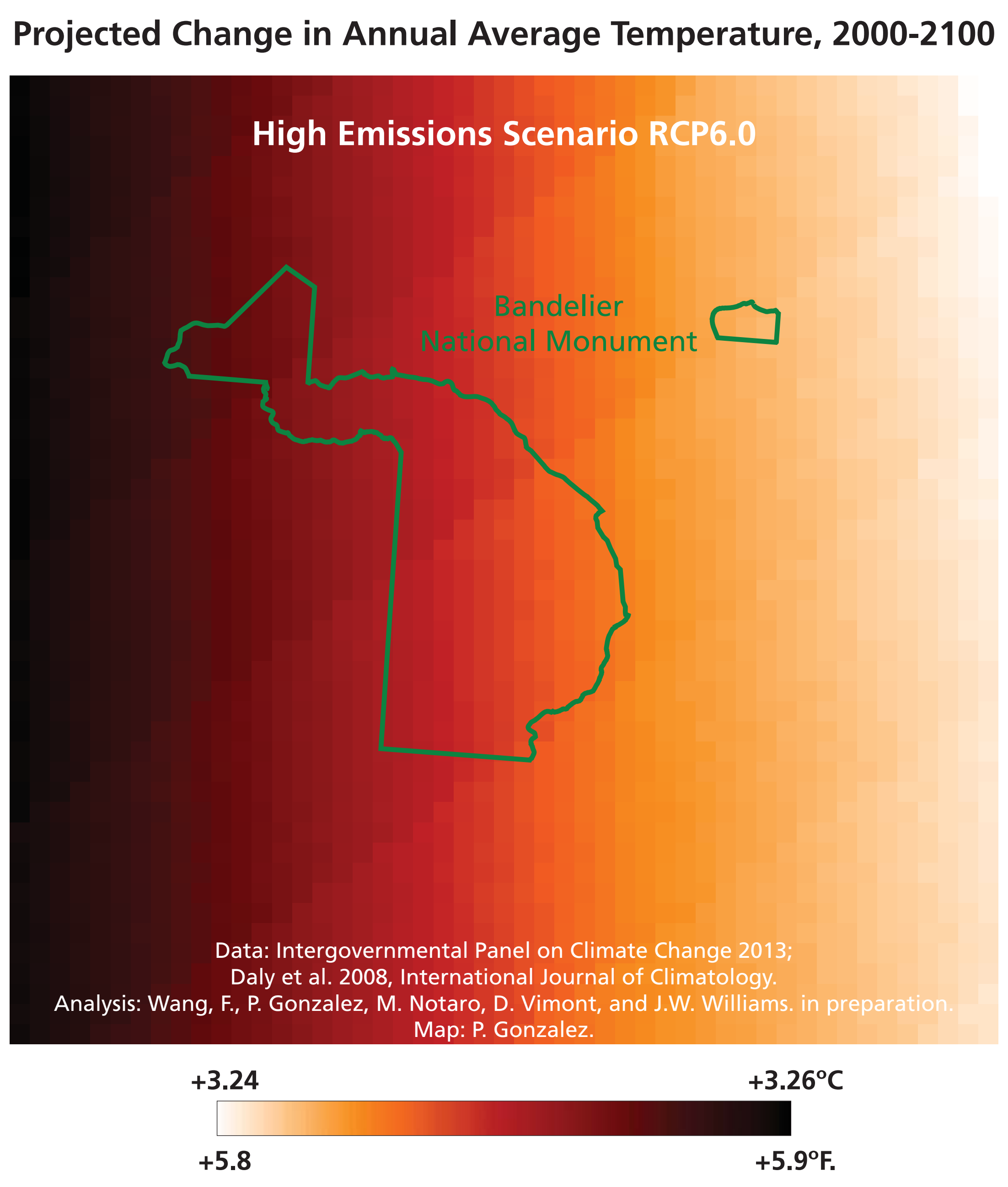
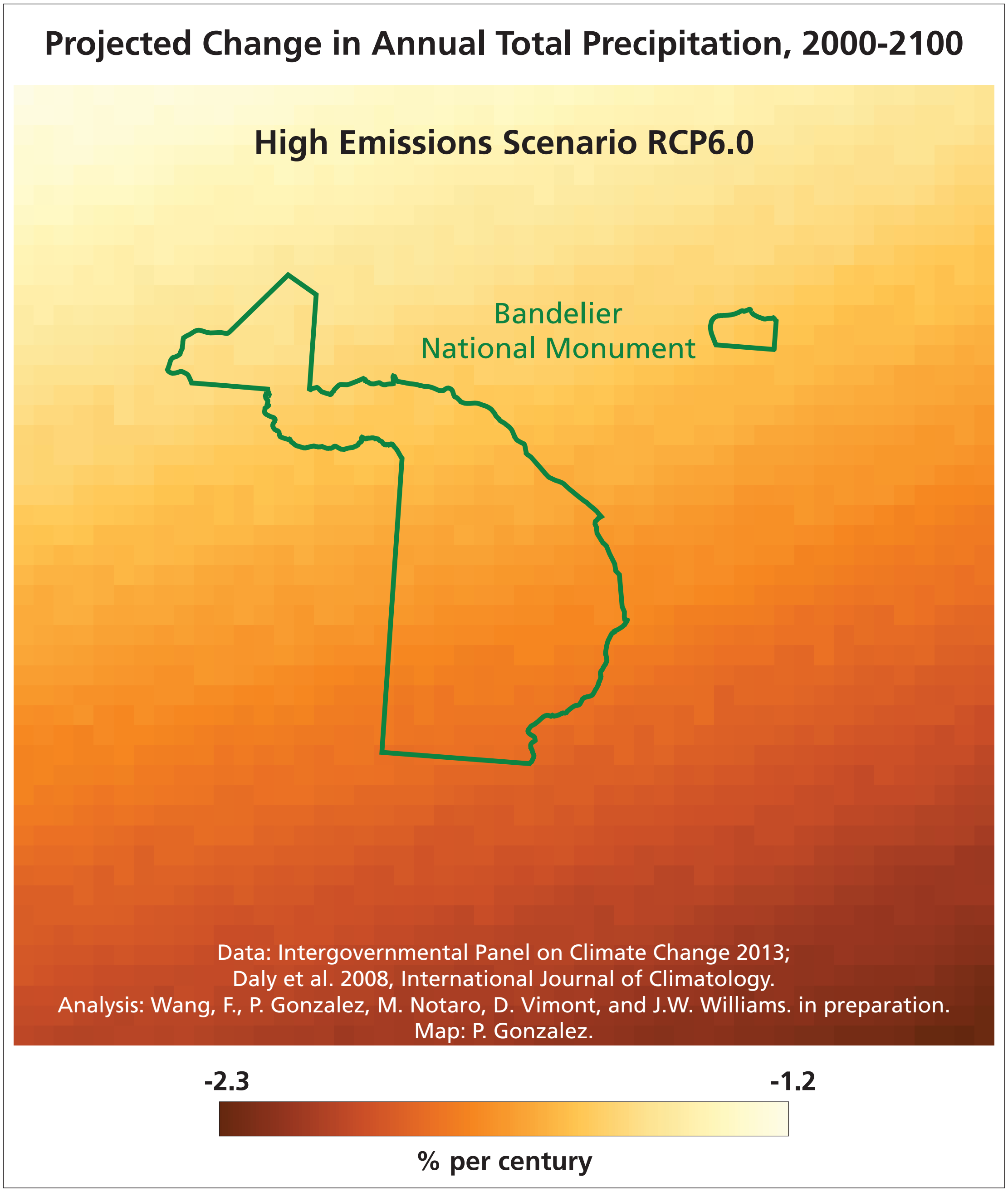


Figure 5.



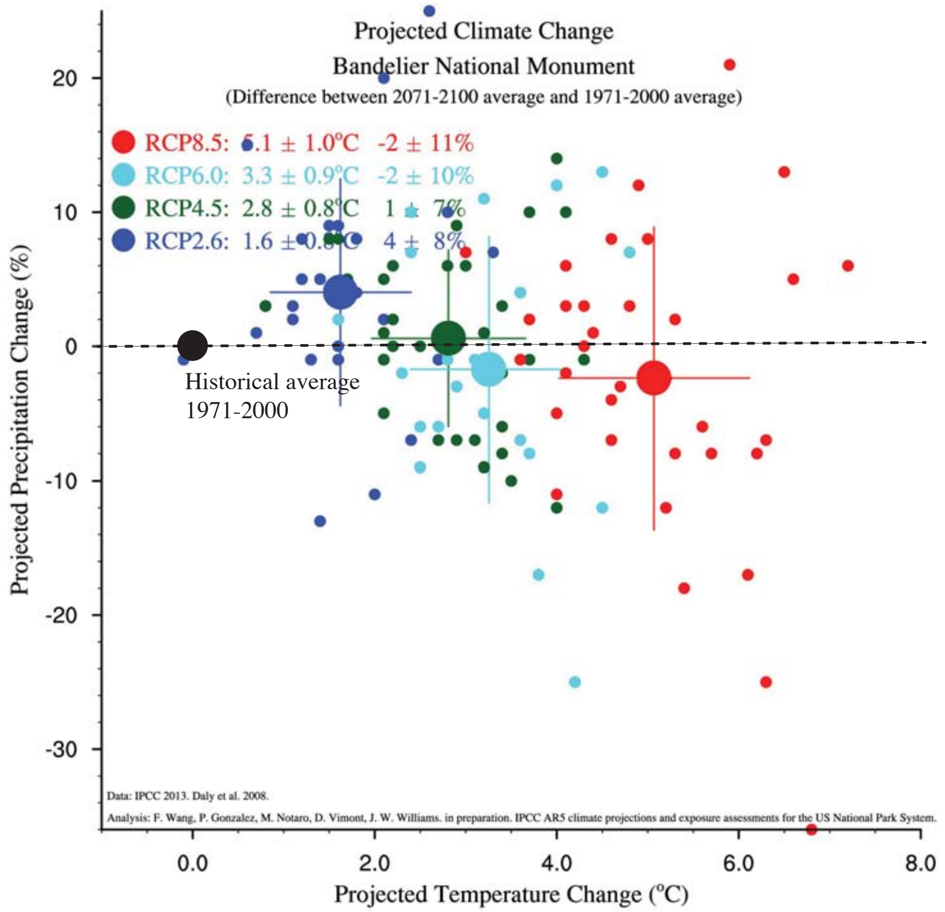
Data: Intergovernmental Panel on Climate Change 2013;
Daly et al. 2008, International Journal of Climatology.
Analysis: Wang, F., P. Gonzalez, M. Notaro, D. Vimont, and J.W. Williams. in preparation.
Map: P. Gonzalez.

Figure 6.



Data: Intergovernmental Panel on Climate Change 2013;
Daly et al. 2008, International Journal of Climatology.
Analysis: Wang, F., P. Gonzalez, M. Notaro, D. Vimont, and J.W. Williams. in preparation.
Map: P. Gonzalez.

Figure 7. Projections of future climate for the area within park boundaries. The large black dot is the current combination of temperature and precipitation. Each small dot is the output of a single climate model. The large color dots are the average values for the four IPCC emissions scenarios. The lines are the standard deviations of each average value. (Data: IPCC 2013, Daly et al. 2008; Analysis: Wang et al. in preparation, University of Wisconsin and U.S. National Park Service).



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Appendix B: Assessment of Vegetation Change in Bandelier National Monument, a "Barometer of Change" in the National Park System

This appendix includes Dr. Matthew Bowker's report to Bandelier National Monument and the Southern Colorado Plateau Inventory & Monitoring Network of the National Park Service. His report formed the basis for vegetation sections of the Bandelier National Monument Natural Resource Condition Assessment.

It also includes, in an appendix to Dr. Bowker's report, a list of data sources used by Dr. Bowker, as well as datasets he generated during his assessment. All data are available in digital format by request to the Southern Colorado Plateau Network, http://science.nature.nps.gov/im/units/scpn/index.cfm.

Assessment of vegetation change in Bandelier National Monument, a “barometer of change” in the National Park system

A summary of 30 years of rapid change, prepared for Bandelier National Monument and the Southern Colorado Plateau Network of the National Park Service Inventory & Monitoring Program, in support of a Natural Resource Conservation Assessment.

Prepared by:

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In consultation with:

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Chapter 1: Analysis of monument-wide trends in multiple ecosystem types

Indicators/Measures:

- ***Change in plant community total areal extent and spatial location through time***
- ***Changes in canopy cover through time***

Background & Importance

Owing to steep elevational and climatic gradients, Bandelier National Monument encompasses vegetation communities ranging from semi-arid woodlands to subalpine forests and meadows (Muldavin et al. 2011). In total, 55 distinct plant communities are mapped in Bandelier with over 800 plant taxa documented (Reif et al, 2009). Although Bandelier was created to protect cultural resources, this impressive diversity of plant community types is significant within the Southern Colorado Plateau Network (SCPN) of the National Park Service's Inventory & Monitoring Program (NPS I&M). The NPS I&M is organized into 32 regional networks. Within each network, the NPS I&M collects standardized data on natural resources (e.g. floras, vegetation maps), and manages a structured monitoring program to inform management decisions. The NPS I&M also synthesizes and distributes these into maps, reports and other information products. Because these are networks of protected areas nested within an ecoregion, they represent a useful frame of reference for assessing the effectiveness of ecoregional biodiversity conservation. The reason for the impressive vegetation diversity in Bandelier is that its elevation gradient ranges from about 1500m to about 3100 m, which is wider than that of Glen Canyon National Recreation Area (~900 – 2300 m), and only slightly narrower than Grand Canyon National Park (~800 – 2500 m). Thus within the SCPN, it is among the most effective units in capturing climatic and vegetation diversity. Bandelier's impressive habitat and floristic diversity is contained within only 13,629ha, which is less than 3% of the area of either Glen Canyon National Recreation Area, or Grand Canyon National Park. Because it encompasses some higher elevations that the majority of NPS units in the area, and because it is positioned at the eastern edge of the SCPN, management of Bandelier as a National Monument represents a valuable contribution to regional biodiversity conservation.

Bandelier National Monument has long been prone to rapid vegetation change. A severe drought occurred in the 1950's in the Southwestern US and northern Mexico, causing widespread tree mortality (Betancourt et al. 1993, Marshall 1957). One of the most dramatic examples of a mass tree mortality event involving ponderosa pine occurred in Bandelier (Allen and Breshears 1998). Within five years, ponderosa pine retreated 2 km upslope, and was subsequently replaced by piñon-juniper woodlands. This example illustrates that in Bandelier, vegetation changes can be simultaneously rapid and occur over a large spatial extent. This susceptibility to change may be related to the steepness of elevation-driven environmental gradients described previously.

While from a land management or conservation perspective this susceptibility is troubling, this change-prone area may be very valuable from a research perspective. It is potentially an opportunity to learn. Climate change ecologists are increasingly attempting to predict likely outcomes of climate change. Gitlin et al (2006), echoing earlier documents prepared by the National Research Council (NRC 1990) and the National Ecological Observatory Network (NEON 2000), argued that highly stressed or otherwise change-prone ecosystems ought to be studied as "barometers of change". Such areas would be ideal locations for long-term monitoring and instrumentation, and would yield a preview of the new community trajectories which emerge under climate change, and provide information on the resilience of various ecosystems. Similar dynamics might be expected to be observed in the future in less stressed or less change-prone ecosystems; thus barometers of change provide a forewarning of the degree and types of changes that could be expected under climate change. Bandelier is rather unique for a monument of its size in that there are multiple decades of data, some of which is from long-term repeatedly measured studies. Bandelier is also located adjacent to a National Laboratory with substantial research infrastructure, and additional observations spanning decades. Thus in many ways, Bandelier has already been serving as a barometer of change. The current document is a first attempt at compiling several of the most information rich datasets available for Bandelier National Monument and surroundings and synthesizing these to assess the condition and trajectory of Bandelier's major ecosystems, and possibly the future trajectory of southwestern woodlands and forests.

Major Drivers of Change in Bandelier National Monument-

At the scale of the entire monument, there are four tightly-linked phenomena responsible for most change that has occurred in recent decades: Climate change, fire, drought/insect outbreak, and past land use, particularly grazing. Other factors strongly affect particular ecosystems, such as elk herbivory or persistent accelerated erosion. These factors are discussed in later chapter.

Before discussing these drivers of change individually, it is worth viewing them as a linked complex of interacting change agents (Fig. 1). 1. A pervasive force, climate change, modifies soil moisture and vapor pressure deficit leading to plant drought mortality (Breshears et al. 2005). 2. Climate change modifies distributions of some bark beetles (Bentz et al. 2010), 3. Warmer and drier conditions favor fire (Williams et al. 2010). 4. Drought stress leads to greater susceptibility of some tree species to bark beetle attack (Wallin et al. 2004). 5. Both drought and beetle-killed plants contribute to fuel load. 6. Greater fuel load creates larger and more intense fires. 7. Accumulated fuel load has been strongly reduced by past grazing, and fire suppression activities (Allen 2004).

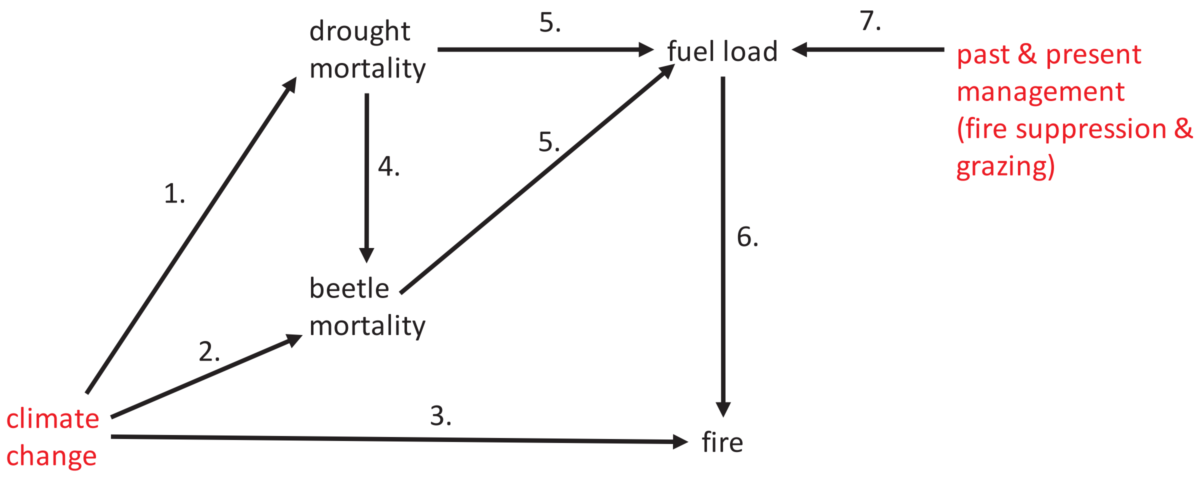


Fig. 1. Linked drivers of vegetation change in Bandelier National Monument and the southwestern US. A variety of direct and indirect effects of forests and woodlands are ultimately exerted by climate change and past & present management. See text for explanation.

Climate change: Climate change is both a direct and indirect influence on vegetation change. Recently, Williams et al. (2012) reported that a forest stress index, based on tree ring widths was about equally and strongly ($R^2 = 0.82$) predicted by warm season vapor pressure deficit (driven by temperature) and cool season precipitation. These two parameters are likely to change in the near future. All available models converge on a scenario where northern New Mexico will experience warming (Karl et al. 2009). Precipitation projections are much less certain and likely to vary among models, although reduced spring precipitation is predicted by a consensus model (Karl et al. 2009). Both of these changes can be expected to influence soil moisture and therefore drought mortality directly, and indirectly affect fire susceptibility (see below). Williams et al. (2010, 2012) project that by 2050, forest drought stress values will routinely be equal to or more extreme than megadroughts seen in the 1200's and the 1500's. This suggests that area currently occupied by Southwestern forests will to a large degree no longer be forest habitat. Breshears et al. (2011) argues that such climate change impacts on vegetation communities may not be gradual, incremental and homogenous (e.g. a small mortality rate every year). Rather climate change impacts on ecosystems may be "big, fast, and patchy". Climate changed linked drought and fire have already caused high mortality impacts in 18% of southwestern forests (Williams et al. 2010). Such impacts give managers and stakeholders few options and little time to respond to climate change driven vegetation change such as the mass tree mortality events now being observed globally (Allen et al. 2010).

Fire suppression and subsequent large scale crown fire: Fire is a part of the evolutionary environment of most of the woodland, savanna and forest types within Bandelier National Monument (Moore et al. 1999, Touchan et al. 1996). It is well demonstrated that two phenomena: livestock reduction of fine fuels, and fire suppression coincide with a shift to less frequent fire cycles (Touchan et al. 1996). We consider fire here as a stressor and a driver of vegetation change because the scale of recent fires has been unusual (Fig. 2). The 1977 La Mesa Fire burned 6354 ha in 1977. The 1996 Dome fire burned 6677 ha, and was followed shortly by the 1997 Lummis Fire which burned another 865 ha.. The Cerro Grande fire of 2000 burned 16909 ha. Most recently, the Las Conchas fire burned 63250 ha in 2011. These fires are

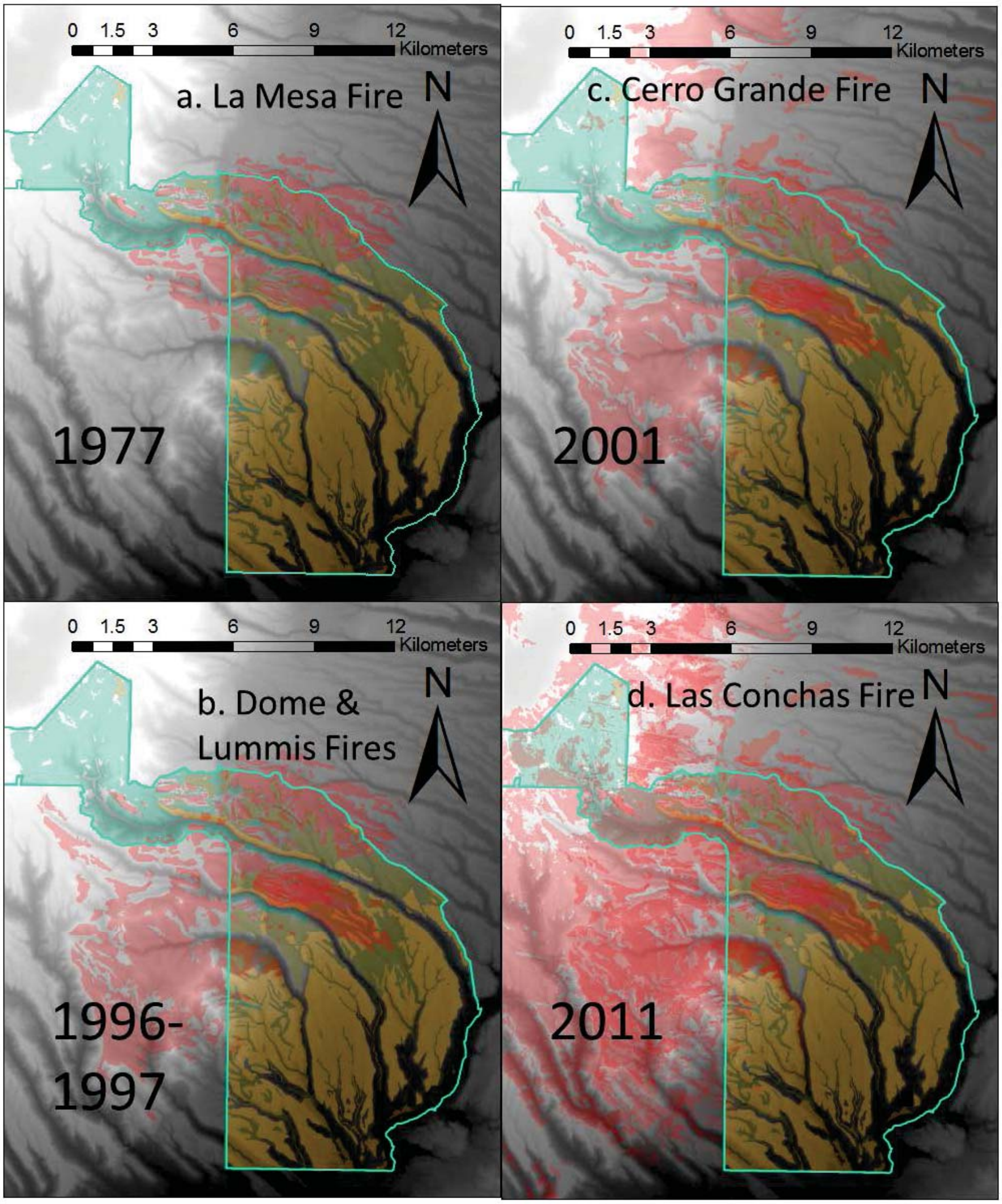


Figure 2. Cumulative footprint of medium and high severity wildfires in the 1981 distributions of the mixed conifer-aspen forest zone (blue green), ponderosa pine forest zone (drab green), and piñon-juniper woodland zone (orange), since 1977. Light red shades indicate a high or medium severity burn. Darker red shades indicate areas where high or medium severity burns have occurred twice. Burn severity data is unavailable for the Lummis fire, so the entire burned area is illustrated.

occurring at higher frequency and intensity and at increasing spatial scale. Fire intensities varied within these fires but all of these events involved significant stand replacement. These large fires are likely a result of over a century of fire suppression and fuel build-up due to fire suppression, and are also facilitated by increasing temperatures and decreased precipitation.

Drought & Insect Outbreaks: Periodically, severe droughts may be associated with pulse mortality of overstory trees (Breshears et al. 2005, 2011). One such drought occurred in 2002. It caused major mortality to *Pinus ponderosa* and *P. edulis*, especially in lower elevations but also impacted *Pseudotsuga menziesii* and *Abies concolor* at higher elevations (Muldavin et al. 2011). Primarily mature trees were affected rather than saplings. This extreme drought was not the driest on record, but was the most impactful in centuries. The severity of die-back events has been linked to the fact that warming climate interacts with drought to induce mass mortality. This was hypothesized by Breshears et al. (2005), and experimentally tested by Adams et al. (2009) using *Pinus edulis*. Using 50 years of tree ring data from multiple species along an elevation gradient in Arizona, Adams and Kolb (2005) found that growth rates were related to both water availability and temperature, and that species in the hotter and drier parts of their range were more sensitive to these influences (i.e. their growth tracked climate more closely than conspecifics from higher sites). Insects such as bark beetles and the fungi they disperse lead to mortality of water stressed trees because water stress can inhibit defense mechanisms such as production of resins (Wallin et al. 2004, Negron et al. 2009). Our data cannot distinguish between temporally congruent drought mortality episodes and bark beetle outbreaks; thus we treat them as a tightly-linked stressor complex.

Reference Conditions

Bandelier National Monument can be roughly divided into three major zones based upon dominant plant communities. From most mesic to most xeric, they are: mixed conifer-aspen forests, ponderosa pine forests, and piñon-juniper woodlands.

Historical reference conditions cannot be known with certainty, but can be partially reconstructed from various sources of information including paleoecology, fire history reconstructions, and historical records (Allen & Breshears 1998, Williams et al. 2012). An overview of human land use history, and its interactions with natural perturbations is useful to understand vegetation changes in recent centuries, and determine the best available period of time to use as a reference state. Bandelier was densely populated from 1150-1550 by Ancestral Puebloans, and this occupation likely modified forest structure due to deforestation, herbivore activity due to hunting and fire history due to human ignition (Sydoriak et al. 2000, Allen 2004). The termination of the occupation coincides with a severe period of drought, which may have led to changes in vegetation structure. The three centuries after abandonment of the Pueblos were likely a time of recovery of forest and woodland cover that may have been previously cleared. The Navajo adopted livestock in the 1600's, which may have impacted the Bandelier uplands to some degree, though this is not known with certainty. Livestock numbers during this time are poorly documented, but most likely low (Allen 2004). As Euro-Americans began to settle New Mexico, livestock numbers increased in general. The presence of nomadic Navajo, Apache and Ute in the Jemez Mountain region was a deterrent to settling by Euro-Americans until these peoples were confined to reservations in the 1860's, thus keeping livestock numbers low until this time (Allen 2004). Subsequent Euro-American settlement and the establishments of large land grants brought about the introduction of large numbers of livestock in the 1880's which are thought to have reduced fine fuel loads and fire frequency until 1932 when the original National Monument was created (Sydoriak et al. 2000, Allen 2004). Active fire suppression followed in the early 1900's (Sydoriak et al. 2000). Because of this long series of perturbations, the period around the 1860's and 1870's is likely the best period of time on which to base a historical frame of reference and is commonly used in the Jemez Mountains region. It is not known if the current anomalously warm climate regimes can support vegetation with the characteristics of communities from this time period. Nevertheless, there are multiple sources of information to draw from that enable researchers to at least know what some of the major ecosystems were like in the late 19th century (Moore et al. 1999). Historical reference conditions are described to the extent possible for individual ecosystems in later chapters.

Staff at Bandelier National Monument have developed a set of *desired future conditions* (Bandelier National Monument 2006). These desired conditions largely are consistent with an interpretation of conditions in the past based on available data and expert opinion, and are specific enough to represent an operational reference condition with which to compare current condition and establish trend. ***Desired conditions are developed more extensively and specifically for each major ecosystem type in later chapters.*** At the scale of the entire monument we can make four broad generalizations about desired conditions.

First, most of the ecosystems have a tree canopy component. Although shrubs and grasses are ecosystem components, it is not thought that large shrublands and grasslands without trees were abundant on the landscape.

Second, in many Bandelier National Monument ecosystems it is thought that tree canopy cover increased through the 20th century, and under a more natural disturbance regime forests and woodlands would be more open. Nearly a century of fire suppression has allowed many forests and woodland stands to attain unusually dense canopies, sometimes at the expense of understory herbaceous biomass. Desired canopy cover can vary on a site-by-site basis. For example, in the piñon-juniper zone, some sites are recently thickened former savannas, in which the desired canopy density is less than 15%. On other, rocky sites in tree-dominated piñon-juniper woodlands, desired canopy cover can exceed 30%. Currently, a typical piñon-juniper woodland exhibits about 15-25% canopy cover, with local areas approaching 50%. Ponderosa pine ecosystems can also vary in their density, but one major desired cover type is the Ponderosa pine savannah with < 5% canopy cover of mature trees. Finally, an open canopy structure would be desired for most mixed confer forests, except for those on north slopes or other mesic conditions.

Third, although fine scale shifting mosaics of different forest and woodland types would be expected based upon normal disturbance and succession cycles, there should be no major directional shifts in the distribution of major vegetation types. For example, large shifts in the lower elevation boundary of a vegetation type, like that observed in the 1950's with ponderosa pine (Allen & Breshears 1998), would be a deviation from desired conditions.

Fourth, there is also the perspective that the landscape has become homogenized, masking small-scale heterogeneity in soil, topographic, and microclimatic gradients which previously created and maintained fine scaled vegetation mosaics (B. Jacobs, pers. comm.).

In summary, a desired trajectory for vegetation *at the scale of the entire monument* would be an overall decrease in tree canopy cover **without** loss of a tree component from the various ecosystems and **without** a directional shift in the broad distribution patterns of the major ecosystem types. These changes would result in greater fine scale heterogeneity.

Data & Methods

Vegetation Maps

Two vegetation maps have been produced in recent decades encompassing most or all of Bandelier National Monument. Allen (1989) produced a map based upon aerial imagery from 1981. Map units were drawn based upon stereographic images at a resolution of 1.5 ha. This map encompassed the large majority of Bandelier National Monument and its surroundings, with the exception of the extreme northwestern portion. Muldavin et al. (2011) produced a map using supervised automated digital processing, assisted by multispectral imaging techniques at a resolution of 0.5 ha. This map was confined to the entire extent of Bandelier National Monument, and was based upon 2004 imagery. To facilitate comparisons among the two maps, both were reduced to their intersection.

The two maps used a different system of map units. Allen (1989) developed a system similar to that used by Potter & Foxx (1981) and USFS (1987), and provides a dichotomous key for its application. It consists of 43 forest and woodland types, and 23 additional patch types. Most of these types identify the dominant tree component (e.g. Mixed conifer, Ponderosa Pine, Piñon-Juniper), and contain further designations based upon other community components (e.g., whether a shrubby understory is present). However 31 of these were mapped only in the areas outside of the Monument, and were not considered further (with one exception, see below). Muldavin e al. (2011) collected a series of ground plot surveys which were analyzed using multivariate statistical methods and classified into a system consistent with the guidelines of the National Vegetation Classification Standard (http://www.fgdc.gov/standards/projects/FGDC-standards-projects/vegetation). A total of 95 plant associations were identified in the Monument. A map unit is a map able complex of one or (usually) more associations. There are 55 Level 2 map units, nested within 24 Level 1 units.

Since a map can be simplified but not made more complex without new data, we focused on reclassifying the more detailed Level 2 map units of the Muldavin et al. (2011) map to the system used by Allen (1989) to enable a direct comparison of the two. Each map unit was ascribed to the *best matching category* in the Allen system. We relied on the description of the associations within each map unit provided by Muldavin et al. (2011). To a lesser extent, we also considered spatial congruence; for example, if two categories are roughly equivalent they would be expected to overlap spatially, at least to some degree. Finally, consultations were made with local experts familiar with both maps (Drs. Craig Allen and Brian Jacobs), to assist with some reclassification decisions. Although this translation was imperfect, and some reclassifications retain some uncertainty, most major Muldavin et al. (2011) map units corresponded reasonably well to major Allen (1989) map units. Most challenging were matching Allen's CWCX (sparsely vegetated canyon wall complex), and various riparian vegetation categories. We relied on spatial congruence more in the case of CWCX; this corresponded spatially to several woodland and grassland map units in Muldavin et al. (2011), all of which invoke terms such as "sparse", "canyon", "rockland", "talus", and "dry wash". In the case of riparian vegetation categories, we primarily relied upon the descriptions provided of the vegetation classes, and assigned them to the proper Allen (1989) map units. In one case we invoked an Allen (1989) unit, riparian-deciduous (RI-D), even though Allen had not previously mapped it within the Monument. Allen (1989) maps Ponderosa Pine (PP) and ecotonal Ponderosa Pine-Pñion-Juniper (PPPJ). Both were clearly best associated with the same Muldavin et al. (2011) map unit, meaning that we could unfortunately not separate PP and PPPJ for our analyses. A few Allen (1989) units had no clear correspondence to the Muldavin et al. (2011) units, and they simply were not used in the reclassification of Muldavin et al. (2011). Due to the die-off of mature *Pinus edulis* in the 2002-

2003 drought-beetle outbreak, several units used by Allen (1989) no longer exist because mature individuals of this species no longer exist as a co-dominant. We retained the convention of Muldavin et al. (2011) and kept separate *Juniperus* woodlands from recent *Pinus*-*Juniperus* woodlands; the latter were reclassified as (P)J. Muldavin et al. (2011) maps a few units of ruderal post-fire vegetation that were simply not mapped by Allen (1989) because they did not exist in the earlier imagery. The full synthetic reclassification system is tabulated below (Table 1).

Analysis

In order to make simpler comparisons, we also created two schemes with fewer categories to visualize the vegetation changes. The first scheme focused only on categories with a major areal coverage or a strong management interest, lumping categories when there were difficulties in separating them in both maps. This scheme compared PJ, PJ-S, (P)J, (P)J-S, GRAS, GRSH, SH-O, SHRU, PMCS, PPMC, RH, and TA. Vegetation types J, J-SAV, and J-S were grouped together as *Juniperus*-dominated communities because the two maps did not make the same divisions among categories (see notes in Table 1). PP & PPPJ were similarly grouped for essentially the same reason. MC-S was grouped with MC because it was a unit of minor spatial coverage. These schemes enable a fairly detailed view of over two decades of vegetation change in all major ecosystem types.

Due to a perceived increase in the predominance of shrubs in many ecosystems noted by several local scientists in multiple ecosystem types, we also grouped all of the map units which specifically denote a shrub component (J-S, PJ-S, SH-O, SHRU, PP-S, PPJS, PMCS, MC-S) and compared change in the areal coverage of these over the period 1981 – 2004.

Table 1. Best-match equivalency and synthesis of Muldavin et al. (2011) and Allen (1998) map units. Notes regarding classification decisions appear in italics.

	Muldavin et al. (2011) map units	**Allen (1990) map units**	**Synthesis**
1	(Piñon) Alligator Juniper/Wavyleaf Oak Woodland		(P)J-S
2	(Piñon) Oneseed Juniper Ravine Woodland		(P)J
3	(Piñon) Oneseed Juniper/Grama Grass Mesa Woodland		(P)J
4	(Piñon) Oneseed Juniper/Rockland Canyon Woodland	CWCX	CWCX
5	(Piñon) Oneseed Juniper/Treatment Woodland	*no equivalent*	J-T
6	(Piñon) Oneseed Juniper/Wavyleaf Oak-Grama Grass Canyon Woodland		(P)J-S
7	(Piñon) Oneseed Juniper/Wavyleaf Oak Mesa Woodland		(P)J-S
8	Apache Plume-New Mexico Locust Mesa Shrubland	SHRU	SHRU
9	Apache Plume-New Mexico Privet Dry Wash Riparian Shrubland		SHRU
10	Archaeological Site	ARCH	ARCH
11	Arctic Rush-Kentucky Bluegrass Montane Wetland/Wet Meadow	MEAD	MEAD

12	Blue Grama-Black Grama Mesa Grassland	GRAS	GRAS
13	Box Elder-Alder-Narrowleaf Cottonwood Montane Riparian Forest	RI-D	RI-D
14	Engelmann Spruce/Sprucefir Fleabane Mesic-Wet Forest	MC-S	MC-S
15	Fendler Ceanothus-New Mexico Locust Montane Mesa Shrubland	GRSH	GRSH
16	Gambel Oak/New Mexico Locust Canyon Shrubland		SH-O
17	Gambel Oak/New Mexico Locust-Mountain Muhly Mesa Shrubland	SH-O	SH-O
18	Grama Grass/Rockland Grassland		CWCX
19	Grama Grass/Talus Grassland		CWCX
20	Lower Montane Valley Bottom Grassland		GRAS
21	Lowland Emergent Wetland		DROW
22	Mountain Muhly-Little Bluestem Lower Montane Grassland		GRAS
23	Oneseed Juniper Dry Wash Woodland		CWCX
24	Oneseed Juniper/Black Grama Canyon Woodland		CWCX
25	Oneseed Juniper/Black Grama-Blue Grama Mesa Woodland	J	J
26	Oneseed Juniper/Blue Grama Mesa Woodland		J
27	Oneseed Juniper/Grama Grass Valley Bottom Woodland		J

28	Oneseed Juniper/Rockland Canyon Woodland		CWCX
29	Oneseed Juniper/Sparse Canyon Woodland		CWCX
30	Oneseed Juniper/Wavyleaf Oak/Sparse Rockland Mesa Woodland	J-S	J-S
31	Oneseed Juniper/Wavyleaf Oak-Grama Grass Canyon Woodland		J-S
32	Open Water	STRE	STRE
33	Other Built-up Land	POND	POND
34	Ponderosa Pine- Broadleaf Mixed Montane Riparian Forest	RI-M	RI-M
35	Ponderosa Pine Canyon Woodland	PP-S	PP-S
36	Ponderosa Pine Dry Wash Woodland		PP-S
37	Ponderosa Pine Ravine Woodland		RI-M
38	Ponderosa Pine/Mountain Muhly-Blue Grama Mesa Woodland	PP	PP
39	Ponderosa Pine/Thurber's Fescue Upper Montane Woodland	PPMC	PPMC
40	Ponderosa Pine/Wavyleaf Oak Mesa Woodland		PP-S
41	Ponderosa Pine-Gambel Oak/Deer Sedge Woodland		PP-S
42	Post-fire Montane Ruderal Herbaceous Vegetation (Cerro Grande)	*no equivalent*	RH
43	Post-fire Montane Ruderal Herbaceous Vegetation (Dome Fire)		RH

44	Post-fire Valley Ruderal Herbaceous Vegetation		RH
45	Public Building/Residential	DEV	DEV
46	Quaking Aspen/Fendler's Meadowrue Mesic-Wet Forest	TA	TA
47	Quaking Aspen/Thurber's Fescue Upper Montane Woodland		TA
48	Recreation Site	COMM	COMM
49	Reservoir Flood Zone Ruderal Herbaceous Vegetation	LAKE	LAKE
50	Road		COMM
51	Rockland/Cliff	ROCK	ROCK
52	Ruderal Herbaceous Vegetation	*no equivalent*	RH
53	Sand Sagebrush/Talus Canyon Shrubland	CWCX	CWCX
54	Sandbar Willow Riparian Shrubland	SAND	SHRU
55	Talus-Rubble Land	TALU	TALU
56	Tamarisk-Sandbar Willow Riparian Shrubland	DROW	DROW
57	Thurber's Fescue-Parry's Danthonia Upper Montane Grassland	MG	MG
58	Wavyleaf Oak/Ruderal Foothill Shrubland	SH-O	SH-O
59	Wavyleaf Oak/Sideoats Grama Mesa Shrubland		GRSH

60	Wavyleaf Oak-Mountain Mahogany/Rockland Canyon Shrubland		SH-O
61	Western Wheatgrass-Ruderal Herbs Mesa Grassland		GRAS
62	White Fir-Box Elder-Thinleaf Alder Mixed Montane Riparian Forest	RI-M	RI-M
63	White Fir-Douglas-fir/Common Juniper-Gambel Oak/Deer Sedge dry-Mesic Forest	MC	MC
64	White Fir-Douglas-fir/fendler's Meadowrue Mesic-Wet Forest		MC
65	White Fir-Douglas-fir/Rocky Mountain Maple Canyon Forest		MC
66	White Fir-Douglas-fir/Thurber's Fescue Upper Montane Woodland Savanna		MC
67	White Fir-Douglas-fir-Aspen/Common Juniper/Deer Sedge Dry-Mesic Forest		MC
68	White Fir-Douglas-fir-Aspen/Fendler's Meadowrue Mesic-Wet Forest		MC
69	White Fir-Douglas-fir-Aspen/Rocky Mountain Maple Canyon Forest		MC
70	White Fir-Douglas-fir-Aspen/Thurber's Fescue Upper Montane Woodland		PPMC
71	White Fir-Douglas-fir-Gambel Oak Canyon Forest		PMCS
72	*no equivalent*	FELS	-
73	*no equivalent; best match is a better match for a different category*	JSAV	CWCX
74	*does not exist anymore*	PJ	-

75	*does not exist anymore*	PJ-S	-
76	*no equivalent; best match is a better match for a different category*	PPJS	-
77	*PPPJ & PP cannot be separated in current map; both correspond to Ponderosa Pine/Mountain Muhly-Blue Grama Mesa Woodland*	PPPJ	PP
78	*current map shows a clear codominance by deciduous spp. ; better match for RI-M*	REMC	-
79	*current map shows a clear codominance by deciduous spp. ; better match for RI-M*	REPP	-

Canopy Cover Point Data

Associated with the Allen (1989) vegetation map, 969 points were sampled on the ground across the Frijoles watershed, so that there was approximately one point per 6.7 ha. Sampling was conducted before GPS technology was widely available, so locations were selected on a paper map. Consequently spatial coordinates have some degree of error associated with them. At each point absolute percentage canopy cover was visually estimated.

Associated with the Muldavin et al. (2011) vegetation map, there was a series of ground measurements of canopy cover taken in between 2003 – 2008. There were multiple types of points sampled: vegetation classification standard plots, vegetation classification quick plots, and accuracy assessment plots. Each of these had a different purpose in the vegetation mapping process, but most importantly for our usage were all spatially explicit (i.e. spatial coordinates were documented), and all contained data on canopy cover usually recorded as the midpoint of a modified Domin-Krajina scale (Mueller-Dumbois & Ellenberg 1974), or in some cases actual percent canopy cover. There were 2305 total points many of which fell within the Frijoles watershed, but they were not confined to the same area as the Allen (1989) data. These data spanned a period of time where many overstory individuals, especially *Pinus edulis* and *Pinus ponderosa* were dying due to drought and bark beetle attack. The lead author, Esteban Muldavin, supplied us with unpublished data which separated cover of dead or dying individuals. We subtracted dead and dying individuals from the total canopy cover prior to using the data.

Analysis

We used these two point datasets to estimate where and to what degree canopy cover has changed since the very wet 1980's to the post-2002 drought (and post-Dome, Lummis, and Cerro

Grande fires) time period. We used a 50 m buffer around the Allen (1989) point data to define the extent of our analysis, which was roughly the Frijoles watershed. 1980's canopy cover was estimated in three ways based upon the Allen (1989) data. First and most simply, the Allen (1989) vegetation map contained an attribute estimating canopy cover at the spatial scale of polygons. We used this data without altering it. These cover values were estimated from 1981 visual imagery at the same time as vegetation communities were mapped. The disadvantage of these data is their coarse spatial scale. The advantage of this method is that the sharp boundaries that occur in nature, for example in transitions among vegetation types or from mesa top to canyon can be easily seen. This dataset was referred to as 1980's A. Second the 1987 point data of Allen (1989) was used to create a surface using the inverse distance weighting interpolation technique, applied with a power of 2 (Franke1982). We also attempted this operation using kriging (recommended for some applications by Zimmerman et al. 1999), but selected inverse distance weighting because it conserves the values of the point dataset and creates a less extreme smoothing of the surface than kriging, which we believed was more appropriate in a highly dissected and heterogeneous landscape. The inverse distance weighting procedure also more accurately depicted important features such as canopy openings created by the La Mesa fire, and higher vegetation coverage in Frijoles canyon. These interpolations capture finer-scale variation in canopy cover that cannot be observed when viewing the polygon-scale data (Allen 1989). The disadvantage of interpolations methods in general is that the resultant product tends to smooth transitions that are sharp in nature, such as boundaries between vegetation communities. This dataset was referred to as 1980's B. Finally, our third method for estimating 1980's canopy cover was simply the average of the previous two methods. The averaging method conserves the sharp transitions that are well captured by polygon data, but allow some degree of variation within polygons. This dataset was referred to as 1980's C.

To estimate post-2002 canopy cover we applied the inverse distance weighting procedure to the point data associated with Muldavin et al. (2011). Although there were a large number of points, these data were less regularly spaced, and contained fewer points in the canyon bottom. There were no polygon data available at this time point, thus only one version of post-2002 cover was created. This dataset was simply referred to as post-2002.

To estimate change in canopy cover we used map algebra to subtract the values of post-2003 from 1980's A, 1980's B, and 1980's C. Thus, three different versions of canopy cover change were created (Diff A, Diff B, Diff C). They are scaled such that positive values indicate canopy loss, and negative values indicate canopy gain.

Condition & Trend

Vegetation in Bandelier National Monument has changed drastically in 23 years between the 1981 and 2004 imagery, and appears to be reorganizing rapidly. We must acknowledge that there are inherent difficulties making comparisons among two maps which were produced by different researchers using different techniques. This undoubtedly creates error, and some apparent differences may simply be errors. However, it seems exceedingly unlikely that some of the broadest and strongest patterns can be explained in this way.

At the lowest elevations of the Monument, it appears that woodlands and savannas with *Juniperus* as the canopy dominant are expanding. This pattern appears independent of any change to other species such as *Pinus edulis*. Further upslope, the most striking difference in vegetative dominance to have occurred is the complete loss of mature piñon pine as a co-dominant species due to drought and beetle mortality. This change occurred in association with the 2002 drought, and Figure 3 illustrates the spatial extent of this change. These former *P. edulis* dominated or co-dominated stands have shifted to Juniper woodlands ***via* subtraction of *P.edulis* rather than migration of *Juniperus***. Although *P.edulis* is retained mostly as saplings, it can no longer be said to be a co-dominant. It may retain the potential to co-dominate again in the future.

Further upslope, there is a clear shrinkage and fragmentation of stands with *Pinus ponderosa* as the dominant, or co-dominant with *P. edulis* (Figure 3, 4). Grass-dominated stands expanded at the apparent expense of *P. ponderosa* coverage. Another factor is that many *P. ponderosa* stands seem to be gaining a shrubby understory. Across the Monument, but particularly in the middle elevations where Ponderosa Pine was dominant, shrubs are becoming more prevalent either as dominants, co-dominants, or prominent features of understories. The coverage of ecosystems wherein shrubs are a major component has approximately tripled (Figure 5). It is difficult to ascribe such a large pattern to differences in vegetation classification or mapping. Allen (1989) was very cognizant of the distinctiveness of communities with and without shrub components in the 1981 imagery, devoting several map units to making this distinction. Compared to Ponderosa pine, growth ring widths of the shrub *Quercus gambellii* are less sensitive to interannual changes in drought severity (Adams and Kolb 2005), providing one possible reason for shrub proliferation in the middle elevations of the monument. Another major reason might pertain to the ability of *Quercus gambellii* to resprout after fire (Abella 2008, Strom & Fulé 2007). Some portions of the monument have experienced repeated fires in the last few decades.

Figure 3. Change in spatial distribution of major vegetation map units from 1981 – 2004. a.1981, modified from Allen (1989). b. 2004, modified from Muldavin et al. (2011).

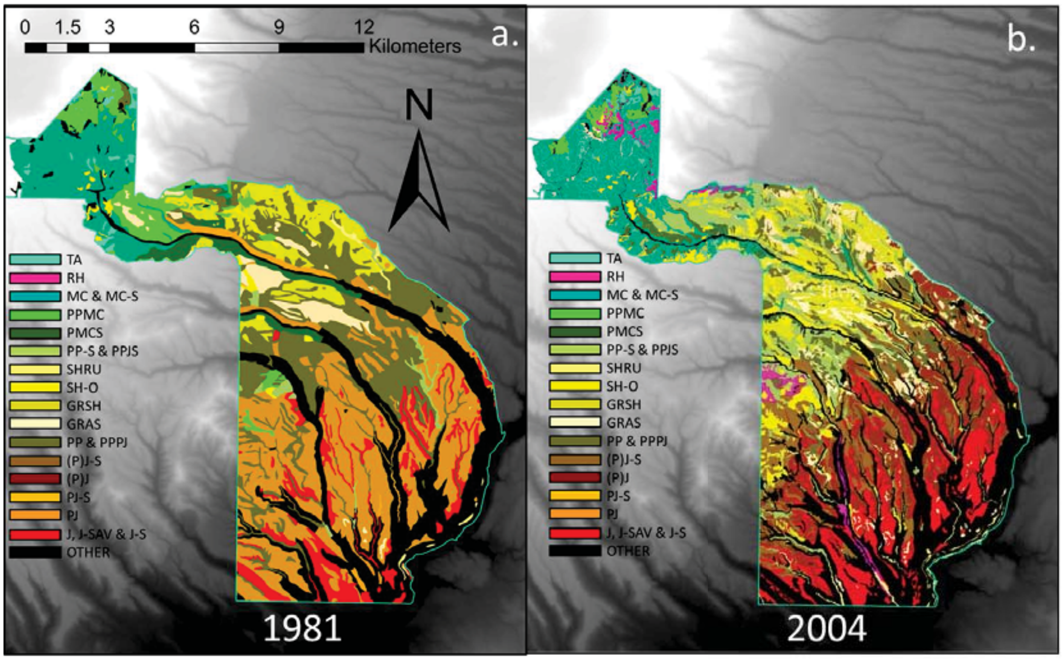


Figure 4. Change in total areal coverage of major vegetation map units from 1981 (white bars) – 2004 (black bars).

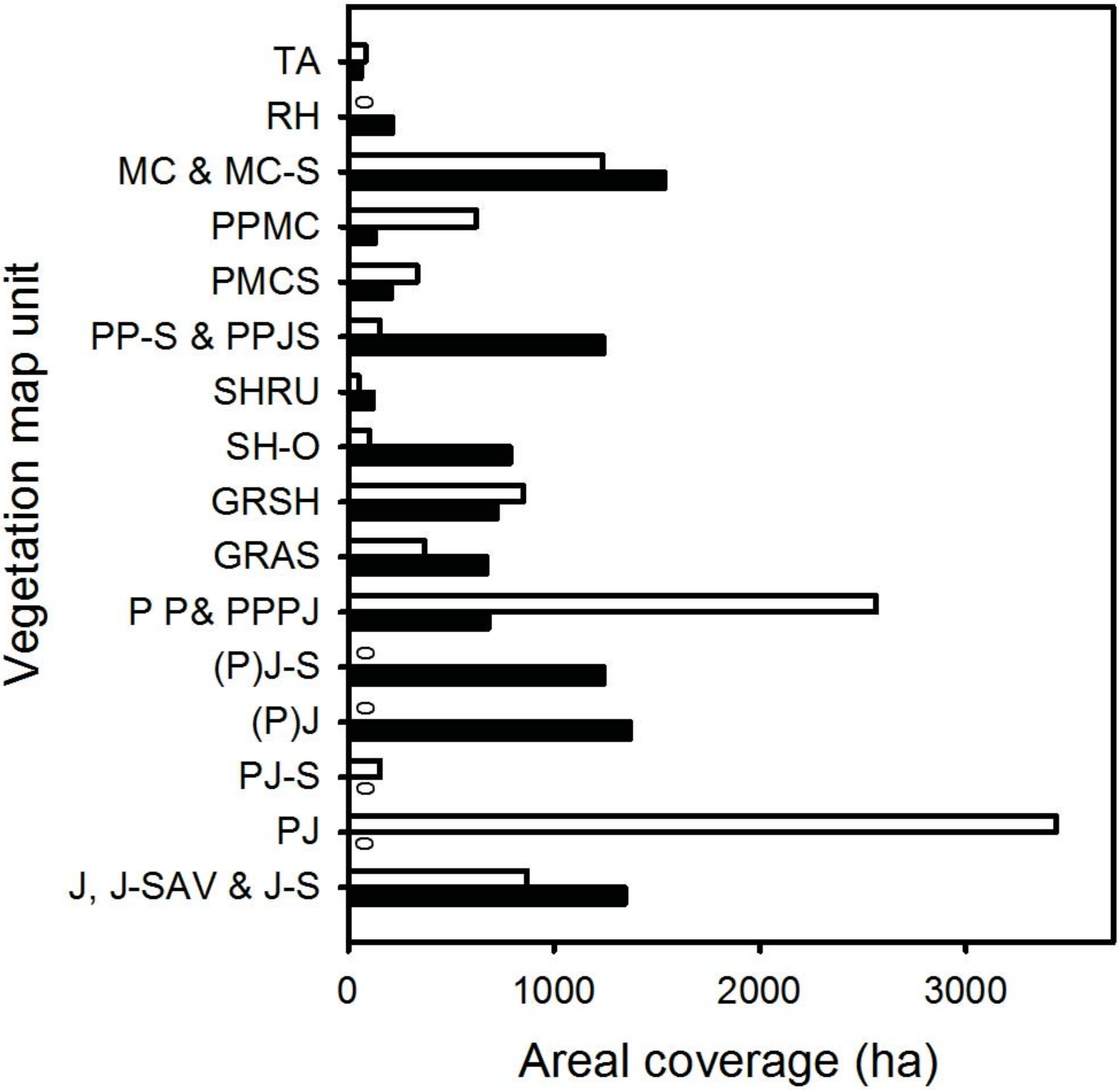
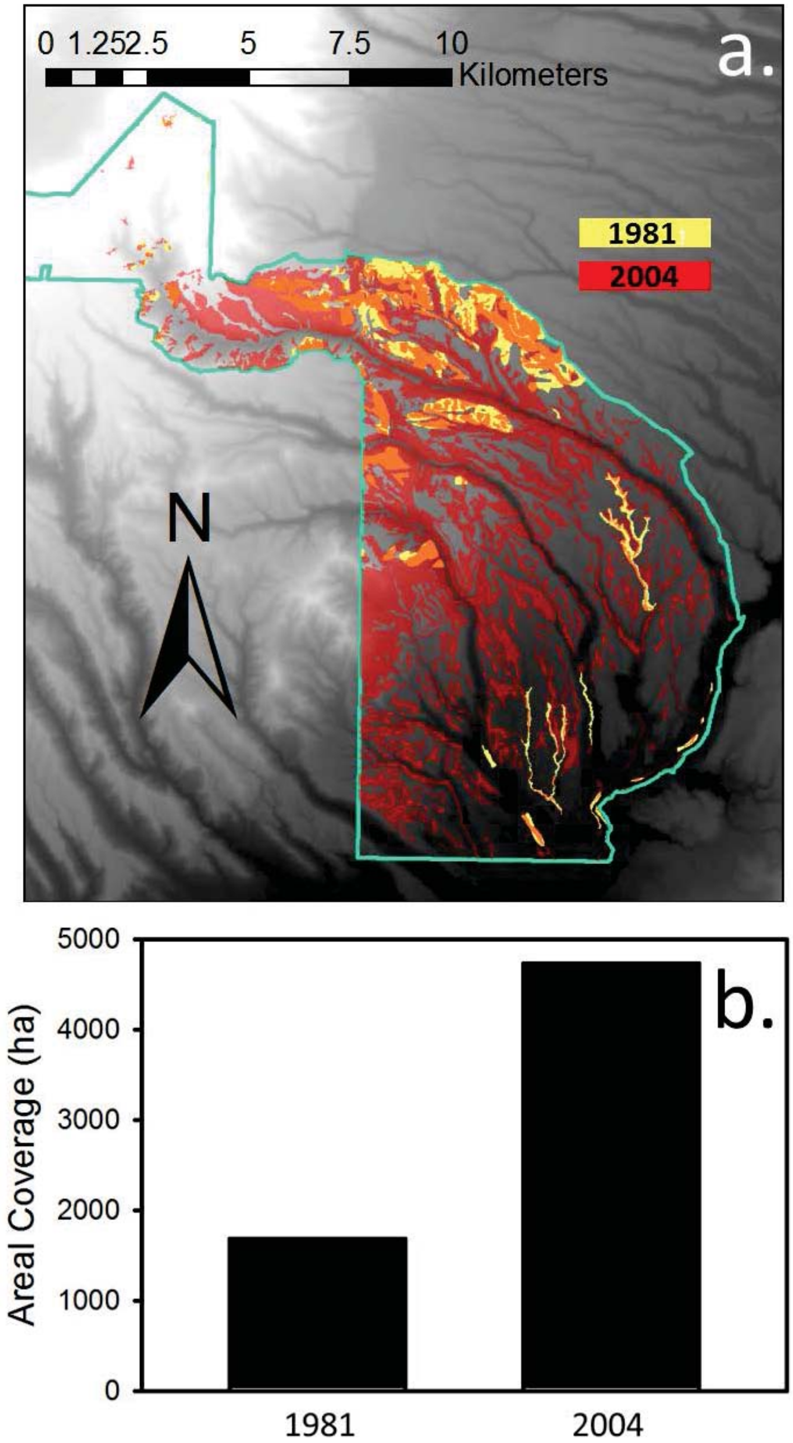


Figure 5. Expansion of plant communities with a shrub component (J-S, PJ-S, SH-O, SHRU, PP-S, PPJS, PMCS, MC-S) from 1981 – 2004.a. Spatial distribution of shrubs as a major community component. b. Areal coverage of shrubs as a major community component.



Further upslope, the spatial distribution of mixed conifer-dominated stands have been fairly stable over this time period, possibly declining by less than one third (Figure 3), although the more recent Las Conchas fire which is not captured in either map has undoubtedly reduced the coverage. Further, the former ecotone between Ponderosa pine-dominated communities and those dominated by mixed conifers is also developing a shrubbier aspect, particularly *Quercus* spp. It is difficult to draw conclusions about *Populus tremuloides* stands. They appear to represent a shifting mosaic, as would be expected for a species associated with disturbance (Figure 3), but do not strongly change in coverage on the landscape (Figure 4).

The analysis of canopy cover change also reveals some intriguing results and all essentially support the same conclusions. First, all three versions of canopy cover in the 1980's indicate similar spatial patterning: the highest canopy cover was found in mixed conifers forests, followed by piñon-juniper woodlands grading into ponderosa pine forests, followed by former ponderosa pine forests which had been impacted by the La Mesa Fire (Figure 5a,b,c). The spatial pattern of post-2002 canopy cover is captured in Figure 5d. It recovers the same pattern in a general sense, but differences among the different portions of the watershed are less pronounced. The changes in canopy cover from the 1980's to the 2000's are presented in figures 6-8. As with the analysis of the spatial distribution change above, we must not overinterpret the results, because some apparent patterns may simply be artifacts arising from differences in the data collection protocols in the source datasets. One result which is dubious is the apparent decline in vegetation in Frijoles canyon. This likely arises from the fact that the Muldavin et al. (2011) dataset has relatively few points in the canyon bottom, thus the interpolated estimates may be strongly influenced by nearby points on mesa tops. Most of the patterns we observe on mesa tops seem to correspond well to known events giving us confidence that our estimates are plausible.

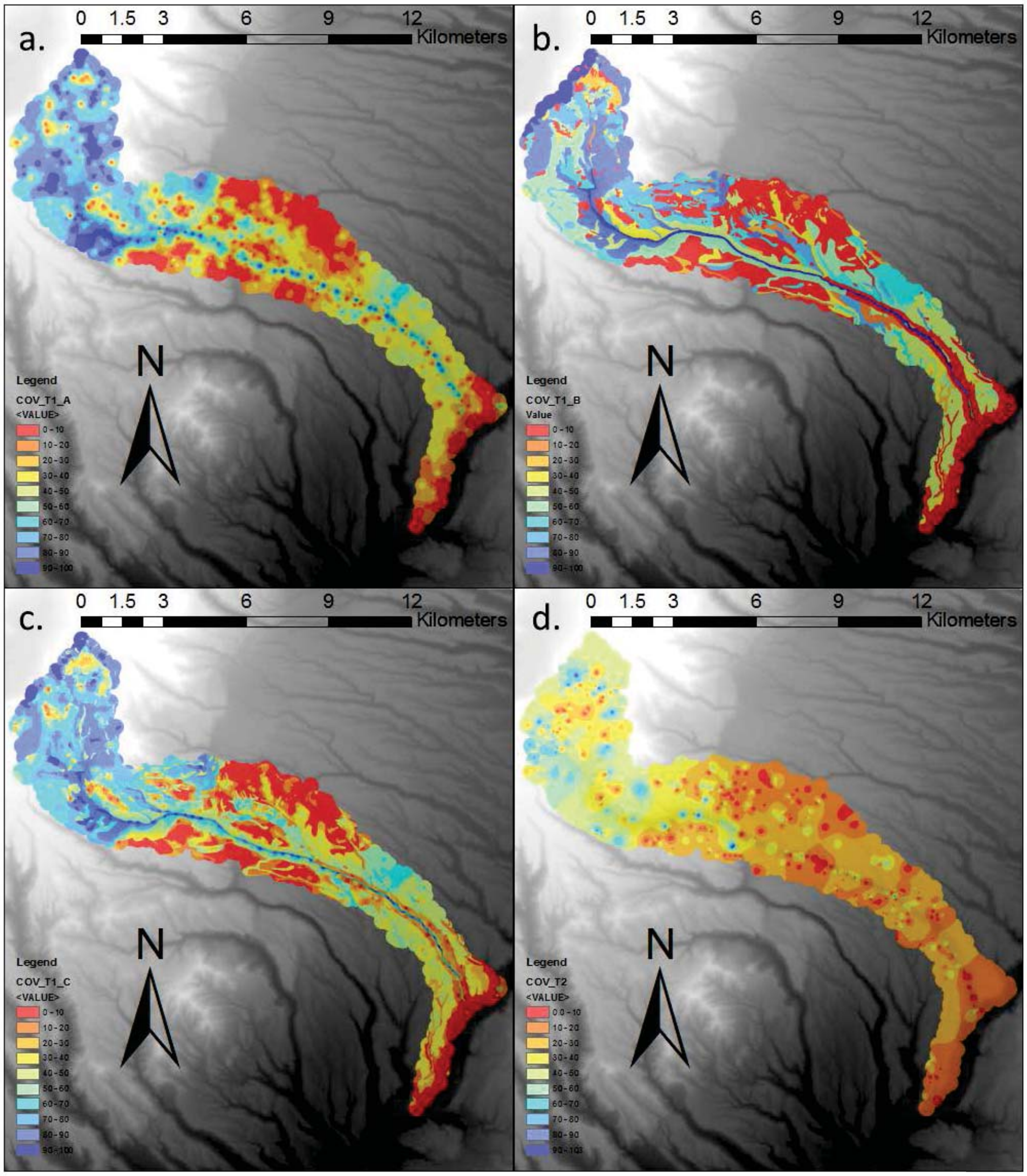


Figure 5. Estimates of canopy cover: a. 1980s A, based on interpolation of ground-based data, b. 1980s B, from vegetation map based on 1981 imagery, c. 1980s C, a mean of 1980s A and 1980s B, d. post-2002 based on interpolation of ground data.

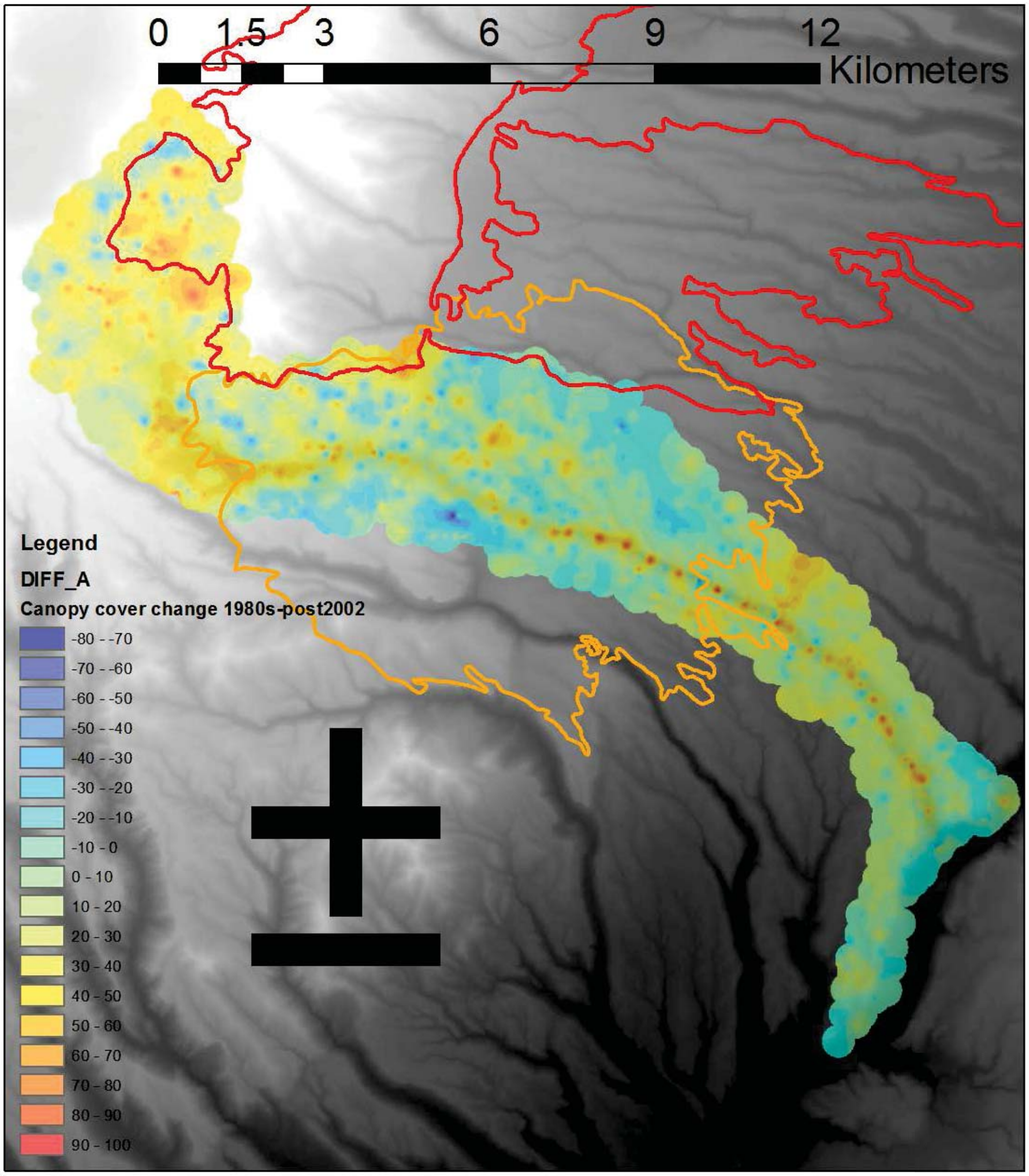


Figure 6. Estimates of canopy cover change in between the 1980's and post-2002 (Diff A). Positive values and reds indicate canopy loss, and negative values and blues indicate canopy gain. La Mesa fire boundary is indicated in orange, Cerro Grande fire boundary is indicated in red.

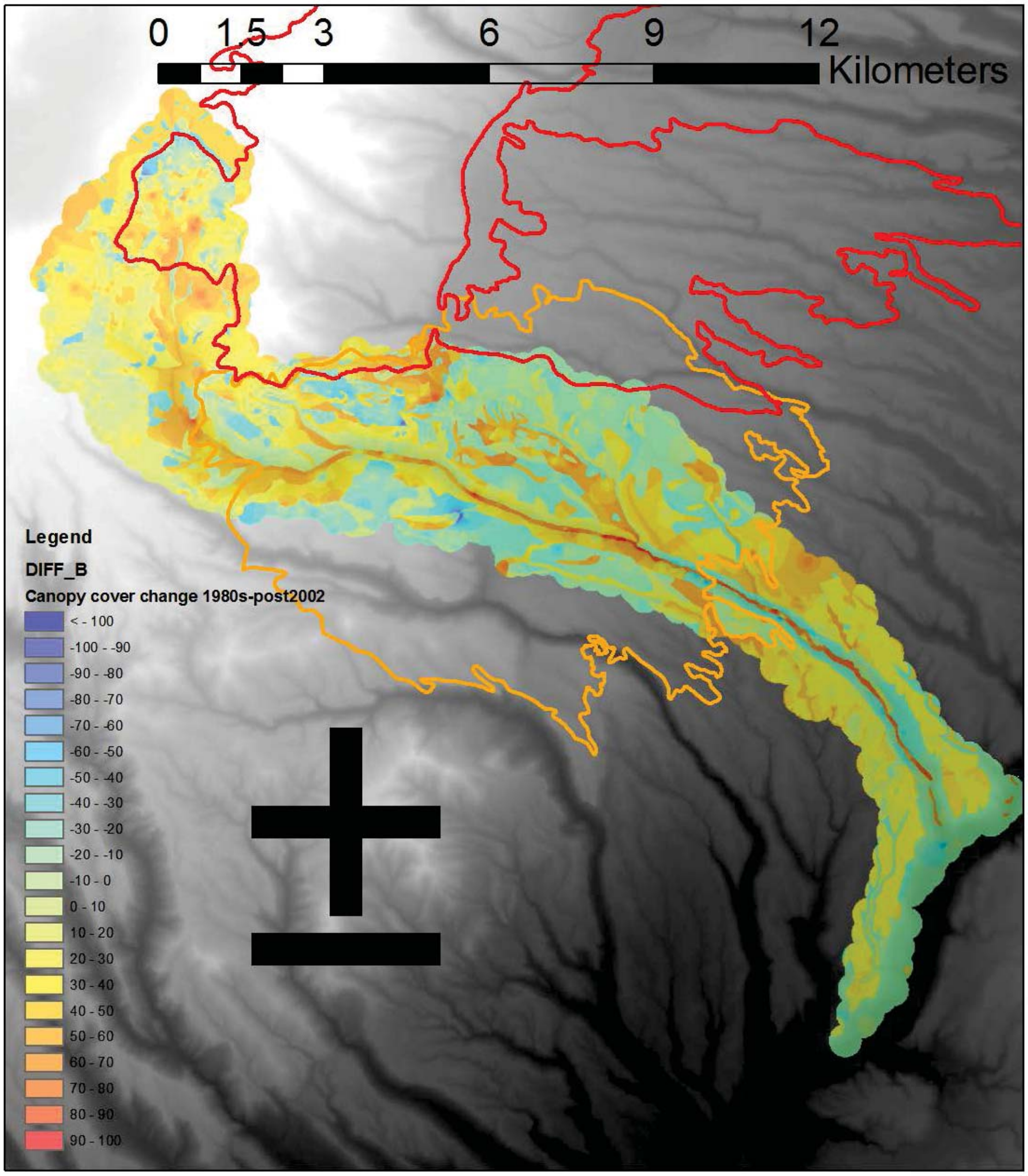


Figure 7. Estimates of canopy cover change in between the 1980's and post-2002 (Diff B). Positive values and reds indicate canopy loss, and negative values and blues indicate canopy gain. Because canopy cover values exceeded 100% in the Muldavin dataset, it is possible for canopy gain to also exceed 100%. La Mesa fire boundary is indicated in orange, Cerro Grande fire boundary is indicated in red.

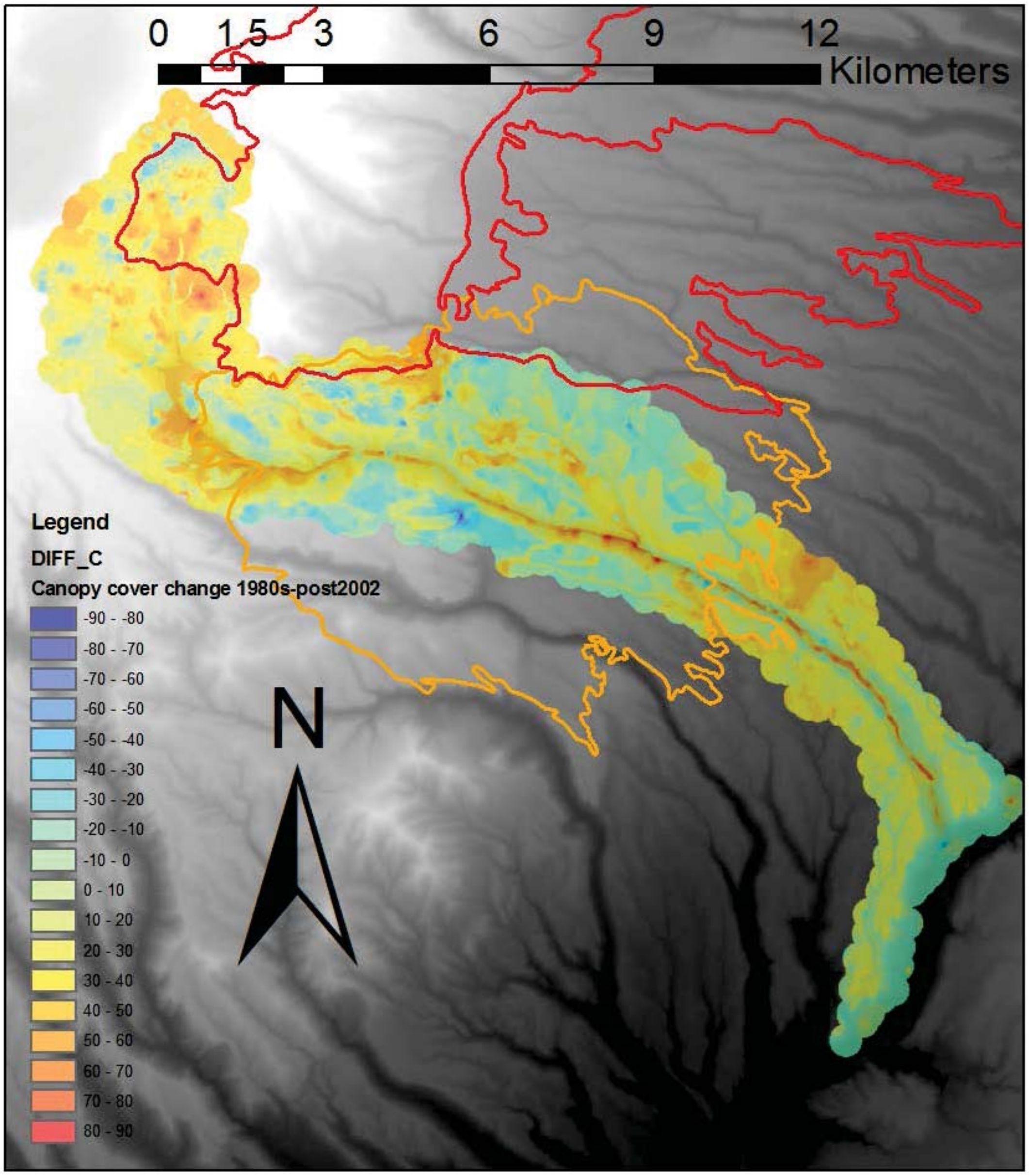


Figure 8. Estimates of canopy cover change in between the 1980's and post-2002 (Diff C). Positive values and reds indicate canopy loss, and negative values and blues indicate canopy gain. La Mesa fire boundary is indicated in orange, Cerro Grande fire boundary is indicated in red.

Averaged over the whole extent of our analysis, the monument appears to have lost between 14 – 15% canopy cover. The lower elevation mesa tops, almost without exception, indicate some degree of canopy loss. Typical canopy loss in this area dominated by former piñon-juniper woodlands and ecotones of this ecosystem with ponderosa pine ecosystems was approximately 30 %, although some local experienced more than twice as much canopy loss. Northward, and higher in elevation from these areas, the area within the boundary of the La Mesa fire is experiencing canopy gain, usually less than 20%, indicating that there is some slow recovery of forest canopy after about 30 years. There are smaller points within this area that have recovered a closed canopy, but they are few and isolated. Still higher in elevation in the mixed conifer – aspen forests, the overall pattern is one of very heterogeneous net loss. Near complete loss of canopy is estimated for the upper portions of the canyon, and further north in areas which were burned by the Cerro Grande fire. Curiously some of these aggregations of canopy loss are adjacent to but outside to the Cerro Grande fire perimeter.

Summary & Projections

In summary, canopy cover has strongly declined in the higher and lower thirds of the Frijoles watershed. This in itself is not a departure from desired conditions, since most forests and woodlands had too much canopy cover, and since canopy reductions in wetter portions of the Monument might result in greater water flows to the canyon due to decreased evapotranspiration (Adams et al. 2012). Associated with this canopy loss is some loss of characteristic trees from some communities. For example, mature *Pinus edulis* were eliminated by drought and bark beetle outbreaks. Ponderosa pine has been essentially eliminated as a dominant from some former habitat by fires. Mixed conifer forests have changed but have largely retained the characteristic trees as dominants; they display a heterogeneous nature of shifting canopy densities.

The La Mesa fire may be slowly recovering canopy cover, though it is unclear if the apparent canopy gain results from differences between the Allen (1989) and Muldavin (2011) data. Our analysis of vegetation communities indicate these areas are gaining shrub cover rather than canopy cover.

There are several clear directional changes in the spatial distribution of different vegetation communities. For example directional expansion of currently juniper-dominated ecosystems to higher elevations has occurred, although these have recently converted to juniper dominance due not to migration by juniper but rather by mortality of its former codominant. *Pinus edulis* showed no signs of moving upward in elevation, rather the large majority of adult individuals were lost in most spatial locations. The lower boundary of ponderosa pine forests has shifted upslope, and ponderosa pine forests have become increasingly fragmented, colonized by shrubs, or otherwise altered. In recent centuries, ponderosa pine has always extended to high elevations where it was often a component of mixed conifer forests, or even a dominant on southerly aspects. The loss of fire led to an expansion in the range of *Abies concolor* and *Pseudotsuga menzissii* and an expansion into lower elevations formerly more dominated by ponderosa pine (Allen 1989). This trend may be reversing, as the upper boundary of ponderosa pine-dominance may be shifting upward to some degree, and the prevalence of mixed conifer forests dominated by species other than ponderosa pine may be declining in this area. Lower ponderosa pine-mixed conifer stands appear to have transitioned in a direction toward ponderosa pine-shrub communities, but otherwise mixed conifer forests have not strongly shifted in their spatial location.

Overall, these losses of distinctive overstory species, directional shifts of some major ecosystem types, proliferation of former subdominants, and drastic or outright loss of some dominant species on the landscape indicate that the Monument as a whole is well outside of desired conditions. These changes may be creating greater fine scale heterogeneity which is desired in a broad sense, but it comes at the expense of strongly altered community compositions and unacceptable loss of dominants. Since all of these changes are indirectly or directly and partially or wholly driven by climate change (Figure 1), we can predict that similar big, fast, and patchy transformations (Breshears et al. 2011) will continue to occur. Even though climate change cannot be managed at a local scale, correction of this trajectory is not independent of management actions. One factor, wildfire, is influenced by climate change because fire is favored when temperatures are higher or vegetation contains less water, but it is also a very much product of over a century of fire suppression and fuel accumulation (Figure 1). What we see in these analyses is a recent pulse of large scale, high frequency, and high intensity fire events because climate and fuel conditions that are ideal for fire have aligned. The fires have consumed considerable fuel, as evidenced by the changes in canopy cover, but management of fuel conditions in the remaining stands that have not been burned intensely may yet discourage similar events from occurring.

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Chapter 2: Analysis of mixed conifer-aspen forests

Indicators/Measures:

- ***Change in plant community dominance through time in response to fire***
- ***Post-fire aspen regrowth with and without the presence of herbivores***
- ***Aspen height, % cover, and degree of browsing as indicators of herbivore pressure***
- ***Spatial correspondence between browsing intensity and subsequent aspen height***

Background & Importance

The mixed conifer- aspen vegetation type is a mosaic of distinct forest types. In vegetation mapping, map units constitute a mappable type of forest patch which may be composed of one or more typical vegetation assemblages. Multiple stands which are discrete in space may be mapped under the same map unit. Muldavin et al. (2011) mapped 6 Level 1 map units, and 13 Level 2 map units; thus one could say there are 6 general types and 13 specific types of mixed conifer-aspen forests. These types differ by the identity of overstory dominants and understory characteristics. These community characteristics are determined by the landscape setting (e.g. slope and aspect, position in a canyon or mesa top), climate, herbivore activity and disturbance history. Overstory configurations include: 1. *Picea engelmanii*, 2. *Populus tremuloides*, 3. *Abies concolor-Pseudotsuga menziesii-Populus tremuloides* 4. *Abies concolor-Pseudotsuga menziesii –Picea pungens-Populus tremuloides* 5. *Abies concolor-Pseudotsuga menziesii*, 6. *Pseudotsuga menzeisii-Abies concolor-Picea pungens*. Understories maybe dominated by graminoids (e.g. *Festuca thurberi* or *Danthonia parryi*) or shrubs (e.g. *Quercus gambelii* or *Acer glabrum*).

In particular, the aspen component of the mixed conifer forest is of great significance for a variety of reasons. Aspen groves are widely regarded as having great aesthetic value. McCool (2001) points out the prevalence of aspen in iconic landscape photography, notably that of Ansel Adams. In addition the fall colors of aspen provide a visual contrast that is widely appreciated (McCool 2001). Aspens contribute to community biodiversity more than other forest types including the mixed conifer forest matrix in which they are embedded. Chong et al. (2001) examining 10 distinct community types, found that the steepest species accumulation curves were found in aspen stands. Because of this property and because they signal successional recovery from fire, aspen has been suggested by some as a key indicator of ecological health in forests (White, Kay and Feller 1997). Finally, aspens clones are significant because they are ancient. It is very rare for aspens to reproduce by seed (Romme et al. 2001), rather they reproduce almost exclusively clonally (Barnes 1966, 1975, Kemperman & Barnes 1976). Seeds require continuous moisture for germination (McDonough 1985), yet germinating seedlings will not persist in the presence of forest floor litter (Barnes 1966). Successful establishment by seed is so rare that it is noteworthy when it occurs (Quinn & Wu 2001). Rather, it has been proposed that almost all extant clones established greater than 10,000 years ago and have maintained themselves over millennia through vegetative means (Mitton and Grant 1996).

The mixed conifer-aspen forests in Bandelier are an important component of the National Park system in the southwestern United States. Although the Colorado Plateau ecoregion is rich in NPS units, few of them have significant acreage of mixed conifer-aspen forests. As stated in chapter 1, the SCPN forms a useful frame of reference when considering biodiversity conservation in NPS units of the southern Colorado Plateau ecoregion. In the SCPN those units that do have significant mixed conifer forests (Grand Canyon National Park and Bandelier

National Monument) are isolated from one another. The montane areas of the Colorado Plateau are islands in an ocean of desert and semi-desert. Much like oceanic islands, each one has some elements that make it distinct.

Although it the most widespread tree species in North America (Worral et al. 2013), *Populus tremuloides* is generally declining in the western US (Kay 1990, Kay 1997, Kulakowski et al. 2013, Schier 1975), with trends varying from place to place. Declines may be caused by a variety of factors. Without fire, *Populus tremuloides* generally cannot compete with conifers. Thus, fire suppression throughout the 20th century has largely omitted the fire-caused stand openings that provide opportunity for *Populus tremuloides* clones to proliferate until recently. Patton and Avant (1970), working in a portion of the Santa Fe National Forest, demonstrated that aspen stems recovering from fire can attain 9m in height in only 6-8 years. In this study herbivore populations were low. Herbivore populations in the Southwest have seen a substantial increase in the last century due to predator suppression, increase in ground water in the form of cattle tanks (Binkley et al 2006) and (re-) introduction (Truett 1996) Now when fires occur, herbivory can constrain the growth of new ramets (Kay et al. 2000, Binkley et al 2006). Sheppard and Fairweather (1994) estimate that 10-15 years of protection from herbivores might be necessary to allow an aspen ramet to attain a sufficient size to make it safe from herbivory.

In recent years the so-called “Sudden Aspen Decline” has lead to mass mortality of mature *Populus* clones. Recently researchers have asserted the cause to be the combination of severe drought and warm temperatures (Anderregg et al. 2013, Worral et al. 2013). While this is a concern regionally, thus far in Bandelier National Monument there are no such cases of unexplained death of mature aspens (S. Fettig personal communication), though it remains a possibility in the future.

Major Drivers of Change in Mixed Conifer-Aspen forests-

Fire suppression and subsequent large scale crown fire: Fire is a part of the evolutionary environment of the mixed conifer-aspen forest and is required for the regeneration of *Populus tremuloide*s. There are unusual cases where do aspens occur in pure stands and conifers are excluded due to dispersal or edaphic, limitations; however, these situations do not seem to occur at Bandelier where aspen regeneration is strongly fire-linked. Allen et al. (1995) describe a frequent fire regime prior to the initiation of fire suppression policies in the 20th century. Many trees survived multiple fires, suggesting that these historical fires were not predominantly stand-replacing. We consider fire here as a stressor and a driver of vegetation change because the scale and intensity of recent fires has been unusual (Fig. 2). Fires affecting at least some mixed conifer-aspen forest include: the 1977 La Mesa Fire, the 1996 Dome fire, the 2000 Cerro Grande fire, and the 2011 Las Conchas fire. Fire intensities varied within these fires but all of these events involved significant stand replacement. These large fires are likely a result of over a century of fuel build-up due to fire suppression.

Climate change: The western US is currently experiencing the warmest temperatures observed in a millennium (IPCC 2007). Coupled with warming have been altered precipitation patterns. These warming trends, and a cool season drying trend are expected to continue increasing in magnitude over the next several decades (Karl et al. 2009). Together, these factors lead to conditions which facilitate larger and hotter fires (Williams et al. 2010). Without the enhanced fuel load conditions due to fire suppression these impacts would not be as great because forests would be thinner and canopy less continuous, making extensive crown fires much less likely.

The current pulse of megafires seems to be an interaction between past management and these new climate conditions (Figure 1). Another major expression of climate change is drought.

Drought & Insect Outbreaks: Periodically, severe droughts may be associated with pulse mortality of overstory trees. Insects such as bark beetles and the fungi they disperse lead to mortality of water stressed trees. One such drought occurred in 2002 (Breshears et al. 2005). It caused major mortality to *Pinus ponderosa* and *P. edulis*, especially in lower elevations but also impacted *Pseudotsuga menziesii* and *Abies concolor* at higher elevations (Muldavin et al. 2011). Primarily mature trees were affected rather than saplings. Canopy loss in mixed conifer-aspens forests is estimated in Chapter 1 (Figs. 6-8).

Elk Herbivory: In addition to several understory shrubs, saplings and suckers of *Populus tremuloides* are palatable to elk (Wooley et al. 2008), which are major herbivores in Bandelier and adjacent Valles Caldera National Preserve. Elk numbers are largely unchecked by native predators, with the exception of some calf predation by black bears or coyotes. In addition, large numbers of elk can winter in Bandelier if snow depth is high in other parts of their habitat. In Bandelier they benefit from protection from hunters. This is significant because in the Jemez mountains, elk populations are restrained by human hunting and some road mortality. Lastly, Elk herbivory has shown to inhibit the regrowth of *Populus tremuloides* groves, eating as much as 59% of new aspen shoots each year (Kay 1990 ,Kay 1997, Kay et al 2000, Wooley et al. 2008).

Elk herbivory can interact with fire to eliminate aspen. High intensity fire may destroy aboveground mature aspens stems. Normally, it would be common for an aspen to resprout after an event like this. However, under intense herbivory, energy stores in roots which grow into new shoots are lost. Because new shoots are unsuccessful, fixation of carbon and replenishing the lost energy stores is not possible. Eventually the root systems can die, and new shoots do not sprout.

These mechanisms in tandem eliminated aspen root systems in two multi-hectare areas within Bandelier in high-intensity burns within the Cerro Grande fire. Failure of root systems at scales like this could mean the permanent loss of an ancient aspen clone.

Reference Conditions

A study of pollen records in the Jemez mountains suggests a timeline for the development of the current mixed-conifer community (Anderson et al. 2008). The area is not known to have been glaciated. Around 14000 ybp *Picea engelmanii* is present in the pollen record, followed by *Pinus ponderosa* around 11500 ybp, followed by *Pinus flexilis* or *Pinus strobiferous* around 10500 ybp, and *Pseudotsuga menziesii* around 10,000 ybp. The period 8500 – 6400 ybp is characterized by frequent charcoal deposits suggesting regular fire. By around 6400 most modern species are present in the pollen record, with a gradual increase in the abundance of Abies over the last 6000 years. Allen et al. (1995) and Toucan et al. (1996) provide a better picture of recent centuries and most notably a strong change in fire frequency after the turn of the 20th century. Prior to the 20th century, mixed conifer forests were shaped by fires, most of which were low intensity ground fires. Crown fires did occur, however they were patchy and were not areally extensive, and possibly mostly confined to wetter north slopes and adjoining areas. The fire return interval was about 15 – 30 years (Allen 2004). Fire histories of the highest and wettest spruce-fir forests is poorly known, but crown fire may have been part of the fire regime (Allen 2004). Euro-American settlement brought about the introduction of large numbers of livestock in the 1870’s which reduced fine fuel loads and coincided with the cessation of fire (Sydoriak et al.

2000). The Wheeler expedition of 1873 reported 12,000 head of sheep, goats and cattle in the Valles Caldera area which is essentially adjacent to upper elevation portions of Bandelier National Monument (Allen 2004). Active fire suppression followed in the early 1900's (Sydoriak et al. 2000).

Most of the current *Populus tremuloides* stands appear to have regenerated in between 1850 and 1910, possibly in response to a series of stand-opening crown fires that occurred at this time. It is important to keep in mind that the stands may have been new, but they regenerated from ancient clones that were already present (Milton and Grant 1996). There is no reason to believe that aspens did not regenerate well after fire in New Mexico during this time period, as they have been shown to do so when not constrained by other factors like herbivory (Patton & Avant 1970). These trees were much less likely to have been constrained by elk herbivory because elk are believed to have been a relatively minor species (<3% of large mammals) in the western US before Euro-American settlement because they are poorly represented in archaeological and paleontological records (Allen 2004). This contrasts strongly with the current situation where across the west there has been major concern about elevated levels of elk herbivory (Murie 1944, Beetle 1962, Basile 1979), and the paucity of aspen regeneration for much of the 20th century (Ripple & Larsen 2001). In Bandelier, Merriam's elk was a native subspecies that likely fed on *Populus tremuloides* that was present in small numbers until extinction due to hunting in the early 1900's (Allen 2004). The elk which currently inhabit Bandelier are a different subspecies (re-) introduced from Rocky Mountain National Park (Truett 1996). There likely are some ecological distinctions between the two subspecies, but the main difference between the pre-1900 and current role of elk in the ecosystem is based on sheer numbers. Elk simply were not abundant before the 20th century. Despite being relatively easy for

Native American hunters to hunt, they appear only very scarcely in archaeological records (Truett 1996). Mexican gray wolves and grizzly bears inhabited the area until extinction in the 1930's, and likely played some role in reducing elk numbers. Another predator, the cougar, has been the target of eradication efforts throughout the 20th century but have persisted in low numbers. Altough the elimination of these predators was likely a boon for elk, human hunting pressure and limited surface water were probably greater constraints. Another difference between the 19th century and now is that elk are not naturally a forest species. They prefer open areas where they feed on herbaceous plants and can see their predators form a distance. With the advent of the rifle, elk can be killed from a distance and have found an advantage in areas with hunting of spending greater amounts of time under forest cover and consequently browsing forest species more heavily. A reference state would ideally be significantly after Puebloan occupation, before large-scale introduction of livestock and fire suppression and during a period of time when elk were present in low densities. Like most ecosystems in Bandelier National Monument, the time period best matching these conditions is between 1860 and 1870. One problem with this time period is that it follows shortly after the Little Ice Age, an anomalously cold period from 1550 – 1850 during which vegetation was likely altered. This cool weather might have seen incursions of forest types typical of higher elevations into lower elevations, suggesting that the 1850s spatial distribution types was inherently transitory. For example, perhaps one reason why mature ponderosa pine retracted upslope in a large pulse of mortality in the 1950's is that the trees established during an unusually cool period (Allen & Breshears 1998). This cool period of time is also unusual because it followed the 1500's megadrought event (Williams et al. 2012). Despite these shortcomings of the historical reference condition concept, it is possible to develop a broad picture of the characteristics of mixed conifer-aspen forests in Bandelier and the western US in general.

All available information regarding this time period suggests a mosaic of forest types in various stages of disturbance containing *Picea engelmanii*, *Abies concolor, Pseudotsuga menzissi*, and *Pinus ponderosa* with a frequent fire regime and episodic *Populus tremuloides* proliferation following isolated crown fires. While we know there were relatively pure stands of aspen in the past (Toucan et al. 1996), it remains unknown what the physical structure of these stands were, or what the height and age distributions of ramets were.

Staff at Bandelier National Monument have developed a more specific set of *desired future conditions* (Bandelier National Monument 2006). These desired conditions largely reflect an interpretation of conditions in the past based on available data and expert opinion, and are specific enough to represent an operational reference condition with which to compare current condition and establish trend. It should also be recognized that “desired conditions” continually shift based on major change events and based on what seems remotely possible in a warmer, dryer future. Two types of desired future conditions are currently recognized, an uneven-aged stand type and an even-aged one. The uneven-aged stand type contains some older growth alongside other age classes, supports multiple species sharing dominance, has an open structure with a patchy shrub or herbaceous understory and thus patchy under- and mid-story fuels, and is maintained by frequent fire every 15 – 25 years. The even-aged stand type is confined to steep, upper elevation environments, and north facing slopes. This type is maintained by episodic, stand-replacing crown fires up to several hundred acres in extent with a return interval > 100 years. In between disturbances these forests accumulate large amounts of litter on the soil surface. *Populus tremuloides* is a common component of mixed conifer forests, but *Populus*-

dominated groves represent a persistent seral community which infills openings created by crown fire. Under desired conditions, mixed-conifer crown fire gaps would be expected to become *Populus* stands if *Populus* was locally present prior to the fire. In the absence of fire, *Populus* stands are gradually in-filled by conifer species.

Data & Methods

Vegetation Maps

Datasets

Two vegetation maps have been produced in recent decades encompassing most or all of Bandelier National Monument. Allen (1989) produced a map based upon aerial imagery from 1981. Muldavin et al. (2011) produced a map using supervised automated digital processing, assisted by multispectral imaging. This map was confined to the entire extent of Bandelier National Monument, and was based upon 2004 imagery. To facilitate comparisons among the two maps, both were reduced to their intersection. Since a map can be simplified but not made more complex without new data, we focused on reclassifying the more detailed Level 2 map units of the Muldavin et al. (2011) to the system used by Allen (1990) to enable a direct comparison of the two (*see Chapter 1 for details*).

Analysis

Within the boundary of the 2000 Cerro Grande Fire, we wished to determine the fate of the major forest types founds within the burn boundary (MC, TA, PPMC) in 1981. To do so, we clipped both vegetation maps to the intersection of Bandelier National Monument and the Cerro Grande fire, and converted both to rasters with 10m grid size. We determined the proportional coverage of different 2004 vegetation community types within the areal extent of each of the three major 1981 vegetation types in this extent.

2001 – 2006 Elk exclosures data

Sampling design

Five elk exclosures were established in areas vegetated by mixed conifer forests, to track the influence of elk herbivory on aspen recovery, in addition to impacts on higher trophic levels. Each exclosure was 60 × 60 m in area, and paired with an unfenced reference plot with a 20 m buffer area between. One pair of plots was burned in the 2000 Cerro Grande fire. For each of the five exclosures, and their adjacent controls, surveys counted the number of aspen individuals, and their frequency by height class.

Analysis

This was a paired repeated measures design, with the unplanned and unreplicated factor fire. The first step of our analysis was to calculate a useful response variable. Since the height classes used were adjusted in various years of sampling, a common system was established which placed them in 0.5m bins up to 2m, and in 1 m bins from 2 – 7 m. Greater than 7m was used as an additional class. A mean height was estimated by averaging the bin median heights, weighted by the proportion of the total trees within each bin. Rather than use degrees of freedom to estimate a blocking term which we were not interested in, we centered all values of response variables within each block prior to analysis. Since one block was burned and the others were not, this essentially omitted any main effects of burning in the data. We conducted a repeated measures ANOVA with the factors: exclosure, time, burning, exclosure × burning, time × exclosure, time

× burning (burning was included to enable interaction terms). Exclosure × burning uses different measurements through time as replicates. Time × burning uses plots in and out of exclosures as replicates.

Aspen recovery in random plots in the Cerro Grande fire vicinity in 2002 & 2005

Sampling design

This dataset consists of random locations sampled in 2002 and 2005. The study focused on *Populus tremuloides* and several other elk herbivory-susceptible shrub species. The majority but not all 2002 points were within the Cerro Grande fire. The majority of the 2005 points were outside the fire boundary. A point consists of a 10-meter by 10-meter plot at each point in which maximum height, number of growing tips of each species (up to a maximum of approximately 200 tips) and, frequency of browsing on current year growth of shrubs was recorded. These data have the advantage of providing information about recent browsing which may differ considerably from year to year. The disadvantages are that it cannot measure the cumulative impacts of multiple years of browsing, and that the data have an upper bound at 100% (additional browsing on older tissues would not be accounted for). The study focused on *Populus tremuloides* and several other elk-susceptible shrub species. This study design enables a predictive model of the determinants of elk herbivory, both in fire-impacted landscapes and areas not recently affected by fire.

Analysis

We used classification and regression trees to model *P. tremuloides* height, and % of tips browsed as a function of spatial coordinates, elevation, pre-fire vegetation type, slope, aspect, and whether or not a site was located within the Cerro Grande fire. The classification & regression tree method partitions response data based upon the values of one or more predictor variables. The most explanatory predictor is found by an algorithm, and the most explanatory value of that predictor is used to split the response data. This results in an “if-then” type rule which ascribes samples to one group or the other (e.g. if elevation is greater than or equal to a, then aspen height is b; if elevation is less than a, then aspen height is c). The result of such a partition is that data are placed into more homogenous groupings, and residual variance declines. The algorithm will continue making partitions, based upon the same or other predictor data, until either stopping rules are invoked or until the modeler terminates the process or “prunes” the tree. In our case, tree size (number of splits) was determined using Akaike’s Information Criterion, which simultaneously takes into account likelihood of a model in addition to penalizing complexity. Trees were grown overly complex, then pruned to the simplest complexity within 2 of the lowest AIC value observed. We used a 5-fold bootstrap crossvalidation method to calculate the predictive power of our model (expressed as R^2). Because sampling strategies differed among the two years they were modeled separately.

Aspen recovery in targeted “refuge” plots in the Cerro Grande fire vicinity in 2006

Sampling design

During the collection of the previous dataset, it was observed that palatable saplings and shrubs appeared to be refuged by on-site elements such as downed logs or rocks, similar to observations by Ripple and Larsen (2001). These refuge elements possibly concealed sprouts or simply increased the risk of injury for elk in browsing these refuged stems, and thus could be an effective temporary aid to aspen sprouts. Eventually, in most cases, the aspen sprout will outgrow its refuge and could be susceptible to browsing if browsing intensity is sufficient. This dataset focused on characterizing the properties of sites in which *Populus tremuloides* or elk-susceptible shrubs may be refuged from elk browsing. Field crews searched the entire Cerro Grande burn area seeking examples of instances where an aspen or shrub may be benefiting from a refuge element. When such a case was located, crews marked and sampled a 100 m^2 plot centered on the refuged individual or individuals. A total of 115 plots were sampled in this way. Within the sampling universe of existing refuged plants, this dataset enables the identification of the specific characteristics of refuge elements that confer greater or less protection. Greater protection is indicated by greater refuged height and/or canopy cover. *Populus tremuloides* was present in a majority of plots but not all plots. Since all plots can be reasonably be expected to be potential *P. tremuloides* habitat, in cases where the species was absent a zero value was used for height and cover.

Crews recorded the spatial coordinates, elevation, slope, aspect and openness of canopy in each site, and whether or not there was evidence that the location had burned during the Cerro Grande fire. Refuge elements considered were standing trees, fallen trees, other live plants, rocks, small sticks, and miscellaneous refuge elements which did not fall into each of these categories. Within each plot, crews measured the cover, height, and diameter (in the case of fallen trees) for each refuge element.

Prior to analysis, summary variables were created of this data in consultation with the original data collector, Steve Fettig: minimum diameter of fallen trees, average diameter of fallen trees, minimum height of refuge element (regardless of the type of refuge), and cover of standing live trees.

Analysis

In an analysis complementing that of the previous 2002 & 2005 datasets, we used classification and regression trees to model *P. tremuloides* height, and cover as a function of the predictors above (northing, easting, elevation, aspect, openness, burned [yes/no], minimum diameter of fallen trees, average diameter of fallen trees, minimum height of refuge element [regardless of the type of refuge], and cover of standing live trees). Slope was omitted at the request of Dr. Fettig for two reasons: 1. The slopes were generally not steep enough to influence elk activity, 2. Slope covaries with and could obscure effects of burning and other factors.

Pooled data: exclosure study, random plots, targeted refuge plots

Datasets

We produced maps of % aspen browsing in 2002 and 2005, as an indicator of early post-fire herbivore pressure, and maximum aspen height in 2006, as an indicator of aspen resprouting success. Both were prepared using the Inverse Distance Weighting technique to interpolate surfaces based upon the points sampled in 2002, 2005, or 2006. To test the hypothesis that early post-fire herbivore pressure constrains aspen height, we calculated a correlation coefficient between the two datasets.

These two datasets and the exclosure data all contained reasonably comparable aspen height data. Within each exclosure or control plot the median of the largest size class present was used as an estimator of the maximum height of aspens. We obtained the maximal height in burned and unburned exclosure and control plots for 2001, 2002, 2005, and 2006. The random surveys of 2002 and 2005 each recorded the tallest individual aspen in a fixed area plot. The targeted survey was an exhaustive search for aspens which might be resprouting in a similarly sized plot. A maximal height was recorded for a similarly sized plot as used in the random surveys. We calculated mean maximal plot-level aspen heights for 2002, 2005, and 2006.

Analysis

Our previous analyses were focused on variation on the landscape in resprouting of aspen and browsing pressure. As a simple means of gauging the park wide recruitment of aspen, and the impact of elk browsing, we simply plotted all of these data through time to enable comparisons.

Condition & Trend

Vegetation change specifically in the mixed conifer zone of Bandelier National Monument is largely driven by fire, herbivory, and drought and associated pathogen outbreaks. After about a century of fire suppression and relatively few fires, large and often intense fires are strongly impacting the mixed conifer-aspen forests of Bandelier National Monument. The current millennium has been characterized by “mega-fires” in the region. The 1977 La Mesa fire, 1996 Dome fire, and 2001 Cerro Grande fire all burned some mixed conifer-aspen forests within Bandelier National Monument (Figure 2). Each of these fires burned moderately to intensely in patches, opening canopies in patches up to hundreds of hectares. The scale of these patchy openings is not outside of the desired conditions of the mixed conifer-aspen forests within the Monument. The Las Conchas fire was much larger and followed only 10 years after Cerro Grande. Virtually the entire southern half of the 1981 distribution of mixed conifer-aspen forests was impacted by Las Conchas (Figure 2d). The scale of this event is clearly outside of the desired conditions. The cumulative outcome of these fires is that more than half of the 1981 distribution of mixed conifer-aspen has experienced at least moderate severity fire.

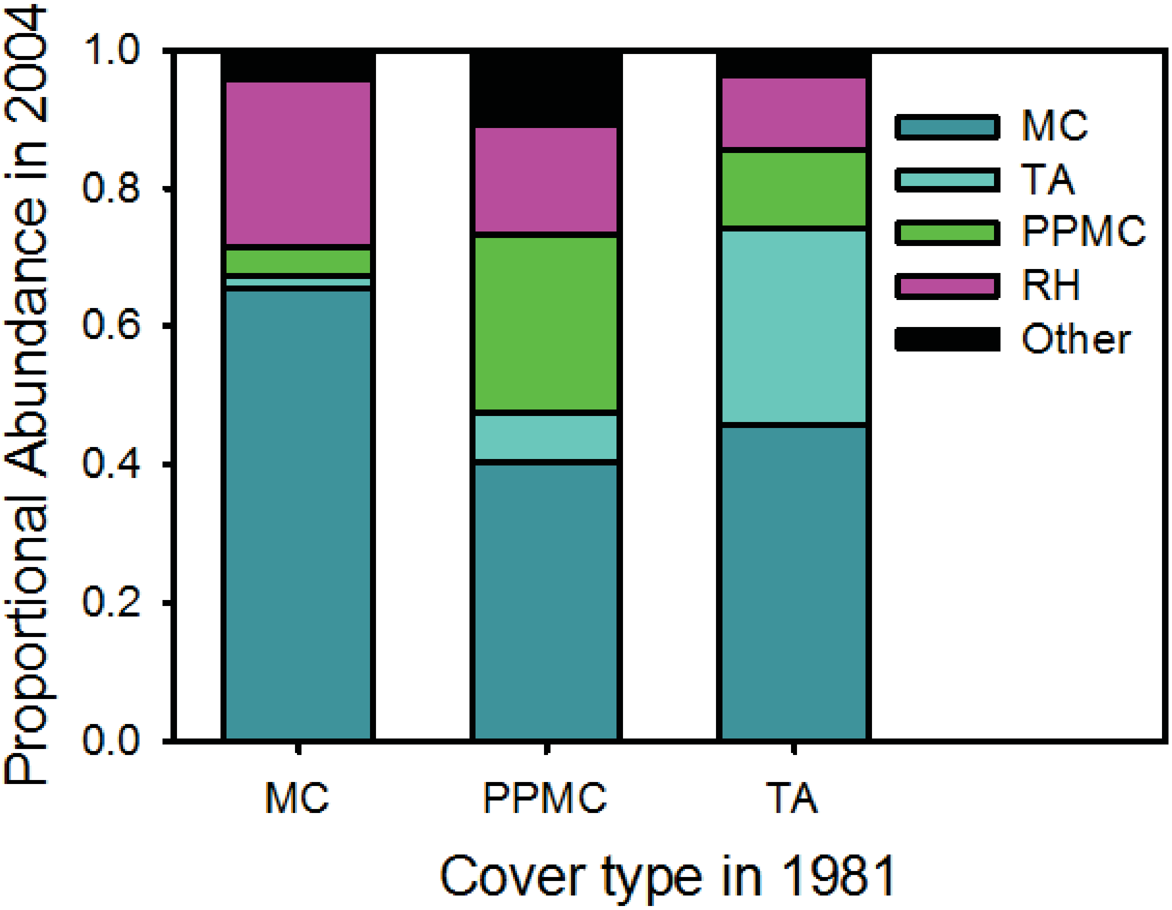

Figure 9. Fate of three key 1981 cover types (Allen 1990) within the Cerro Grande fire perimeter, based on the cover types mapped by Muldavin (2011) based on 2004 imagery.

The extent of the impacts is clearly the majority of mixed conifer-aspen forests in the Monument (Figure 2), but the consequences of these large burns and rapid transitions is not entirely clear. If the ecosystem were in a reference state or desired condition, it would be expected that recently burned areas would experience a pulse of aspen clone recruitment from root suckers and an increase in abundance of resprouting shrubs such as *Quercus* spp. It is too early for substantial data to be available to track the long-term response of the recent Las Conchas fire in mixed-conifer aspen forests. However it is possible to observe change within

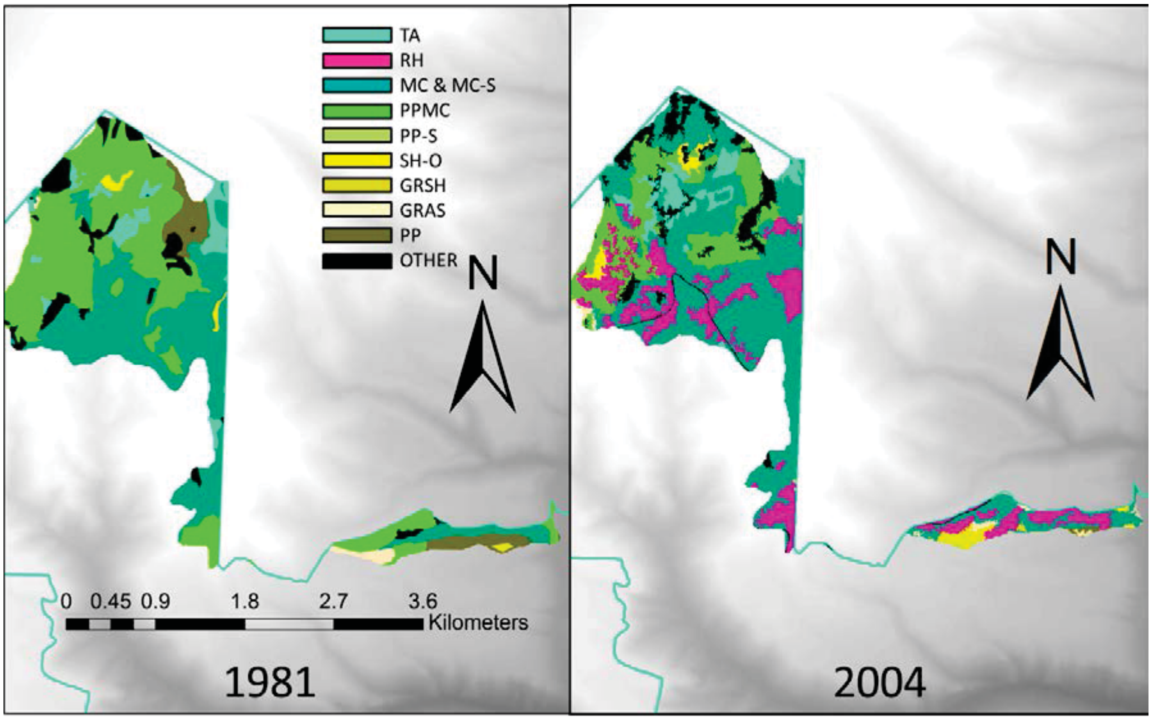

Figure 10. Vegetation communities within the Cerro Grand fire perimeter as mapped based on 1981 imagery (Allen 1990) and 2004 imagery (Muldavin 2011).

major cover types from 1981 – 2004 within the Cerro Grande fire perimeter (Figure 9, Figure 10). Most impacts of the Cerro Grande fire within the Bandelier National Monument boundary were actually low intensity. We cannot attribute all vegetation change within this time period to the fire, which occurred in 2000. Another important event was the drought mortality pulse, which began around 2002; however the impacts of this drought were more extreme in other ecosystems within the Monument such as piñon-juniper woodlands. Within areas mapped as mixed conifer (MC) by Allen (1989), over 60% retained this cover type, but about a quarter have been converted to a ruderal herbaceous community (Figure 9). One notable fire disturbance-linked community type that does not appear to be increasing are aspen groves (TA). The most notable difference 4 years after the fire are the ruderal herbaceous communities, which comprise about 25% of the former area of communities mapped as MC within the burn perimeter. An examination of the communities mapped in 1981 as Ponderosa Pine - Mixed conifer (PPMC), indicates that about half of these stands best match either PPMC or MC after the fire (Figure 9). Less than 20% of these stands have transitioned into ruderal herbaceous communities. A small proportion of these stands are mapped as aspen-dominated after the fire. Only about a quarter of 1981's Aspen stands (TA) were mapped as such in 2004, after the fire (Figure 9). Rather, about half are conifer-dominated post-fire. Less than 10% of these stands converted to ruderal herbaceous vegetation however. Spatially, the overall trends are the apparent decline in total coverage of ponderosa pine/mixed conifer communities, the increase in ruderal herbaceous vegetation, and a decline in contiguous patches of similar community types (Figure 10).

The trajectory ofapost-fire recovery of mixed conifer or aspen forests does not seem to be toward proliferation of aspen clones. Aspen is a difficult community type to map unless it is a pure stand. Relatively pure stands are what would be expected to increase after stand opening fires, which were also thought to be the agents that allowed recruitment of the current mature aspen groves. In this fire perimeter, aspens are neither gaining nor losing total area; rather they seem to create a shifting mosaic on the landscape gaining area in some places while losing similar amount in others. This lack of aspen increase is perhaps the starkest deviation from reference or desired conditions. To what degree can these results be believed? These conclusions are based on vegetation maps produced by different teams using different methods. It is possible that criteria for distinguishing among mixed conifer and ponderosa pine/mixed conifer stands were simply different, thus an apparent expansion of ponderosa pine dominance could be an artifact. Another stark deviation between the two maps is the new ruderal herbaceous communities. According to Muldavin (2011) these are dominated by *Pascopyrum smithii* along with *Bouteloua* spp. and a high diversity of forbs. We cannot rule out that these will eventually be colonized by trees (aspen or conifers), or have been since the 2004 imagery on which the maps are based, but there is insufficient data available to know if this has occurred or not. It also appears as though a finer-scaled heterogeneity is emerging (Figure 10). This observation may be influenced by a slightly smaller grain size used in the Muldavin (2011) map compared to the Allen (1989) map, but is a result that would be expected in a disturbed environment and is in line with the desired conditions for this ecosystem type.

The elk exclosures are instructive in suggesting why aspen-dominated stands did not increase post-fire. We found clear evidence that elk herbivory constrains *P. tremuloides* growth (Figure 11). In the first five years following the Cerro Grande, elk exclosures and nearby controls did not have clearly different numbers of aspens. The numbers of aspens were highly variable among the different exclosures-control pairs, and in some pairs the exclosure had a greater number of sprouts, while in others control plots had more sprouts. This variability likely results from three linked factors: 1. Whether the plot burned, 2. The degree of shading in the understory, and 3. Soil temperature fluctuations due to different degrees of canopy closure. This suggests that the ability of P. *tremuloides* to *initially* resprout in an area as suckers from roots of a surviving clone is not clearly reduced by elk in this dataset. Elk browsing and debarking of mature aspens could reduce sprouting vigor, but these data account only for exclusion of such impacts since 2000.

Regardless of their numbers, these individuals are subject to intense herbivory which constrains their size. On average *P. tremuloides* saplings were constrained to around 40 cm in height and did not show an upward growth trend. Inside exclosures, aspens were clearly

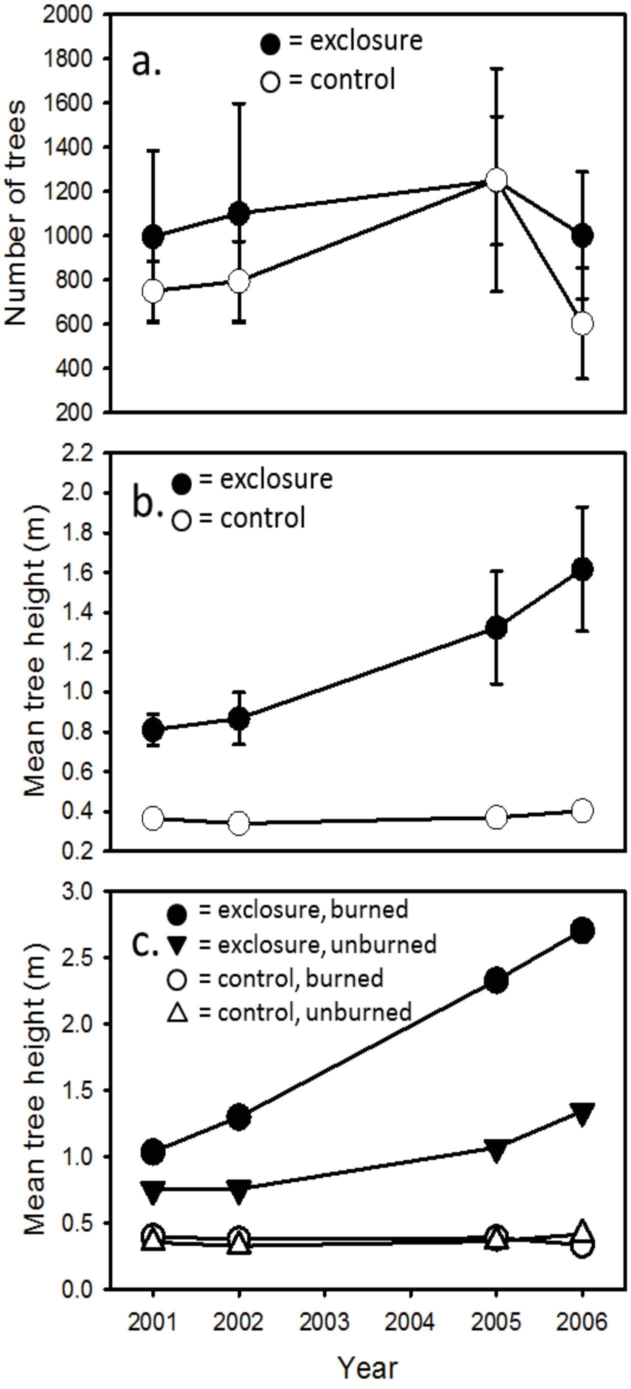

Figure 11. Number and mean height of aspen stems inside elk exclosures and in control plots outside of exclosures over five years. a. None of the considered factors affected number of aspen stems (Exclosure: *F*=1.6, *P*=0.26 Time: *F*=1.3, *P*=0.39), b. Mean aspen height was much greater inside exclosures, increased over time inside exclosures, and increased most inside burned exclosures (Exclosure: *F*=137277, *P*<0.0001; B × E : *F*=57.2, *P*=0.0003; Time: *F*= 160.0, *P* 0.0001; E × T: *F*=8.9, *P*=0.10), c. Illustration of the B × E interaction.

becoming larger over time and the growth rate increased at each increment. In 2006, they had attained a height about 4 × that of their elk-affected counterparts. While in the short-term, this herbivory does not seem to result in mortality of aspens, it is reasonable to assume that a long-term continuation of this pattern would eventually result in the depletion of energy reserves and death of these stems as observed in two 10 acre areas within Bandelier and elsewhere (Lindroth & St. Clair 2013). Perhaps most interesting was the fact that fire intensity and elk exclusion seem to interact. In Figure 11c, we can see that control plots have similar, small aspen heights regardless of fire intensity. But, when protected from herbivory, the burned plot had aspens sprouts nearly double the size of the unburned plots. This suggests that aspen are well-equipped to recover from fire as long as they are not constrained by herbivory. These results are not surprising and are in line with several other studies of the elk-aspen interaction in western North America.

Several of our analyses focused on the variation in indicators of elk herbivory on *P. tremuloides* across the landscape. We wished to determine which are the strongest influences upon intensity of herbivory and whether these could be predictively modeled. We had three datasets available pertinent to the recovery from the Cerro Grande fire dating from 2002, 2005, and 2006. In 2002 and 2005 we had direct observations of browsed tissue *on P. tremuloides*, supplemented with information on maximal sapling height, which would be expected to be under the influence of elk herbivory, but also climate and soil characteristics. In 2006, we had available two indirect indicators of elk activity: *P. tremuloides* maximal height and cover.

Perhaps the most striking conclusion that can be drawn from this set of analyses is that there is a strong spatial component to elk herbivory, regardless of the year (Figure 12, Figure13). Krantz (2001) compiled elk observations in Valles Caldera National Preserve in 2001, and found that the Rincon area of Valle Grande consistently had the largest number of elk numbering in the hundreds for most observations. This result is attributed to the fact that it is a large area featuring considerable forest edge. Because the Valles Caldera elk prefer to spend the night in open areas and spend days under tree cover, this area favors large numbers of animals. This demonstrates that the natural distribution of elk is highly variable spatially, and that there can be consistent hotspots of elk activity in a given year. The spatial pattern of elk browsing indicators in Bandelier National Monument may be demonstrating something similar. Interestingly, in 2002, it was the far eastern edge of the northern portion of the monument which experienced the greatest herbivory (Figure 12, 13). The northernmost portion of this area exhibited the smallest aspen heights. In 2005, models were much poorer in explanatory power (not shown: % browsing R^2 = 0.05; maximum height R^2 = 0.18) but the % browsing model indicates the greatest browsing N of Bandelier in lower lying areas, i.e. Valle Grande. Also in 2005, the smallest maximal tree heights were observed in the Bandelier portions of the Cerro Grande fire but were about twice as large east of the Monument boundary. The 2006 *P. tremuloides* height data suggests more northerly locations had shorter aspens, and that in between 3967997 and 3968536n, there were no aspens observed. Again this suggests a spatial heterogeneity in herbivore pressure. Some places experience much greater herbivory than other locations, and this patterning may shift from year to year (Wolf 2003).

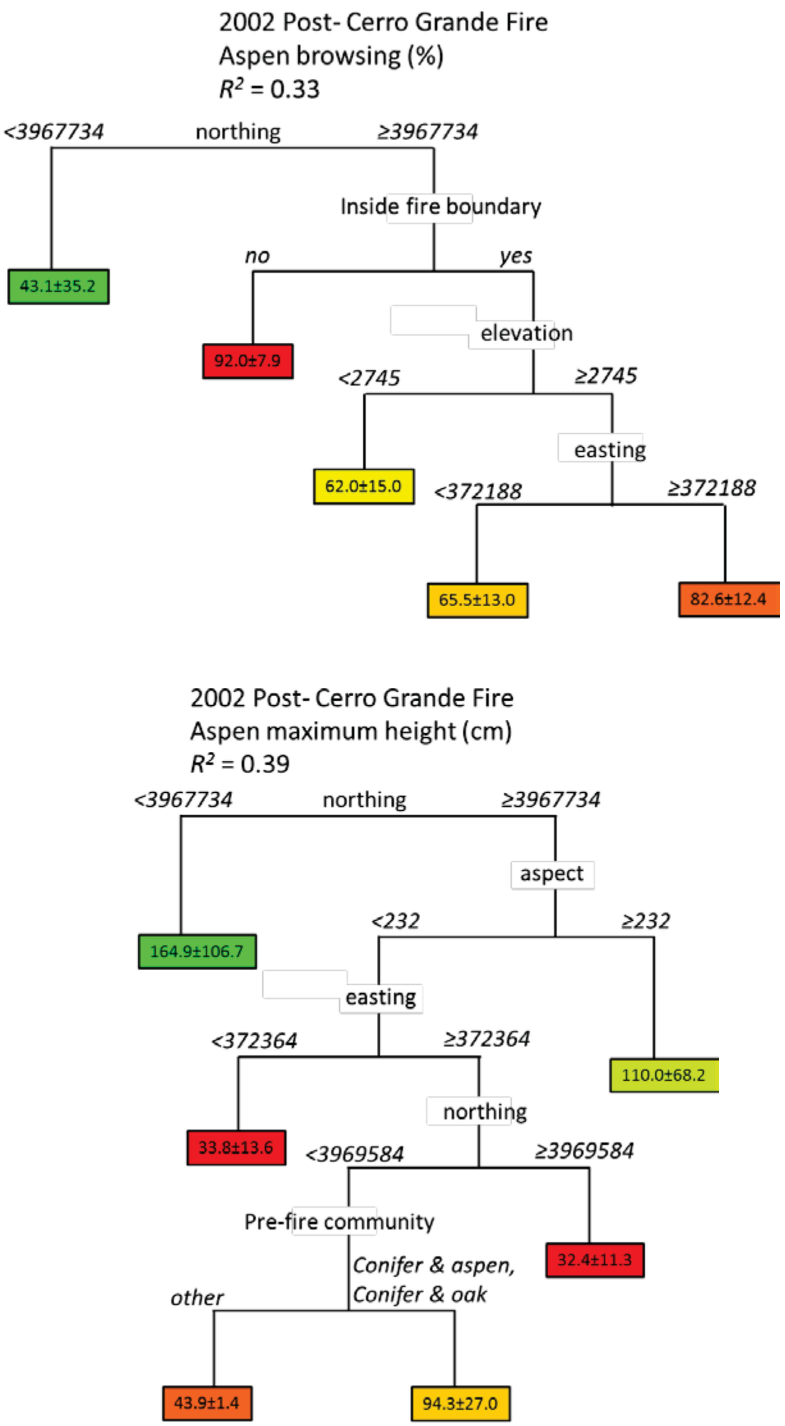

Figure 12. Regression trees modeling indicators of aspen recovery from the Cerro Grande fire in 2002, as a function of site characteristics. Figures are read from top to bottom like decision trees. Mean (± SD) estimates of an indicator of elk herbivory (top: % browsing; bottom: aspen maximum height) are reported in boxes. Boxes are color coded with reds indicating greater evidence of elk herbivory, greens indicating lesser evidence, and yellows intermediate.

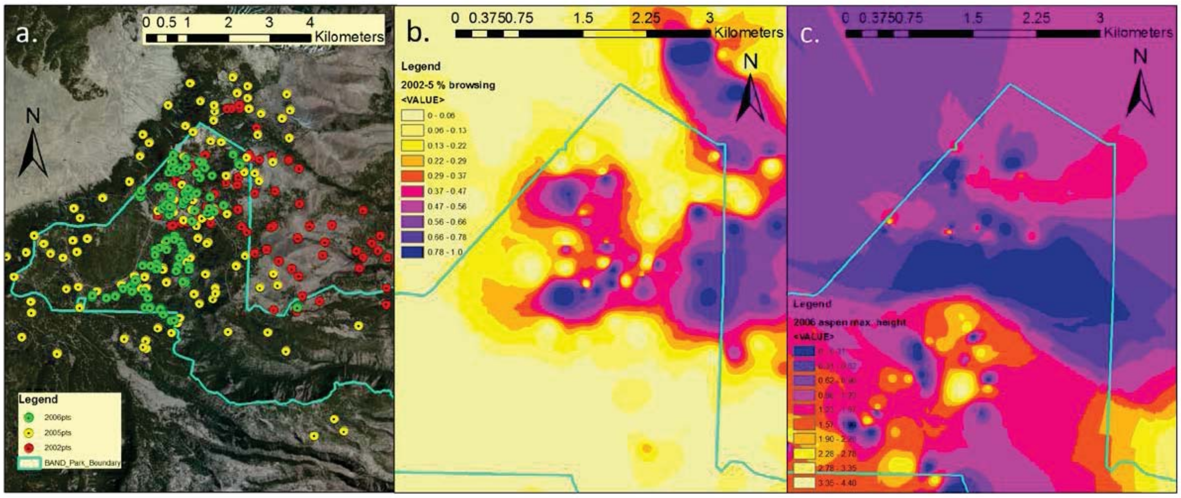

Figure 13. Map surfaces based on Inverse Distance Weighting interpolation of points corresponding to measurement of % aspen browsing or maximum height of aspens (m). a. Location of sampling plots in 2002, 2005, and 2006. b. % browsing on random plots sampled in either 2002 or 2005, representing the spatial location of elk herbivory in early years after the Cerro Grande fire. c. Aspen maximum height in targeted plots sampled in 2006, showing the spatial location of aspen reestablishment. These two datasets are strongly negatively correlated (r = -0.90), implying that earlier intense browsing constrains tree height later.

Aspect was important in 2002 and 2006 datasets. In 2002, the tallest aspens were observed in between 232 and 360° (Figure 12). In 2006 the smallest aspens were found between 0 and 104°, intermediate-sized aspens were found between 104 and 183°, and the tallest aspens were found between 183 and 360°. In 2006, the greatest aspen cover was observed closer in between 80 and 360° (Figure 14).

The intensity of browsing in both 2002 and 2005 was greatest in areas outside of the Cerro Grande fire. Despite this pressure, aspens tended to be taller in these areas unaffected by recent fire in both 2005 and 2006.

The 2006 dataset allowed us to also conduct an analysis which took into account some of these larger scale patterns driven by spatial position or site attributes in combination with onsite elements which could hypothetically protect aspens from herbivory and create refuges (Ripple and Larsen 2001). This analysis resulted in conflicting patterns. While refuge height and diameter of downed trees were both invoked frequently in the models (Figure 14), it cannot be said that larger refuges generally lead to taller aspens or greater aspen cover. In more southerly areas on less favorable aspects, minimum fallen tree diameters > 21 cm were associated with shorter aspen stems. In more northerly locations if the minimum non-tree refuge height was > 63 cm, aspens were about twice as tall (Figure 14). When this condition was satisfied, average fallen tree diameters > 14cm seemed to dampen positive effects upon aspen height. If however the minimum non-tree refuge height was < 63 cm, then minimum fallen tree diameters > 26 lead to aspens that were nearly twice as tall. These complex and confusing results likely reflect that although refuge elements likely do reduce herbivory (Ripple and Larsen 2001), they also may correlate with other site characteristics which are unfavorable for growth such as shading. For example large rock refuges may indicate a soil that is poor for rooting. Likewise, downed trees

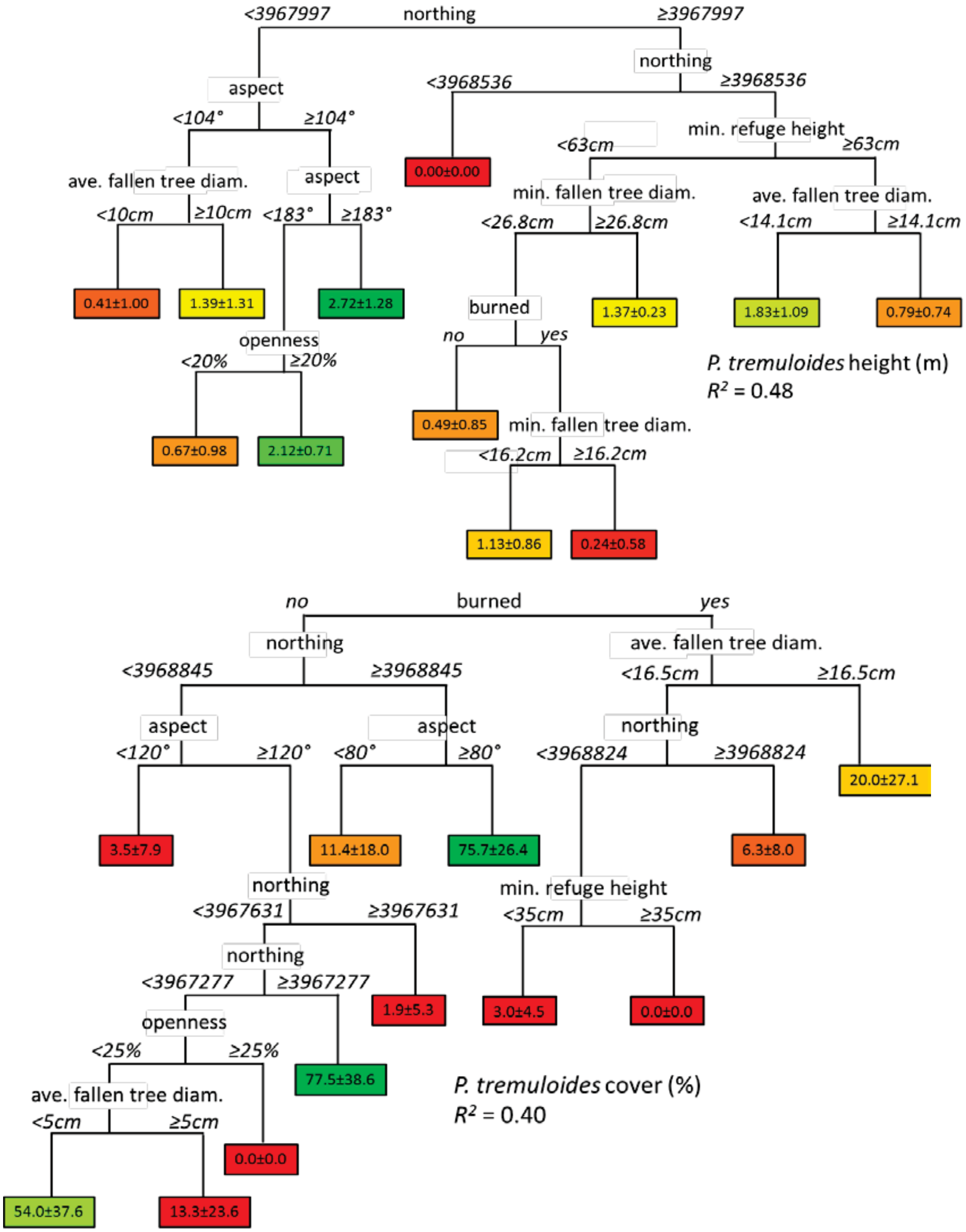

Figure 14. Regression trees modeling indicators of aspen recovery from the Cerro Grande fire in 2006, as a function of site characteristics, and charcteristics of on-site refuge elements. Figures are read from top to bottom like decision trees. Mean (± SD) estimates of an indicator of elk herbivory (top: height; bottom: % cover) are reported in boxes. Boxes are color coded with reds indicating greater evidence of elk herbivory, greens indicating lesser evidence, and yellows intermediate.

could simultaneously protect and shade aspens. The effect of refuge elements is best addressed experimentally in the field.

While we can demonstrate that herbivory and size of aspens stems is heterogeneous across the landscape, and some of this variation can be modeled, it is also useful to estimate the monument-wide effect of elk herbivory on re-establishment of aspens. From Figure 13, we can state that there is a clear negative correlation between the spatial locations of heavy elk browsing in 2002 – 2005. From Figure 15, we can state that in 2002, 2005, and 2006, aspens are generally

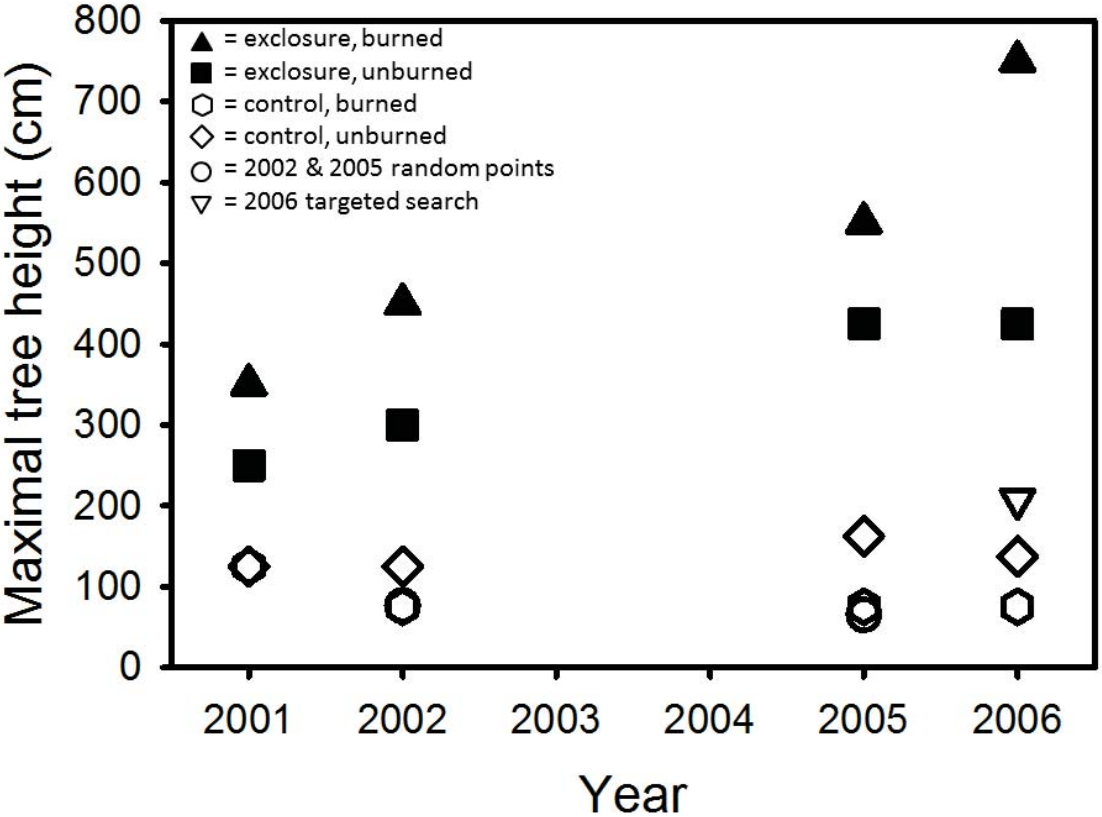

Figure 15. Aspen recovery was strongly limited by elk herbivory from 2001-2006. Aspen maximum height data were pooled across multiple studies. Dark symbols are plots inside exclosures. All open symbols are plots subject to elk browsing.

not attaining a size beyond which they can be considered safe from herbivory. Unless they reach this threshold, the sprouts are not likely to persist on the landscape. Most are constrained to around two meters, whereas protection from browsing can produce stems twice that height or even greater than three times that height in high intensity burn areas. The aspen heights measured in control plots in the exclosure study are very similar to randomly sampled plots in 2002 and 2005, indicating that the larger plot size of the exclosure study is not driving the results (Fig. 15). Because so few aspens were attaining sufficient size to escape browsing, the 2006 sampling was actually an exhaustive search for aspens which might be becoming large enough. Nonetheless, the average size of these refuged individuals was dwarfed by comparable sizes inside elk exclosures. This leaves us with the simple messages that 1. elk browsing is responsible for the general lack of recruitment of aspens in the Cerro Grande fire perimeter, 2. Aspens could be substantially larger in the absence of browsing.

Summary & Projections

Mega-fires appear to be a part of the future for the entire southwest region. This is due to a combination of high fuel loads due to a history of fire suppression (Allen et al. 1995), and also due to climate change (Williams et al. 2012). Megafires have already impacted a large portion of Bandelier, and will likely continue doing so until the vegetation structure is altered such that it can no longer support these fires. The current fire behavior is outside of the desired and reference conditions from this ecosystem. The fires are changing both community types on the landscape and their spatial distribution. They are leaving a sizable proportion of the landscape in a ruderal herbaceous state, and it is not known what successional trajectories are possible from this state. Historically, fires would have favored aspen proliferation, but this did not happen to in response to the Cerro Grande fire. A large part of this particular deviation from desired conditions appears to be related to elk herbivory since in the various datasets, only elk exclosures indicate establishment of aspens stems greater than about 2m. There has been some recent aspen sprouting activity after the Las Conchas fire, which is not captured yet in any extant datasets (S. Fettig personal communication). While this is promising, only a single heavy snow year could conceivably drive elk into Bandelier to winter reversing this trend and causing a failure of aspens to regenerate similarly to after the Cerro Grande fire (Wolf 2003). Herbivore intensity is very localized in particular areas (possibly different areas in different years) meaning that browsing intensity may be extremely high in affected areas, and that areas experiencing low browsing pressure in one year are not guaranteed to not experience it in subsequent years. Based upon the failure of aspen stems to mature outside of elk enclosures, the recovery of this major community component will likely be suppressed to some degree until the elk pressure is reduced. One possible action that could reduce elk herd sizes is to allow hunting within Bandeleier National Monument (Anonymous, undated). Perhaps most concerning is that continued heavy browsing in fire affected areas could conceivably lead to the loss of ancient clones. This has not been documented yet, but also has not been studied.

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Chapter 3: Analysis of ponderosa pine forests

Indicators/Measures:

- ***Cumulative footprint of fires***
- ***Change in plant community dominance through time in response to fire***
- ***Species interactions in post-fire succession***

Background & Importance

Like mixed conifer forests, the Ponderosa pine vegetation type is a mosaic of distinct forest, woodland and savanna types. These stands all contain ponderosa pine as the dominant or co-dominant species in the overstory. At higher elevations, ponderosa pine may intermingle as a subdominant in mixed conifer forests. At lower elevations, ponderosa pine occurs in mixed ecotones overlapping with piñon pine and juniper communities. In some areas understory shrubs may be important elements of the community (e.g. *Quercus gambelii*, *Robinia neomexicana*). Most commonly in southwestern ponderosa pine forests, understories are dominated by grasses (McGlone et al. 2009). In Bandelier, Muldavin et al. (2011), describe different associations with *Bouteloua gracilis*, and *Muhlenbergia montana*, and a unique high elevation association of ponderosa pine and *Festuca thurberi*. Exotic species are becoming increasingly prevalent in these understories, and may be favored by fire or slash pile burns (Korb et al. 2003). In many places, cheatgrass (*Bromus tectorum*) is becoming a major problem (McGlone et al.2009), but in Bandelier, *Bromus inermis* is a major species of concern that has been increasing in prevalence (Barclay et al. 2004). Canopy density ranges from nearly closed to open and savanna-like. At similar elevations grassland or shrubland openings may occur, but the larger of these are openings in ponderosa pine canopy created by the La Mesa fire of 1977.

The ponderosa pines of the southwest are a subspecies (var. *scopularum*) that is distinct from populations in California and the Northwest (DeWald 2008). The forest of *Pinus ponderosa* var. *scopularum* straddling the Mogollon rim at the southern edge of the Colorado Plateau is actually the largest contiguous forest dominated by ponderosa pine of any variety in the world. However, most ponderosa pine forests in the southwest are isolated from one another, and as a result have become genetically distinct (Rehfeldt 1993). Even at a sub-hectare scale there are detectable genetic differences between pre-settlement clumps of individuals and their offspring and post-settlement individuals that filled in the space between clumps (White 1982, Kolonoski 2002). These genetic distinctions may confer greater local adaptation to particular environments. In fact in an early provenance study, ponderosa pine individuals from other subspecies were unable to survive long-term in Northern Arizona (DeWald 2008). Thus, many NPS units in the SCPN that support substantial ponderosa pine forests support a distinct set of adaptations and genes that may be only weakly connected with other populations. Each one (e.g. Grand Canyon, Bandelier, Walnut Canyon, Sunset Crater) contains a portion of the regional gene stock and a portion of the potential for this species to adapt to future climates.

Southwestern ponderosa pine forests are generally in danger. Most forests are overstocked due to fire suppression and grazing of fine fuels (Allen et al. 2002). Many of today’s forests have been referred to as “doghair thickets” because of the large number of small diameter trees crowded on the landscape. Overcrowding of trees leads to less soil water (Zou et al. 2008), and therefore more mortality during droughts, but even more devastatingly it promotes extensive crown fires which were not part of the evolutionary environment of this species (Moore et al. 1999). We are entering a time period where the combination of extremely large wildfires and extreme droughts coupled with warm temperatures are compromising large swaths of ponderosa pine forests at an alarming rate. One needs only to consider the increasing fragmentation and shrinking coverage of this forest type in Bandelier (Fig. 3) to question whether these forests, and the genetic stock they contain, have a viable future.

The National Park system is one of the gene banks for ponderosa pine. The controversial practices of assisted migration (movement of a species into a previously unoccupied habitat in anticipate of a suitable climate) and "assisted gene flow" (movement of genetic stock within the current range of a species in anticipation of a suitable climate) will be increasingly discussed in coming decades (DeWald 2008, Hewitt et al. 2011). Resource and land managers must weigh whether the costs of introducing non-locally adapted genetic stock and the possible genetic pollution that comes with it is less than or greater than the benefit of any increase in the sustainability of the forest that is attained. Every isolated ponderosa pine forest contains genes that can potentially contribute to this use.

Major Drivers of Change in Ponderosa pine forests-

Climate change: The western US is currently experiencing the warmest temperatures observed in a millennium (IPCC 2007). As stated in previous chapters, global circulation models converge on a scenario where northwestern New Mexico will very likely experience continued warming, and possibly experience less spring precipitation (Karl et al. 2009). Warming is currently increasing the lethality of drought for trees (Adams et al.2009). Fuel accumulation due to fire suppression also interacts with climate change to produce larger and more intense fires.

Drought & Insect Outbreaks: One of the most important expressions of climate change are increasingly warm drought periods (see previous chapters; Adams et al. 2009). Ponderosa pine is one of the more susceptible trees to mass mortality during drought in the Southwestern US, for example in the 1950's (Allen and Breshears 1998), and in the years following 2002 (Breshears et al 2005, Gitlin et al. 2006) many individuals or even entire stands died. Multiple species of *Ips* bark beetles and the fungi they disperse lead to mortality of water stressed trees (Negron et al. 2009). Water stressed trees are less able to invest in defense mechanisms such as resin production (Wallin et al. 2004). Here, we consider drought-bark beetles to be a linked stress syndrome, and do not attempt to consider them separately.

Large scale crown fire: Fire is a part of the evolutionary environment of the mixed ponderosa pine forest and assists in opening its semi-serotinous cones (Moore et al. 1999). We consider fire here as a stressor and a driver of vegetation change because the scale and intensity of recent fires has been unusual. Rather than being similar to the ground fires characteristic of 1500 – 1900, the fires since the 1970's have burned in tree crowns. These large, intense fires have become frequent occurrences: The 1977 La Mesa Fire, the 1996 Dome fire, the 1997 Lummis fire, the 2000 Cerro Grande, and the 2011

Las Conchas fire all burned at least some ponderosa pine forests (Fig. 2), with the La Mesa, Dome, and Las Conchas being most impactful for ponderosa pine forests. All of these events involved significant stand replacement. These large fires are likely a result of over a century of fire suppression and fuel build-up interacting with warmer temperatures and drier fuel brought about by recent climate trajectories.

Reference Conditions

The historical state of southwestern ponderosa pine forests in general is well known due to excellent preservation of fire records and other lines of evidence. Paleoecological methods can provide a glimpse of recent millennia. For example, Anderson et al. (2008) find that ponderosa pine is first present in the pollen record about 11500 ybp. These sources of information may tell us what species are present and when they arrived but they tell us little about what these forests were like. Fortunately, there are many sources of information to draw from to help us understand forest conditions in the last two centuries. As with mixed conifer forests, determining a period of time to serve as a reference state is fraught with difficulty due to a history of flux in the last several centuries, including: 1. Settlement and abandonment by Ancestral Puebloans, 2. Megadroughts in the 1200's and 1500's, 3. Introduction of pastoralism, 4. The little Ice Age, and 5. Modern policies of fire suppression (See previous chapter). Most researchers have settled on the period just prior to euro-American settlement in the late 19th century, because this period seems to have displayed a natural fire cycle and because there are good sources of information about forest conditions and characteristics at this time, including: long-term or legacy datasets (Woolsey 1911), historical photographs and other records (Moore et al. 1999), stand structure reconstructions (Sanchez Meador et al. 2010), and reconstructed fire histories based on burn scars (Touchan et al. 1996). Not all researchers agree that this time period should be upheld as a strict-sense target for restoration because it is a much cooler time period that the anomalously warm current regime (Allen et al. 2002). Another source of information are extant isolated habitats (e.g. mesa tops) that have burned at a natural frequency due to a lack of historic grazing and the prohibitive expense of suppressing fire there (e.g. Powell Plateau, Grand Canyon National Park; Laughlin et al. 2004). The value of such sites is that *they exist now*, meaning that they have been shaped by recent climate trends rather than solely by a cooler past. The problem with these sites is that they are rare on the landscape. The Jemez Mountains lack such a true physical reference site.

An active research group at the Ecological Restoration Institute in Flagstaff, Arizona have invested considerable research into defining reference conditions for southwestern ponderosa pine forests in general (e.g. Covington and Moore 1994, Fulé et al. 1997, Moore et al. 1999), aided by a long-term presence of forestry research in the area of Flagstaff, Arizona and the Grand Canyon (Sanchez Meador et al. 2010). The picture that emerges is one of mostly open canopied forests and savannas (Allen 2002, Fulé et al. 1997). Tree densities ranged from about 8 – 25 per hectare based on legacy data (Allen 2002, Woolsey 1911). Spatial clumping of trees was a common characteristic, with most recruitment near adult trees because recruitment is *via* seeds which are not well dispersed far from the parent (Covington and Moore 1994). Understories were primarily grassy due to sufficient light reaching the soil surface, and provided the fine fuels necessary to carry ground fires. Shrubs such as *Quercus gambelii* may have been components of understories, or canopy co-dominants (Abella 2008). Low intensity ground fires were a frequent occurrence which maintained this structure (Allen et al. 1995). Such fires cull recruiting ponderosa pine saplings, but allow the survival of many adult trees. In summary, unlike many of today's southwestern ponderosa pine forests which are characterized by large numbers of small diameter trees, ponderosa pine forests of the past were open stands mostly composed of small numbers of large diameter trees.

The historical fire regime is one aspect of the ponderosa pine reference condition that can be reconstructed very well specifically for New Mexican forests, and the forests of the Jemez Mountains. Prior to the 1860's, Utes, Apaches and Navajos using the Jemez Mountains area were a deterrent to settlement by Euro-Americans (Allen 2004). Shortly afterward in the 1870s and 1880s livestock were introduced in large numbers into what is now Bandelier National Monument, at very high stocking rates (Allen 2004). The grazing pressure diminished fine herbaceous fuels, curtailing the spread of fire and the size of fires. The previously high-frequency fires were further curtailed by the adoption of fire suppression policies in the early 20th century. Long-term fire scar records confirm that this omission of fire since the late 1880s in the Jemez Mountains is an anomaly. This fire regime can be tracked over periods of centuries rather than just a window of time just before the 20th century. Allen et al. (1995) and Touchan et al. (1996), documented the last 500 years of burn scars, and provide strong evidence of a near complete cessation of fire in around the turn of the 20th century. In the 400 years prior to the 20th century, ponderosa pine forests experienced frequent fires, most of which were low intensity ground fires. The fire return interval was about 5 – 15 years. Working in El Malpais, also in New Mexico, Grissino-Mayer and Swetnam (1997) documented an even more frequent fire regime. These researchers also note a decrease in fire frequency coinciding with the introduction of large cattle herds around 1880, and a second decrease around 1940 associated with adoption of fire suppression policies in that area. Kaye and Swetnam (1999) confirm a similar fire regime at a third New Mexican ponderosa pine site in the Sacramento Mountains.

Staff at Bandelier National Monument have developed a specific set of *desired future conditions* (Bandelier National Monument 2006). These desired conditions strongly reflect the above characteristics of ponderosa pine forests in the state they were likely in approximately 1860 – 1870. They include *Pinus ponderosa* as the dominant overstory species. Most forests would be expected to be open, and cover of mature trees might be as low as 5%. Closed canopy stands might occur, particularly in situations such as higher elevations or in canyons. A mix of age classes spanning hundreds of years would also be expected. A discontinuous canopy and discontinuous ladder fuels would restrict the frequent fires primarily to the ground surface, where they would periodically (every 5-15 years) consume herbaceous understory and litter accumulations. The summary of desired conditions specifically allows for patchy crown fires to maintain heterogeneity.

Methods and Data

Vegetation Maps

Two vegetation maps have been produced in recent decades encompassing most or all of Bandelier National Monument. Allen (1989) produced a map based upon aerial imagery from 1981. Muldavin et al. (2011) produced a map using supervised automated digital processing, assisted by multispectral imaging. This map was confined to the entire extent of Bandelier National Monument, and was based upon 2004 imagery. To facilitate comparisons among the two maps, both were reduced to their intersection. Since a map can be simplified but not made more complex without new data, we focused on reclassifying the more detailed Level 2 map units of the Muldavin et al. (2011) to the system used by Allen (1990) to enable a direct comparison of the two (*see Chapter 1 for details*).

Analysis

Within the boundaries of the 1996 Dome Fire and 1997 Lummis Fire, we wished to determine the fate of several major community types either dominated by, co-dominated by or presumed to be formerly dominated by *Pinus ponderosa*, found within the burn boundary (GRAS, GRSH, PP, PPPJ) in 1981. To do so, we clipped both vegetation maps to the intersection of Bandelier National Monument and the two fires, and converted both to rasters with 10m grid size. We determined the proportional coverage of different 2004 vegetation community types within the areal extent of each of the three major 1981 vegetation types in this extent.

Post-Dome fire transects

Sampling Design

After the 1996 Dome fire, an emergency seeding operation was initiated on the Santa Fe National Forest portion of the burn, but not in the Bandelier National Monument portions. Within the National Forest, one small area was left unseeded. In 1997, a year after the Dome fire, 49 50m long transects were established in and around the wildfire to monitor vegetation changes. Ten of the transects were within Bandelier National Monument, and the others were in the National Forest. The sampling took advantage of the mosaic of different burn intensities experienced, and to the greatest extent possible contrasts among seeded and unseeded areas. Since this was a monitoring study opportunistically initiated after an unplanned event, these factors are imperfectly crossed and may suffer from spatial autocorrelation. Nonetheless, similar datasets are rare, making this one quite valuable. Twenty two transects were established in the moderate burn intensity areas, 20 in the high burn intensity, one in the low burn intensity and six in unburned areas. Because of the lack of replication of the low burn intensity, we excluded this level from our analysis. At each transect, a line-intercept method was used to measure herbaceous vegetation. A 20 x 50m plot, centered around each transect, was created to score tree cover data. All trees inside the plots were recorded to species. For more information on experimental design, see Barclay et al (2004). All plots were measured in 1997 and 1998, Bandelier plots were remeasured in 2002, and National Forest plots were remeasured in 2008.

Analysis

We used multiple approaches to examine the effects of fire intensity, seeding and year on vegetative composition. In addition to these three explanatory factors, which were included in all statistical models described below, we also included spatial coordinates to control for spatial heterogeneity in vegetation characteristics. Even though data was initially collected on both live and dead cover, we focused all of our analyses on the live portion of the community as this is portion of the community that will show how the community has changed due to the factors described above, as well as indicating the future trajectory of the vegetation community. In some analyses, we analyzed total live vegetative cover and in other analyses, we analyzed percent cover or each functional group. For these latter analyses, we divided the cover of each functional group by the sum of all functional groups (forbs, grasses, trees, shrubs and sub-shrubs). Analyzing the percent cover of each functional group will reveal if there is differential increase or decrease of certain functional groups with time. For example, if we see a decrease of trees after fire, this could be because all vegetation decreases after the fire and trees are just responding like other functional groups. Or, it could be because trees are decreasing more than other groups, signifying a compositional shift in the vegetative community. Analyzing percent cover will distinguish between these two scenarios. For each analysis described below, we specify if total or percent cover was used.

We first used perMANOVA (Anderson 2001) to statistically test for differences in the overall composition of functional groups. Functional groups were categorized as forbs, grasses, shrubs, subshrubs and trees, a system commonly used in the National Monument. In our perMANOVA, we included data on both basal and canopy cover. To run the perMANOVA, we used the adonis function in the vegan package of R (Oksanen et al 2008), specifying Bray-Curtis distance with 10,000 permutations.

Next, we used non-metric multi-dimensional scaling (NMDS; McCune and Grace 2002) to graph functional group composition across fire intensities (high, moderate and unburned) and seeding (seeded vs. not-seeded). With the goal of complimenting each other, we used the same community matrix to construct our NMDS plot as were used in the perMANOVA (*see above*).

In addition to looking at the overall composition, we examined different components of the plant community and how they may have changed across burn intensities, years and seeding. We used ANOVA to look for differences in canopy cover of individual functional groups (forbs, grasses, shrubs, trees and subshrubs), species richness, total live vegetation canopy cover and canopy cover of two exotic species, *Lolium multiflorum* and *Bromus inermis*. Note, in the data sets we used, the data on functional groups were divided between canopy and basal cover as well as live and dead. Finally, based on discussion with local scientists, we investigated the consequences of using various configurations of canopy and basal cover data (e.g. the sum of basal and canopy cover, interpolation of non-zero basal values when canopy values were zero). Canopy cover was routinely one to two orders of magnitude greater than basal cover and canopy cover was very strongly related $R^2 > 0.95$ to a combination of canopy and basal cover. Thus, based on this knowledge, we decided that the simplest and most informative cover data to use in most of our analyses of these datasets was live canopy cover. For data on individual species and total vegetative cover (which was the sum of all the species), we used another dataset that did not distinguish between canopy and basal cover or live and dead cover. Thus, we simply analyzed “cover” in this case.

Finally, to better understand the interactions among various functional groups within the vegetative community, and their responses to fire and seeding, we also constructed a structural equation model. We restricted this analysis to only samples from the Santa Fe National Forest, because there were only 10 observations in Bandelier

National Monument and the time of sampling was disjunct from the Santa Fe National Forest Samples. Using only National Forest samples enabled us enough sample size to build a model spanning 1997 – 2008. Prior to modeling we developed four groupings of vegetation: 1. Seeded herbs, consisting of all species included in seed mixes (most prominently *Lolium multiflorum*, *Bromus carinatus*, and *Elymus trachycaulum*); none of these species were common in unburned areas and thus were easily tied to seeding activity, 2. *Bromus inermis*; an exotic perennial, rhizomatous species of concern that is becoming more prevalent, and can spread vegetatively to create local monocultures; was a minor contaminant in post-Dome fire seed mixes, but had been previously introduced to close timber and mining access roads prior to the fires, 3. Unseeded native herbs, all native grasses and forbs that were not known to be present in seed mixes, 4. Native shrubs and sub-shrubs. Tree cover was trivial except in unburned plots, thus this component was not considered in our model. We compiled live canopy cover data for all of these groups. To improve linear fit among variables, we applied a log transformation.

Structural equation modeling begins with an *a priori* model. Our model stated: 1. Burn intensity will influence all vegetative components, 2. The seeding treatment will influence abundance of seeded species. 3. Because seeded species are intentionally introduced early, their presence will influence all other vegetation components, 4. *Bromus inermis* is able to proliferate faster than most unseeded native herbs and shrubs, therefore its abundance will influence those vegetation components. 5. Finally, herbaceous native plants tend to establish prior to woody species, thus their abundance can be expected to influence the abundance of woody plants.

Based on conversations with local scientists including Craig Allen and Brian Jacobs, successional dynamics after fire were also regulated by other factors such as location in space, elevation, aspect, and whether or not the pre-fire forest type was ponderosa pine or ponderosa pine-mixed conifer. Thus we screened all of our response variables in each year to determine if any were related to these influences. The answer differed based on year and the vegetation component considered, but we found that northing, easting, elevation, and pre-fire canopy all potentially had effects on the abundance of at least some vegetation components. To purge these effects from our analysis we fit each vegetation component in each year as a function of northing, easting, elevation, and pre-fire forest type, then saved the residuals. The residuals were substituted for the original variables and analyzed in the structural equation models.

We used a modeling approach called multigroup modeling, with the "groups" being different years of observation. This approach allows the modeler not only to estimate the relationships between different variables, but also to observe how these relationships change through time. Prior to engaging in a multigroup model building protocol, it is common to fit the *a priori* model to each group of data and apply a goodness of fit test to ensure that it is a reasonable model structure for each time point. If the model structure does not fit one or more groups, the likelihood of fit of the multigroup model is reduced. In doing so, we discovered one post-hoc alteration that needed to be made. In 2008, the abundance of *Bromus inermis* was too strongly correlated to the seeding treatment, despite that it was virtually unrelated to the abundance of seeded species. This suggested a different mechanism than facilitation by seeded species, perhaps reflecting that it was a contaminant in seed mixes (up to 3% of high elevation seed mixes were *Bromus inermis*). Thus we added a direct relationship from the seeding treatment to *Bromus inermis*. Applying the same logic, we also included a path from seeding to unseeded native herbs for consistency.

A multigroup model starts from the position that all model parameters are the same among groups. For example, this means that the influence of seeding on seeded species abundance is of the same magnitude no matter the time period. This initial model usually fits poorly because different patterns do occur in different time periods. The modeler’s goal is to iteratively relax constraints that force parameters to be the same until a satisfactory fit is obtained. This is accomplished by viewing residuals for each parameter. The parameter with the greatest residual is identified as one that should be allowed to be different among time periods. We used Akaike’s Information Criterion to determine the best number of parameters to relax. We relaxed six constraints, and obtained a satisfactory model that also fit very well based on absolute fit criteria.

Fire Monitoring Plots (FMP)

Sampling Design

The fire monitoring plots were established to examine shifts in vegetation characteristics after prescribed burns, mechanical thinning, and wildfire. Though they span multiple ecosystems, we focused on plots representative of various types of ponderosa pine stands, or former ponderosa pine stands. We used plots established in four areas, defined as: upper elevation ponderosa, low elevation ponderosa, previous ponderosa (area of La Mesa fire) and the area of mechanical thinning. Most plots were surveyed multiple times across several years to monitor the effects of thinning or burning through time. Burned plots were surveyed before the burn (i.e. in an undisturbed state), immediately after the burn, at one year, two years, five years and 10 years after the burn. Plots were sometimes burned multiple times, in which case the cycle of monitoring was reset. For example, in one case, a plot was burned, monitored up until 9 years after the burn when the plot was burned again. At that point, the plot was surveyed at all time points mentioned above, relative to the second burn. In this case, because the plot was surveyed before the second burn, which happened to be 9 years after the first burn, there is also data for 9 years after the first burn. Similar to burned plots, thinned plots were surveyed immediately before (undisturbed) and after thinning. While thinned plots were not surveyed in years following thinning (like the burned plots), there were some instances that inadvertently surveyed plots several years after thinning. A few plots were thinned a second time. In these cases, plots were surveyed immediately before the second thinning, which corresponded to seven years after the first burn. Finally, some plots were left in an undisturbed state (i.e. they were neither burned nor thinned). These plots were usually surveyed once, but in some cases were surveyed twice. The effects of prescribed fire on Bandelier's vegetation communities are monitored through permanent monitoring plots. All of the prescribed fire monitoring plots installed in Bandelier to date are 20 x 50 meter (m) macro plots that contain various sub-plots and transects of different dimensions (Fig. 16). These vegetation plots are randomly located and installed in areas designated for prescribed fire and data is collected before the burn is implemented. The first visit to a plot is recorded as pre-burn. Subsequent visits are immediate post-burn, 1 year, 2 year, 5 year, and 10 years post-burn. Plots are then read at 10 year intervals after the 10 year post-burn visit.

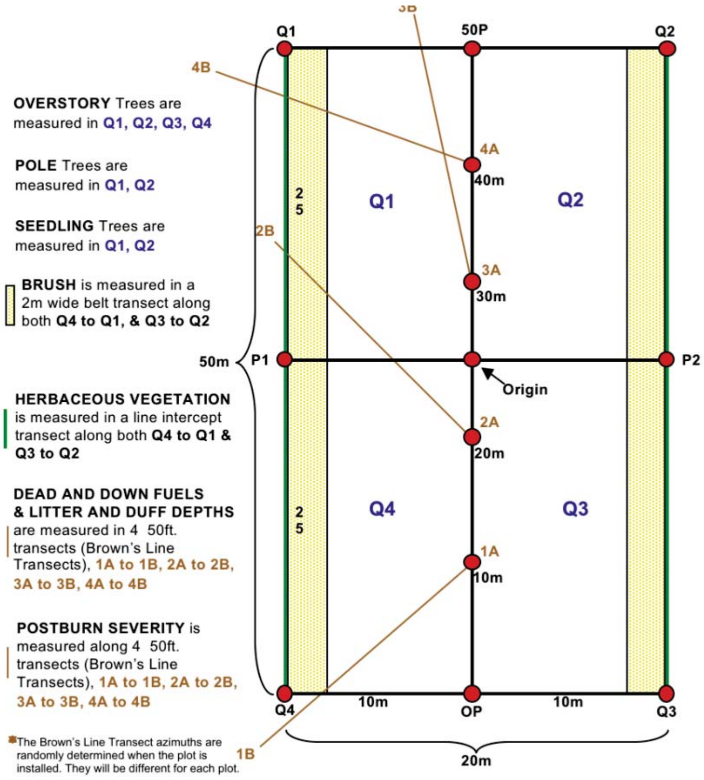

Figure 16. Structure of plot design used for all prescribed fire montoring plots in Bandelier National Monument. The plot is 20 × 50m and is divided evenly into 4 quadrats. The protocols for surveying overstory trees, pole trees, shrubs and understory vegetation are summarized above.

Vegetation data was collected for overstory trees (defined as trees with diameter at breast height (dbh) greater than 15cm), pole trees (with dbh greater than 2.5 cm, and less than 15cm), shrubs and understory vegetation (which includes exotic and live forbs and grasses). Overstory trees were recorded in the entire plot (all four quarters), while pole trees were recorded only in quarters 1 and 2. In both cases, values were extrapolated to provide a metric of tree density (trees per acre). Shrubs were surveyed in a 50m long belt transect with a width of 2 or 5m, depending on the survey. Like tree data, values for shrub counts were extrapolated to provide a density metric (number of shrubs per acre). Finally, herbaceous vegetation (native grasses and forbs and total herbaceous exotics) was scored

Using 2, 50 m point-line intercept transects along the exterior of a plot using a two meter sampling pole. The genus and species of the plant touching highest on the pole is first recorded, along with the height. Any additional species touching the sampling pole are also recorded, but their height is not recorded. Whether the plant is live or dead is also recorded at each point. This is done at 0.3m (30 cm) intervals for the entire length of both 50 m transects, equaling 166 points along each transect, or 332 points per plot. Herbaceous vegetation was recorded as total per plot.

In our analysis, there were 58 plots, several of which were surveyed multiple times, after thinning and burning, for a total of 344 surveys. It is important to note that trees were scored in all surveys. However, shrubs were scored in only a subset of surveys. Specifically, shrubs were not surveyed in the thinned plots and were surveyed in only a subset of the burned plots. Further understory plants were scored in separate subset of surveys. Ideally we would have lumped all data sets to provide a more comprehensive picture of vegetation change. If we had reduced the dataset to the samples in which all types of data were available, it would have resulted in a ~50% reduction in the data and an unacceptable loss of statistical power. Thus, because of the inconsistent sampling of vegetative groups across plots, we analyzed trees, shrubs and understory plants separately, using different plots as needed.

Analysis

This dataset was not established like a typical planned replicated experiment. Rather, it was meant to inform managers about the results of specific management actions. As such, it required some creativity to perform a statistical analysis. Because of the different types of disturbance (fire and thinning) and the number of times the disturbance was manipulated, we created an explanatory factor “disturbance type” which included following levels: undisturbed, burned once, burned twice, thrice-burned and thinned once. We also created a factor “years since disturbance” which examined how vegetation characteristics may change with time after fire or thinning. Because not all plots were disturbed in the same calendar year, we included “year” as another explanatory variable to account for overall interannual trends in vegetation composition. Finally, because vegetation is known to vary across space, we included northing and easting coordinates as covariates in our statistical model.

We used the same statistical models to analyze trees and shrubs and a slightly different model to analyze herbaceous understory vegetation. For these former data sets, we used ANOVA (JMP Pro 9; SAS Institute Inc.; Carey, NC), including the above-described explanatory factors (year, years since disturbance, disturbance type and northing as easting as covariates) to explain variation in total tree overstory cover, total tree pole cover and total shrub density. Because this was not a planned, replicated experiment, we did not always have crossed factors. Therefore we conducted a main effects model and omitted interaction terms. Because of the nature of data, where one plot was often measured several times across years, a repeated measures design could have been used. However, this approach would have omitted nearly half of the data (e.g. we could not include any undisturbed plots) which we considered an unacceptable cost. The cost of not accounting for repeated measures might be possible inflation of power to detect effects of disturbance type. Because several undisturbed plots were only measured once and because disturbed plots were not always regularly surveyed across years, we chose to use a standard ANOVA approach. Finally, because of singularity issues (i.e. values of one explanatory variable could be explained completely by a linear combination of other explanatory variables), we could not include all explanatory variables in the model for understory herbaceous vegetation. Instead, we ran a model with four of the five explanatory variables, extracted the residuals and used a model that used the fifth explanatory variable to explain variation in the residuals. This was repeated, in turn, for the three explanatory variables (i.e. we did not do this for the spatial covariates). The purpose of this analysis was to determine how much variation in the understory data could be explained by each factor and could not be attributed to any other factor.

Condition and Trend

Vegetation change specifically in the ponderosa pine zone of Bandelier National Monument is largely driven by fire and associated invasive species impacts, and drought and associated pathogen outbreaks. After about a century of fire suppression and relatively few fires, large and often intense fires are strongly impacting the ponderosa pine forests of Bandelier National Monument (Fig. 2). The 1977 La Mesa fire exerted major impacts to the distribution of ponderosa pine forests. Our earliest glimpse of the spatial distribution of vegetation types was a vegetation map created from imagery after this fire. Within Bandelier National Monument, nearly all of the area that experienced at least moderate severity burns, are mapped as a vegetation type other that ponderosa pine forests, usually grasslands or shrublands. Since it is known that most of the burned areas were ponderosa pine forests, this observation suggests a fire-triggered type conversion (Fig. 2a). Most of these areas have still not reverted back to ponderosa pine forests based on more recent 2004 mapping. The 1996 Dome fire impacted ponderosa pine forests, but mostly outside of the Bandelier National Monument boundary on the National Forest side. Within the park, a sizable block (~ 1km^2) of contiguous ponderosa pine forests experienced moderate to high intensity fire west of Capulin Canyon (Fig. 2b). The 1997 Lummis fire was almost entirely in ponderosa pine habitat, including some areas previously converted during the La Mesa fire (Fig. 2b). This fire occurred in the largest remaining contiguous block of ponderosa pine forest remaining in Bandelier, and its perimeter encompassed more than half of that block. The Las Conchas fire was a much larger fire, but its impacts on ponderosa pine forests within Bandelier were smaller, perhaps because of the fuel consumption effects of other recent fires (Fig. 2d). On adjacent National Forest lands, many ponderosa pine and former ponderosa pine stands that were impacted by the Dome fire reburned in Las Conchas. Because no vegetation mapping was conducted prior to the 1977 La Mesa fire, it is difficult to know how much of Bandelier's ponderosa pine dominated areas have been lost. Since that time however, perhaps a quarter of remaining ponderosa pine forests have experienced fire. Some of these larger crown fire runs are clearly outside of desired conditions, as they can span multiple km^2. We are able to estimate the trajectory of vegetation change by observing the community transformations after fires (Figs. 17, 18). The Dome and Lummis fires burned in areas currently or probably formerly dominated or co-dominated by *Pinus ponderosa*, in addition to areas dominated by Pinyon-Juniper ecosystems (addressed in Chapter 4). There was a large grassland opening within the boundary of the Lummis fire which was likely created by a high intensity fire in Ponderosa pine forests during the 1977 La Mesa fire (Fig. 18). This area, having experienced another burn 20 years later has still not recovered a *Pinus ponderosa* overstory. Rather these areas which were

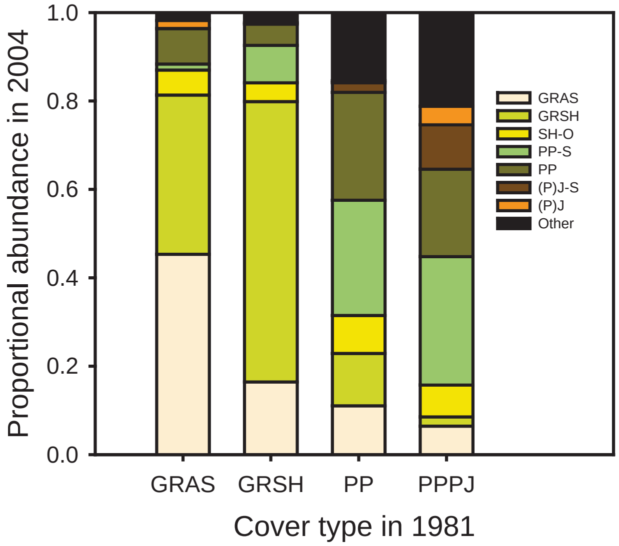

Figure 17. Fate of three key 1981 cover types (Allen 1990) within the Dome (1996) and Lummis fire (1997) perimeters, based on the cover types mapped by Muldavin (2011) based on 2004 imagery.

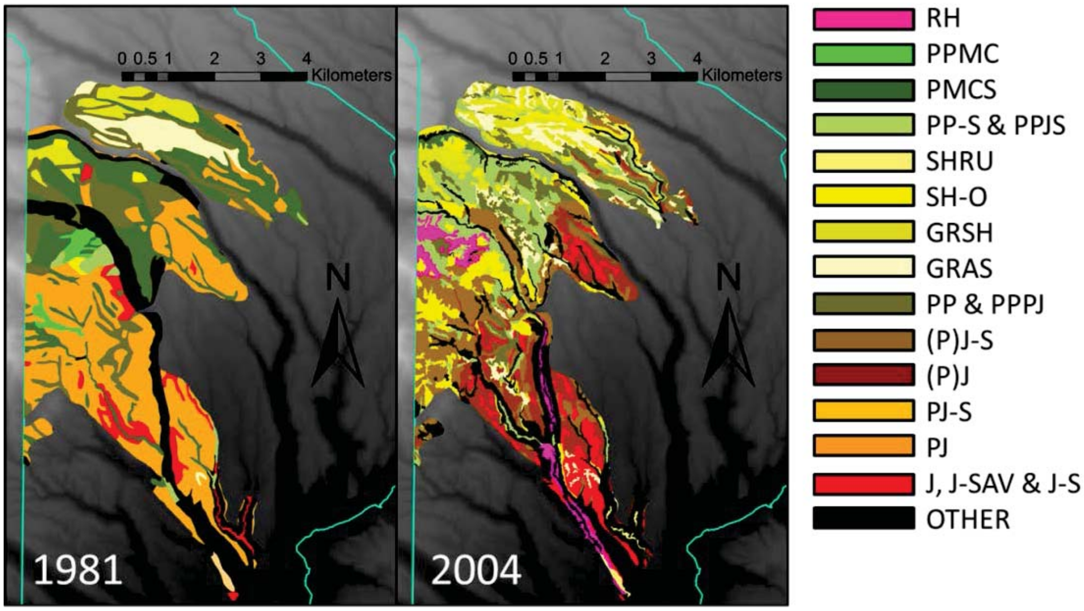

Figure 18. Vegetation communities within the Dome (1996)and Lummis (1997) fire perimeters as mapped based on 1981 imagery (Allen 1990) and 2004 imagery (Muldavin 2011).

grasslands in 1981, by 2004 appear to increasing be fragmented by the development of mixed shrub-grass communities. This indicates an increasing preponderance of shrubs. Interestingly, the 1981 mixed grass-shrub communities largely remain mixed grass-shrub after the 1996-7 fires, suggesting a community type that is stable under repeated burning (Fig, 17). In stark contrast, pure *Pinus ponderosa* forests appear highly likely to convert to another vegetation type following the 1996-7 fires. Only about 25% remain pure ponderosa pine stands in 2004 (Fig. 17). These stands are fairly likely to convert to grasslands or shrublands, but the single most common conversion is to retain a *Pinus ponderosa* overstory with a shrub understory. Once again, these conversions suggest a proliferation of shrubs. Mixed ponderosa pine-piñon-juniper communities underwent a variety of changes. *Pinus edulis*, as discussed in other chapters, was essentially eliminated as a dominant species by drought and insect outbreak. Thus, several transitions are brought about by the subtraction of this species. However, nearly half of these ecosystems either became shrub-dominated or retained trees with a shrub understory. In summary, ponderosa pine forests with herbaceous understories have been greatly reduced in coverage on the landscape, and are becoming increasingly fragmented. Grasslands may initially be promoted by fire but eventually become fragmented. Shrubs appear to be emerging as either dominant species or major community components (along with grasses). Shrublands which resprout after fire, or resprouting shrub-grass associations are a common community type in various parts of the US and the world. It seems likely that increasing fire frequency would perpetuate this community type and possibly prohibit recruitment of *Pinus ponderosa*.

In the post-Dome fire transects and the FMP datasets we have some data which may shed some light about the successional mechanisms that lead to grass or shrub dominance after a fire in a ponderosa pine dominated ecosystem. This data also highlights some of the roles of exotic grasses in vegetation change.

Post-Dome fire monitoring transects - After accounting for spatial position (i.e. UTM coordinates), all three explanatory factors (year, fire intensity and seeding) explained a significant amount of variation in functional group composition of plant communities (Fig. 19).

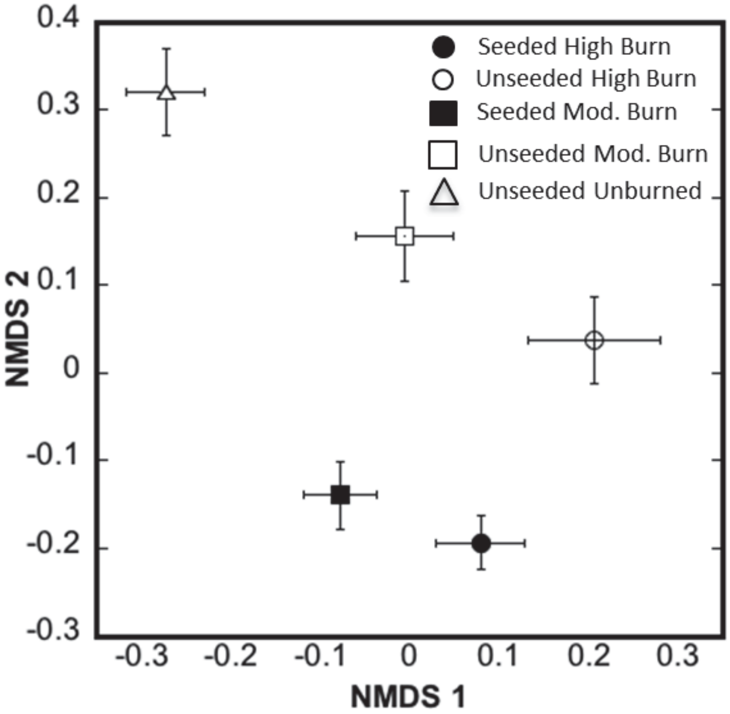

Figure 19. An NMDS plot showing differences in functional group composition across levels of fire intensity and seeding. Points and error bars represent mean NMDS score +/- 1SE. Points that are close together represent functional group composition that is relatively similar while points that are far apart represent functional group composition that is relatively different.

Consistent with the perMANOVA, the NMDS shows separation among burn intensities and between seeding treatments (Fig. 20). The results from the perMANOVA and NMDS provide evidence that the overall structure of vegetative functional groups is changing in response to wildfire, seeding and annual variation. In conjunction with the NMDS, we used a vector analysis to explore how certain functional groups may be changing with seeding and burn intensities.

In analyzing components of the community that changed with fire, we found a trend ($0.05 < p \leq 0.10$) suggesting a difference in species richness (Fig. 20), such that richness was highest at moderate burn intensities. We also found a trend suggesting total vegetative cover (Fig. 20) differed across burn intensities, such that cover was highest in unburned plots and lowest at high burn intensities (Fig. 20A).

In examining canopy cover of individual functional groups, we found significant differences ($p \leq 0.05$) in the cover of forbs, shrubs and trees across burn intensities. In conducting a post-hoc analysis (Tukey's HSD), we did not find differences between the high and moderate burn intensities, but between unburned and at least the high or moderate burn intensities. Specifically, we found a 50% decrease in forbs, a 200-235% increase in shrubs and a 89-93% decrease in tree cover with fire (Fig. 20B). Interestingly the sharp decrease in tree cover coupled with a greater than 2-fold increase in shrub cover, suggests catastrophic wildfire changes the vegetation community from a forest-dominated landscape to a shrubland community (Fig. 20B).

In the following analyses and graphs, we include a temporal component to show the effects of burning and seeding change with time. Species richness was not affected by seeding or year and was only moderately affected by burn intensity (Fig. 22A), where species richness was the lowest in early years, in unseeded, high burn intensity areas. However, by 2008, there is no apparent difference in species richness across levels of any of the explanatory variables (Fig. 22A).

Total vegetation cover was not influenced by seeding or burn intensity (Fig. 22B). This is somewhat surprising since fire should reduce vegetative cover, while seeding should increase it. We did find that year influenced cover (Fig. 22B), such that cover seemed to have dramatically increased in 2008 compared to previous years (1997, 1998 and 2002).

There were both surprising and unsurprising patterns with forb cover in response to fire, seeding and time. Forbs were highest in the first two years after wildfires, then decreased with time. This is not surprising, given that forbs are an early successional functional group and are known to proliferate after fires or other disturbance (Laughlin et al. 2004). However, it was somewhat surprising that this happened only in the unseeded plots. In other words, in seeded plots, we did not see a substantial increase in forb cover after the fires, suggesting that seeded graminoid species inhibit forb recruitment after fire (Fig 21C). This could be important in terms of erosion as another analysis (see SCPN analysis in Chapter 4) suggested forb cover improves aggregate soil stability.

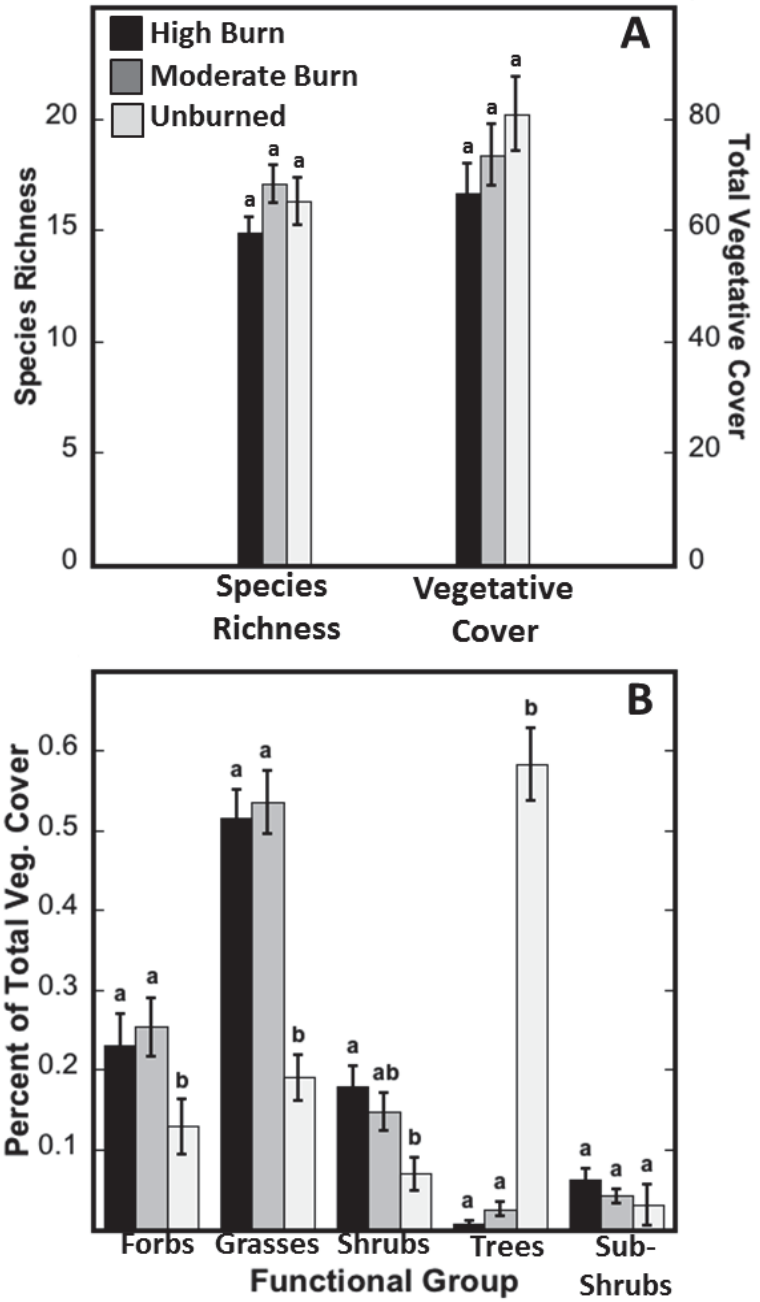

Figure 20. Bar graphs showing how species richness, total vegetative cover (A) and cover of individual functional groups (B) differed across burn intensities. Bar heights and error bars represent mean values +/- 1SE.

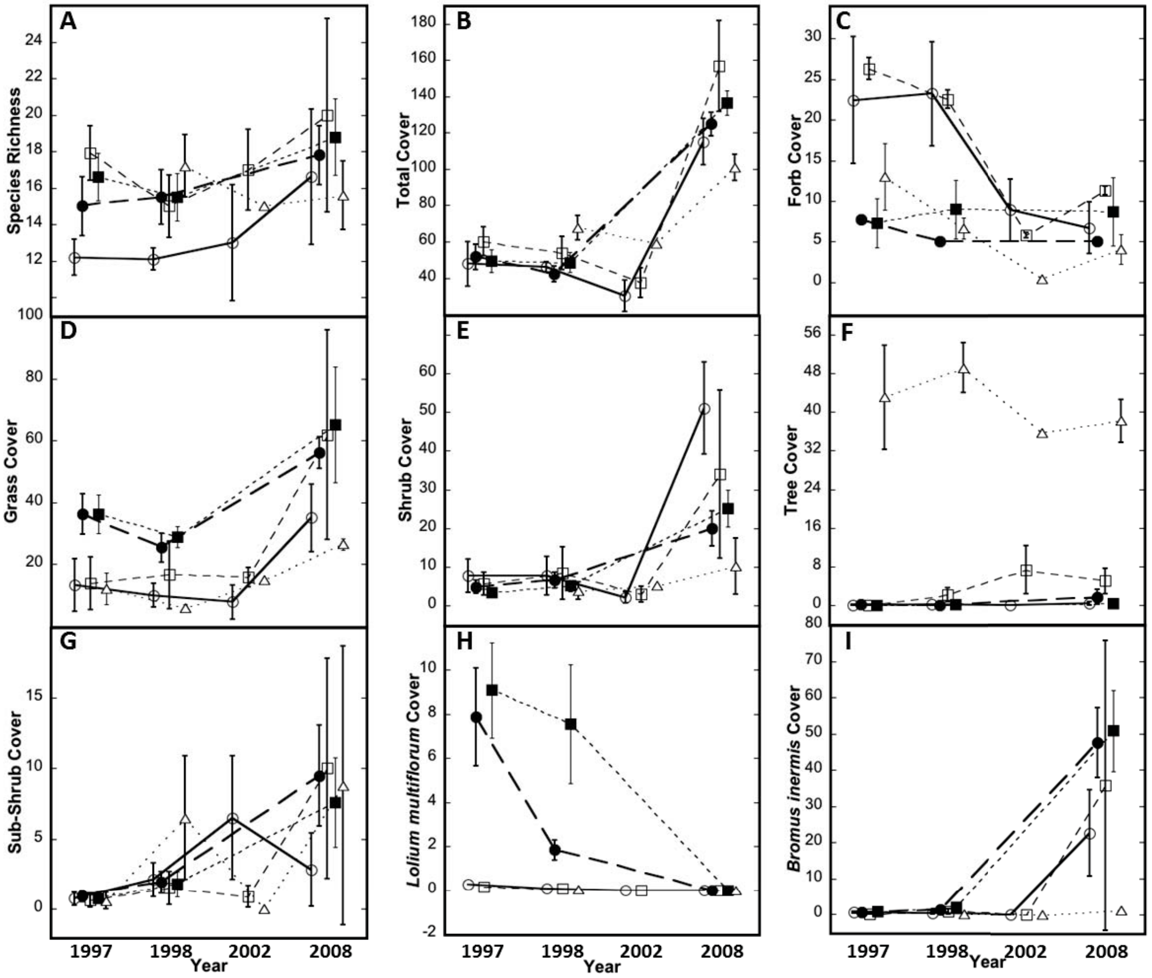

Figure 21. Plots showing the influence of year, seeding and burn intensity on species richness (A), total vegetative cover (B), forb cover (C), grass cover (D), shrub cover (E), tree cover (F), sub-shrub cover (G), Lolium multiflorum cover (H) and Bromus inermis cover (I). Solid shapes represent seeded plots and hollow shapes represent unseeded plots. The different shapes represent different burn intensities, where circles are high intensity, squares are moderate intensity and triangles are unburn plots. Point and error bars represent means and one standard error.

Grass cover was influenced by seeding and year, but not burn intensity (Fig. 21D). Because the seeding treatment was grass seed, it is not surprising that grass cover was higher in the seeding treatment. Interestingly, grass cover increased in 2008. This was somewhat surprising because the species that comprises the bulk of the seeding mixes, *Lolium multiflorum* decreases with time (*see below*). The increase in grass cover seems to be driven by the sharp increase of another exotic, *Bromus inermis* (*see below*).

Shrubs dramatically increased with time and that increase seemed to be influenced by both seeding and fire (Fig. 21E). In 2008, shrub cover increased to nearly 50% in some plots. Specifically, plots that were not seeded but experienced wild fire, had the highest cover of shrubs, providing more evidence that seeding (as currently practiced) may disrupt natural recovery processes. Plots that burned and that were seeded had moderate levels of shrubs, while the unburned and unseeded plots had the lowest shrub cover. Interestingly, there does not appear to be an increase in shrub cover in the unseeded and unburned areas. Again, these results suggest that catastrophic wildfire may facilitate the conversion of forest to shrubland in Bandelier.

Tree cover also varied across burn intensities. Not surprisingly, tree cover was highest in unburned plots and near zero in burned plots. What was surprising was that tree cover seemed to increase in unseeded, moderate burn intensities, but stay near zero in seeded, moderate burn intensity plots, suggesting that seeding may inhibit regeneration of trees. This could be problematic for erosion as canopy cover was seen as the most predictive factor promoting soil stability (see SCPN analysis, Chapter 4). It is also problematic for the eventual replacement of ponderosa pine.

Lolium multiflorum, which is the major component of the seed mixes and an exotic species, rapidly declines to zero abundances in 2008, ~10 years after seeding (Fig. 21H). This decline is positive because this species is not a desired component of the post-fire community, rather it is used as an erosion reduction tool. Conversely, we see a sharp increase in a different exotic species, *B. inermis*. In 1997 and 1998 *B. inermis* was nearly non-existent, but in 2008, the exotic composed ~50% of the vegetative cover. Interesting, the increase of this exotic seems to be associated with human perturbation where plots that experienced both wildfire and seeding had the highest cover of *B. inermis*, while unseeded and unburned plots had almost no *B. inermis* (Fig. 21I).

Our SEM of the post-Dome fire transects is useful in elucidating interactions among different plant functional groups, and determining which groups alter successional trajectories (Fig. 22). In 1997, burn intensity actually had few strong effects on vegetation, except for a minor positive effect on seeded species (Fig. 22a). The seeding treatment very clearly influenced the abundance of seeded herbs, mostly *Lolium multiflorum*. The increase of seeded herbs had several interesting effects on other vegetation components. *Bromus inermis* was strongly promoted by the abundance of seeded herbs, and unseeded native herbs were strongly suppressed by seeded herbs. *Bromus inermis* also exerted an unexpected positive influence on native unseeded herbs, possibly suggesting early facilitation while *Bromus inermis* cover was still low.

In 1998, the majority of patterns were essentially the same, with the exception of the disappearance of the positive effect of *Bromus inermis* on unseeded native herbs (Fig. 23b). In fact this influence became weakly negative.

10 years later in 2008, many successional dynamics are considerably different (Fig. 22c). As *Lolium multiflorum* disappears from the community, the influence of seeded herbs on other vegetative components diminishes, most notably its effect on *Bromus inermis* (strongly positive in 1998, weakly negative in 2008). Interestingly, in 2008 there is an exceptionally strong influence of the seeding treatment on *Bromus inermis* abundance. What does this mean? Initially we altered our *a priori* model to accommodate this path, based on the possibility that *Bromus inermis* was a contaminant of the seed mix. However, this hypothesis is not supported by the results from 1997. In 1997, the seeding treatment mostly affected *Bromus inermis via* facilitation by seeded herbs. Since *Bromus inermis* is a seed contaminant we would expect a positive direct path from the seeding treatment to *Bromus inermis* abundance. This path is actually negative, a phenomenon which we cannot explain.

We suspect that in 2008, *Bromus inermis* is more abundant in seeded areas because it was facilitated by seeded herbs in prior years. Because it is already well established, its abundance is poorly related with abundance of seeded herbs in 2008.

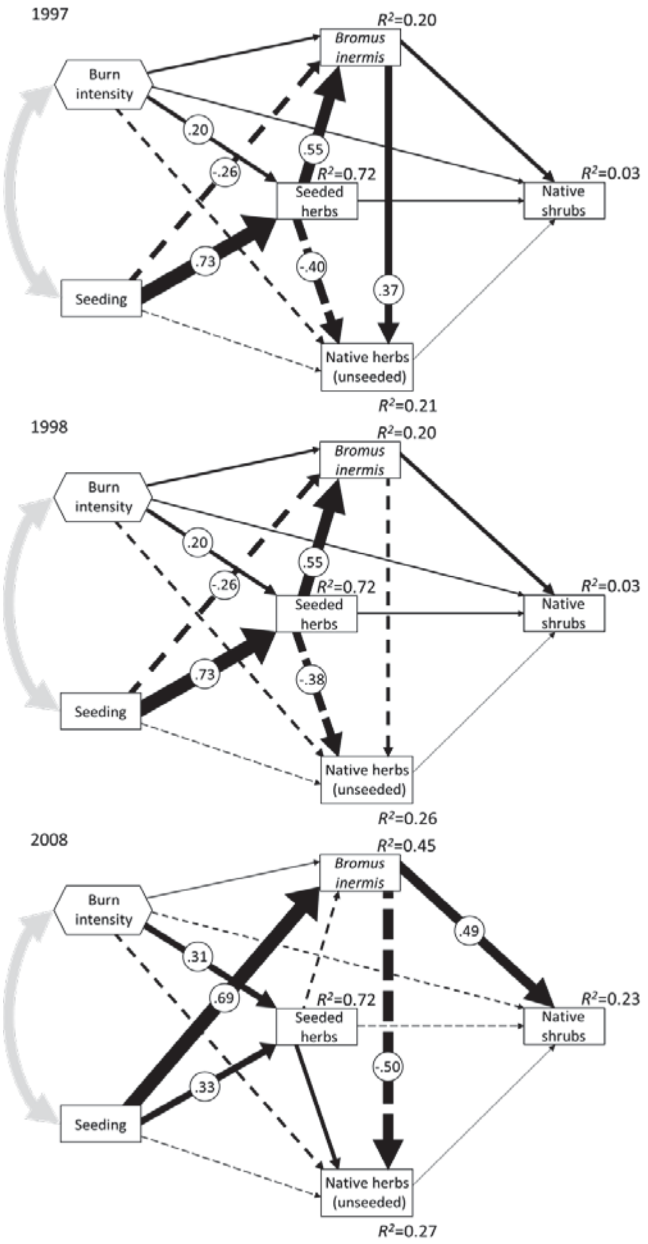

Figure 22. Structural equation models depicting the influence of seeding and burn intensity on post-fire succession, in addition to the interrelationships among various components of the vegetative community. Rectangles represent measured continuous variables. The hexagon represents an ordinal variable, burn intensity. Single headed arrows indicate a hypothesized influence that one variable exerts upon another. Double headed arrows represents an unresolved correlation. Width of arrows, and associated path coefficients (shown in circles when absolute value is greater than 0.20). Solid arrows represent positive relationships among pair of variables; dashed arrows represent negative relationships.

The facilitation of *Bromus inermis* by seeded herbs appears to trigger a successional cascade later. Although *Bromus inermis* initially in 1997 seemed to be promoting succession back towards an herbaceous (mostly grass) native understory, this effect quickly disappeared and after 10 years as *Bromus inermis* has attained a much higher abundance and exerts a clearly negative influence on native herbaceous plants. Perhaps most interesting is that as native herbaceous plants are suppressed, native shrub species appear to be promoted. Most of the native shrubs are *Quercus gambellii* or *Robinia neomexicana*, both of which attain substantial cover by 2008. These species are significant in that they are both capable of forming thickets, and of resprouting from root crowns after fires.

While the present dataset clearly illustrates the role of seeding in facilitating this successional cascade, *Bromus inermis* invasion is also occurring on burned areas that were never seeded. It remains a minor community component on unburned transects. What may be emerging is a distinct stable state of oak-annual grass savannahs. This is both a fire prone, and fire-resilient community type. Repeated fires could conceivably maintain this community type and retard the recovery of open ponderosa pine savannahs with perennial grass understories.

Fire Monitoring Plots - The three factors of interest (calendar year, years since disturbance and disturbance type) all explained a significant amount of variation in overstory tree density. There was an 85-fold difference in live overstory tree density among years ($F_{18, 305} = 11{,}734.6$, $p < .0001$, $R^2 = .13$). Our analysis included every year from 1993 to 2011 (with the exception of 2005). Pairwise comparisons showed no differences between most pairs of years (Fig. 23C). The significant year effect seems to be driven by the high densities in years 1995 and ’96 and low densities in years 2000, ’03 and ’08 (Fig. 23C). At this point, it is unclear why overstory tree density would vary so much among years. A post-hoc analysis, examining the relationship between the current- and previous year’s Palmer drought severity index (PDSI) showed obvious relationship between the two variables.

Years since disturbance, which included post-burning, post-thinning and undisturbed plots, also explained a significant amount of variation in live overstory tree density ($F_{9, 305} = 5.7552$, $p < .0001$, $R^2 = .08$). Not surprisingly, the undisturbed plots had the highest live overstory tree density (99.2 trees per acre) and hence can serve as a benchmark to measure the effect of thinning and prescribed burning. Immediately following burning (labeled as “0.5 years” since being disturbed), overstory tree density dropped to its second lowest density in the survey (8.7 trees per acre), less than 10% of the density as the undisturbed plots (Fig. 23B). This low density may have been due to a false perception of dead trees following the burn. In other words, if some trees had burned needles, they may not have been included in the survey of live overstory trees. The next year (one year after the burn), overstory tree density was 51.5 trees per acre. It is likely that trees previously thought dead, had leafed out and were again surveyed as live overstory trees. This seems more likely than a ~6-fold increase in overstory tree density from 0.5 to 1.0 years post burn. During subsequent surveys, in years 1, 2, 5 and 10 after burning, overstory tree density increased such that there were no differences among those years or between those years and the undisturbed plots (Fig. 23B). Interestingly, 9 years post burning had the lowest density (1.6 trees per acre) of live overstory trees. These results seem somewhat anomalous, as the following year (10 years

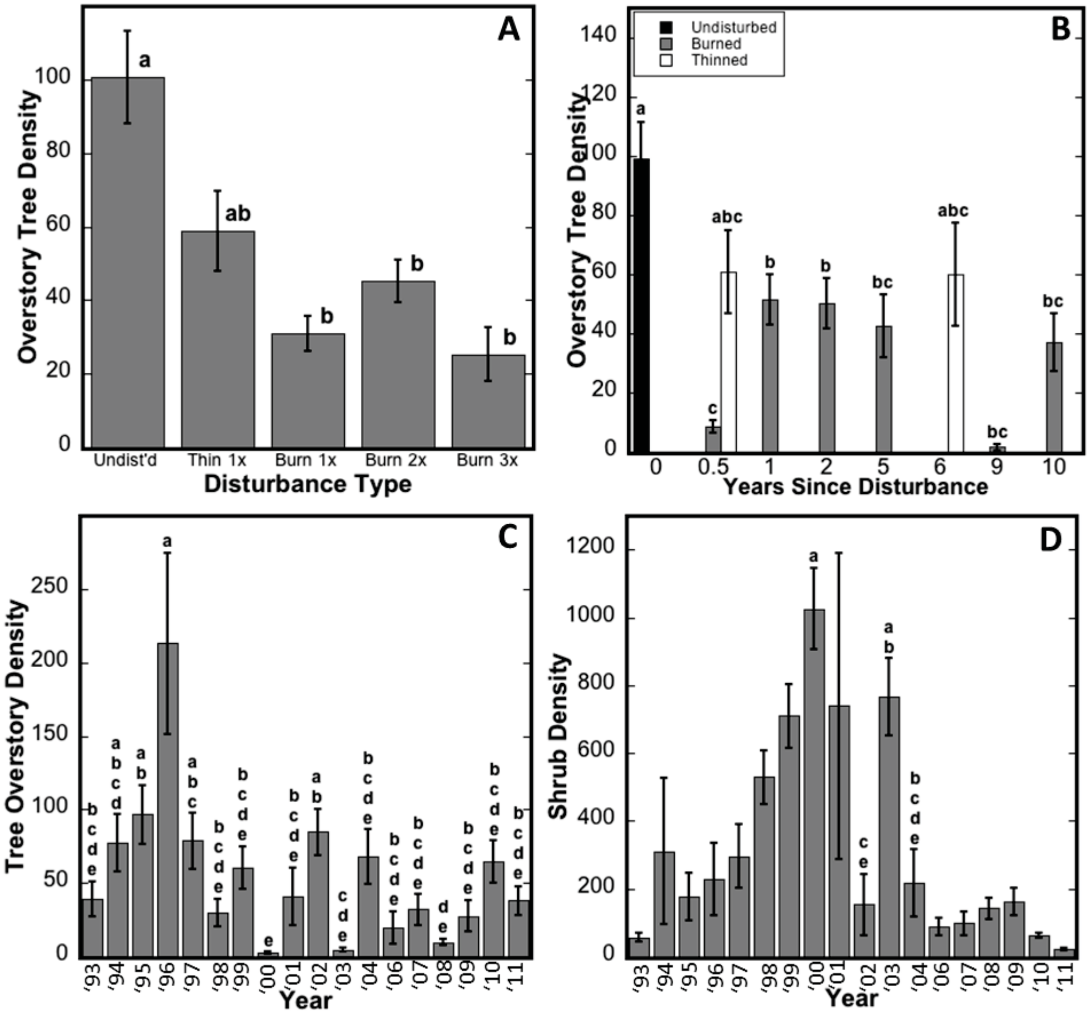

Figure 23. Mean (+/- 1SE) density of overstory trees as a function of disturbance type (A), years since disturbance (B) and calendar year (C). Panel (D) shows mean (+/- 1SE) of shrub density as a function of calendar year.

post burn), live overstory tree density was 37.0 trees per acre. We are not sure why 9 years since burn had such a low density of trees. Perhaps it was due to the confounding effect of year or the particular sites sampled 9 years after burning. All plots measured 9 years post burn happened to be scored in 2008, which had one of the lowest overstory tree densities (see below).

We did not find any differences between the thinned plots and the burned or undisturbed plots. Even though the means were ~40% less (Fig. 23B), there was not a significant difference between undisturbed and thinned plots. Similarly, even though they were consistently lower, there was not a significant difference between the burned and thinned plots (Fig. 23B).

The "disturbance type" factor, which was created to distinguish undisturbed sites, sites that were thinned and sites that were burned once, twice and thrice, explained 3% of the variation in overstory tree density (Fig. 23A; $F_{5, 305} = 4.5329$, $p=.0005$, $R^2 = .03$). In a post-hoc, pair-wise comparison, the undisturbed plots had significantly higher overstory tree density than all of the burned plots, but were not different than the thinned plots (Fig. 23A). We also did not find a difference between the thinned and burned plots or among plots that varied in the number of times they were burned (Fig. 23A).

Of the three explanatory factors of interest, only year explained a significant amount of variation in shrub density. Interestingly, there seemed to be a consistent increase in shrub cover from 1993 to 2000, where shrub cover in 1993 was 56 shrubs per acre and by 2000 shrub density had increased 18-fold, to 1025 shrubs per acre (Fig. 23D). By 2002, shrub density was ~1/7th that of 2000. This drop may have been caused by the 2002 drought (Breshears et al. 2005). With the exception of 2003, shrub density tended to stay relatively low in the years after the 2002 drought (Fig. 23D). In attempting to explain yearly variation in shrub cover, we examined the relationship between Palmer drought severity index (PDSI) and shrub density. We used two models. We regressed shrub density onto the current and previous year's PDSI. In neither case did we find a pattern, suggesting that some other factor besides drought severity, prescribed fire or thinning is influencing shrub density. It is possible that the interaction of multiple variables is driving changes in shrub density, but for whatever reason there seems to have been a pulse of shrub recruitment just prior to the drought.

We found significant effects of year on native forbs and exotic herbs. Exotic herb density seemed to stay relatively similar up until 2001 (the year before the drought of 2002). The year following the drought, exotic density more than doubled and stayed at that level for several years (Fig. 24A), suggesting that climate change is facilitating the introduction and maintenance of exotic species. Year also affected forb density, however, there was no pattern with regards to the 2002 drought (Fig. 24B). Forb cover varies more than 30-fold among years, being as low as 0.4 per 100m survey in 1993 and as high as 12.3 per 100m survey in 2004.

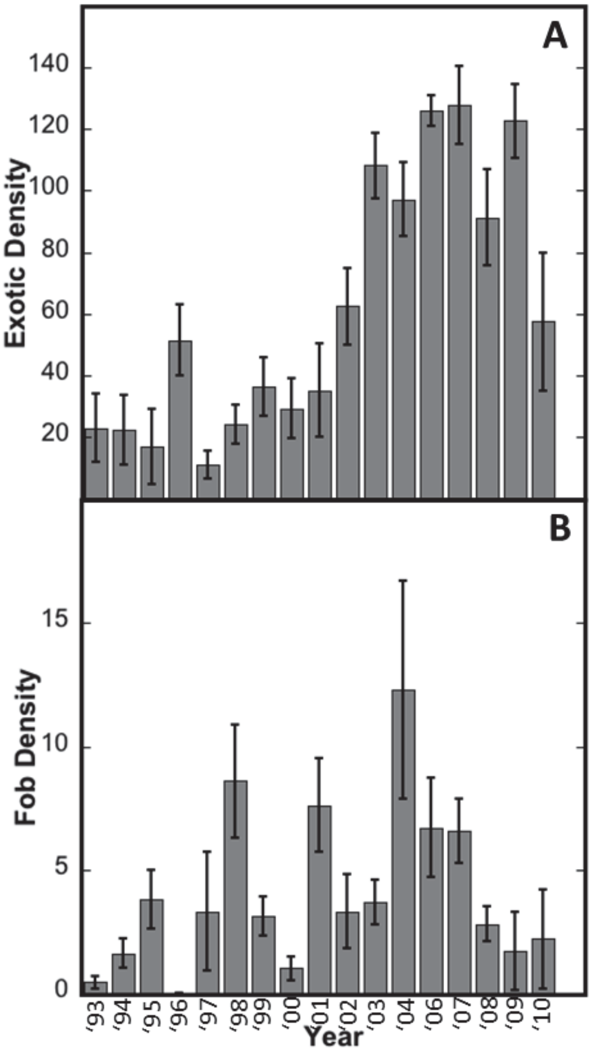

Figure 24. Mean (+/- 1SE) density of (A) total exotic herbs (forbs + grasses) and (B) native forbs across years.

Summary and Projections

Over the past several decades, there have been major transformations of the distribution and the vegetation structure of the ponderosa pine forests of Bandelier. Some of these changes have been induced by deliberate human action (thinning, prescribed burning and seeding), while other changes have been brought about by inadvertent human perturbation (climate change and wildfire). Wildfire in particular has had the greatest impact. Its overall outcome has been the fragmentation of the ponderosa pine forests, and the establishment of large grassland patches. Wildfire has also promoted shrubs both as a dominant species, and as an understory subdominant. As climate increasingly favors fires, it can be expected that their frequency and magnitude will continue increasing (Williams et al. 2010). The recent fires were a unique outcome of changing climate and fuel and canopy conditions brought about by a century of fire suppression. Since canopy cover and continuity of ponderosa pine forests has already changed so much, we can expect that future fire behavior and outcomes could be different. Grassland patches are likely to experience frequent low intensity fire, but are being colonized by shrubs which will change their fire behavior. Shrublands and forests with shrub understories are an interesting phenomenon which are likely to experience moderate to high intensity fire. If trees are part of the community or adjacent to these communities, the fire may be capable of jumping to tree crowns. The major shrubs (oak and locust) are fire resprouters. While we are seeing significant departure from desired vegetation conditions which formerly supported a surface fire regime, one positive aspect is that a more crown fire-resilient vegetation type is emerging. These scenarios described above would logically lead to a future wherein grasses and shrubs are becoming more prevalent and promoting their own persistence by modifying fire behavior and fuel characteristics. It would seem that ponderosa pine will have a lesser role in these forests and former forests in the future.

Major drought events, such as that of 2002 (Breshears et al 2005) is another force influencing vegetation dynamics. As stated elsewhere in this report, the future will very likely be characterized by increasingly warm, and therefore more impactful, droughts (Adams et al. 2009, Williams et al. 2012). Though not specifically addressed in most of the data we had to work with, it is clearly a factor which reduces the probability of the persistence of ponderosa pine stands (Allen & Breshears 1998, Gitlin et al. 2006). Less well understood are its effects on the recruitment of shrubs and exotic grasses. In the Fire Monitoring Plots, there were consistent annual increases in shrub cover such that in 8 years there was an ~18-fold increase in shrubs from 1993 to 2000 (Fig. 23 D). After the drought of 2002, shrubs decreased to 1993 densities. While shrubs decreased, exotic cover increased (Fig 23 A). Our analysis did not explicitly examine the dichotomy between pre- and post drought. However, inspecting the annual variation (Fig. 23 A), there is a ~2.5-fold increase in exotic cover after the 2002 drought. This suggests a drought sensitivity of shrubs, and that possibly exotic species will be the beneficiaries.

In the remaining ponderosa pine stands which have withstood crown fire and drought, management activities may alter the successional course. Prescribed burning and mechanical thinning reduced overstory tree density by ~50 and 40%, respectively from undisturbed controls. The reduction due to thinning was not statistically significant, however, the lack of significance could be due to a low sample size (there were 48 burned plots but only 10 thinned plots) combined with the fact that thinning focuses on small diameter trees which contribute less to overstory canopy density. The reduction in overstory trees is consistent with desired conditions, and would likely avert or reduce the extent of crown fire runs in the future, allowing the persistence of ponderosa pine as a major community member. Interestingly, in contrast to wildfire, we did not find an influence of prescribed burning or thinning on shrub density or understory herbaceous vegetation. It was thought that opening the canopy would promote grasses and forbs, which is another part of the desired conditions. However, we did not find that to be the case. The fact that prescribed burning is less likely to promote shrub communities would seem to suggest that forest to shrubland conversions are avoidable.

While it may ameliorate short-term problems like accelerated erosion, seeding with mixes rich in *Lolium multiflorum* is a management activity that is not likely to lead to desirable long-term vegetation outcomes. Seeding clearly increases grass cover (Fig 22 D). In the Dome study, grass cover was ~2-3 fold greater in seeded areas compared to areas not burned by wildfire. This difference was consistent across 11 years of the surveys, suggesting that seeding immediately after fires may have long-term effects. A large portion of this grass cover is made up of the exotic grasses, *Bromus inermis* and *Lolium multiflorum*. The desired condition of lower erosion risk may have been met, but it came at the price of having undesirable (i.e. exotic) species. Further, the desired condition of greater native grass cover is retarded by the proliferation of exotic species. This does not suggest that seeding solely with native species would produce the same outcome.

Another interesting pattern in terms of seeding, was the complex interactions with shrub cover. Several of our analyses report a “shrubification” or conversion of forests to shrubland. Wildfire seems to catalyze the increase in shrub cover, such that in the last year of the survey, shrub density in the moderate intensity burned areas was ~3x that in unburned areas. This pattern seems to be reduced in seeded plots such that shrub densities in seeded plots were somewhat higher than unburned plots, but lower than unseeded burned plots (Fig. 21 E). This likely suggests that seeded herbs such as *Lolium multiflorum* initially inhibit shrubs. However, there is also evidence that there is an indirect positive influence in that seeding promotes *Bromus inermis*, which in turn promotes shrubs somewhat (Fig. 22). In any case, high intensity wildfire without seeding is the strongest catalyst of shrub proliferation (Fig. 21 E), but seeding may exert multiple negative and positive influences on shrubs with the net outcome of reduction of shrub cover.

In this ecosystem which is so prone to change, desired conditions should be viewed as a moving target. The extent, abundance, and continuity of ponderosa pine has been compromised, and the best guess at the future is a continuation of this trend and an increasing "shrubification" and an increasing presence of exotic grasses. This future does depend on the management actions which are undertaken. There is some evidence that thinning and prescribed burning could slow this trend, by reducing the probability of crown fire in remaining stands, and removing the force which creates shrublands – high intensity fire.

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Chapter 4: Analysis of piñon-juniper woodlands and savannas

Indicators/Measures:

- ***Changes in plant composition and growth form through time in response to restoration activities, drought and fire***
- ***Erosion rates before and after a drought***
- ***Soil aggregate stability and its relationship to vegetation characteristics***

Background & Importance

Piñon-juniper woodlands occur on nearly 40 million ha in North America. (Romme et al. 2010). They are actually a series of different communities, each of which contain any combination of several *Pinus* or *Juniperus* species occurring in semi-arid climates usually at medium elevations (~1200 – 2500m). The overlap of these two genera is not always perfect, rather it is common to find *Pinus* without *Juniperus* at the higher elevations of the range, and *Juniperus* without *Pinus* at the lower end of the range (Martens et al. 2001). This complex of different communities, depending on geographic location, may contain *Pinus edulis, Pinus monophylla or Pinus cembroides*, in addition to one or more of four *Juniperus* species: *Juniperus monosperma*, *Juniperus osteosperma*, *Juniperus deppeana*, *Junipers scopularum* (Romme et al. 2010). In Bandelier National Monument, these woodlands were, until recently, dominated by *Pinus edulis*, and *Juniperus monosperma*, with *Juniperus deppeana* either a subdominant or co-dominant in local areas (Muldavin et al. 2011). In addition to this variation in overstory species composition, different piñon juniper woodlands differ fundamentally in community structure and disturbance regimes. A synthesis by Romme et al. (2010) identifies three general types: Persistent piñon-juniper woodlands, wooded shrublands, and piñon-juniper savannas. Persistent piñon-juniper woodlands are productive and may have nearly closed canopies; these are thought to experience infrequent stand-destroying fire with a return interval of multiple centuries (Huffman et al. 2008). Wooded shrublands exhibit a dominant shrubby aspect, for example *Artemisia tridentata* in the Great Basin with varying tree abundance. Again, this type is expected to experience infrequent stand-replacing fires. A final type is the piñon-juniper savannah, which has widely spaced trees and a grassy understory. This type of system may have been maintained by frequent, spreading ground fire, fueled by herbaceous plants. Such a regime would have constrained tree recruitment, retaining only widely spaced adult trees in burned areas. Evidence for this assertion is reviewed below (*reference conditions*).

Of course many intermediates and combinations of the previous types also exist. In Bandelier, multiple types can be found. Based on 1981 imagery, Allen (1989) maps 5 distinct piñon-juniper communities including savannas, woodlands (with grassy understory), and piñon-juniper-oak communities. Based on 2004 imagery, Muldavin et al. (2011) map 11 distinct types of piñon-juniper communities, including savannahs, wooded shrublands, woodlands (with grassy understory), and piñon-juniper-oak communities. It is thought that a grassier, more open type of woodland was more prevalent in Bandelier (Brockway et al. 2002, Gottfried et al. 1995, Jacobs & Gatewood 1999), and that woodlands have thickened in area at the expense of herbaceous plants. Nevertheless, most piñon-juniper woodlands in Bandelier best fit the persistent woodland type. These have been called savannas in various works (Brockway et al. 2002, Gottfried et al. 1995, Jacobs & Gatewood 1999) but are distinct from the Romme et al. (2010) concept of a frequent fire maintained ecosystem (BF Jacobs personal communication). The situation at Bandelier is made more complicated because there likely were ponderosa pine savannas interspersed with persistent piñon-juniper woodlands. Some of these ponderosa pine savannas were relatively recently replaced by piñon-juniper woodland (Allen & Breshears 1998).

Similarly to any of the other ecosystems in this report, piñon-juniper woodlands harbor their own distinctive fauna, sub-dominant flora, and networks of mutualists and microbial associates. In fact, it has been estimated that about 1000 species are dependent on or interact with *Pinus edulis* in Arizona (Whitham et al. 2003) These woodlands are also a source of piñon nuts, which represent a historical indigenous food, and a small-scale local economy (Jeffers 1995). They also have value for their ability to store carbon in drylands (Huang et al. 2009). Aside from these values, these woodlands have particularly high scientific value. It can be argued that these woodlands function as a model system in climate change science (Bowker et al. 2012). First, as previously mentioned, piñon-juniper woodlands, particularly *Pinus edulis*, are susceptible to climate change induced drought. In fact, piñon pine has been featured as a primary object of study or as an instructive case study in a number of recent high profile scientific works (Breshears et al. 2005, Adams et al. 2009, Williams et al. 2012). While piñon pine may be the current standard bearer for drought research, another aspect of this intriguing system are the strongly divergent responses of piñon and juniper to extreme droughts. All vascular plants obtain CO_2 through stomates on their leaves. Whenever stomates are open to allow in CO_2, they simultaneously allow H_2O to escape. *Pinus edulis* is isohydric, meaning it has adopted an evolutionary strategy of closing its stomates or narrowing their aperture during drought in an effort to conserve water by reducing stomatal conductance McDowell et al. 2008). In extended droughts, the tree may suffer from the closing of stomates by reducing the conductance of CO_2 into the plant. This is the carbon starvation hypothesis (McDowell et al. 2008). It is also thought that a carbon-starved tree is less able to defend itself from insects because it does not have the surplus carbon to synthesize defensive resins (Christiansen et al. 1987, Cobb et al.1997; Classen et al., 2005). In stark contrast *Juniperus* does not close its stomates in response to dry soils; it is anisohydric. This is a risky strategy because water potential in the plant will decline as the soil moisture is depleted, although there is some evidence that foliar absorption of moisture can alleviate this stress (Breshears et al. 2008). If this continues long enough, then the column of water in the plant's xylem will break, leading to cavitation. Once cavitation occurs a plant is no longer able to conduct water and will die. These two plants also differ in regards to the organism they support, especially regarding mycorrhizal fungi. *Juniperus,* like most of the grasses and shrubs in its understory is a host of arbuscular mycorrhizal fungi of the glomeromycota, whereas *Pinus* hosts ectomycorrhizal fungi that are ascomycetes and basidiomycetes. *Pinus edulis* is the major host of ectomycorrhizal fungi in most of these woodlands, thus when it experiences mass mortality, and entire assemblage of fungi no longer have a host or carbon source and could be lost from the system. Thus, the aboveground transformations that have been seem on the landscape (Breshears et al. 2005) likely also signal belowground transformations (Haskins & Gehring 2004, Haskins & Gehring 2005, Swaty et al. 2004). After a drought mortality event, the lack of appropriate fungi in soils can present a barrier to recolonization by *Pinus edulis* (Haskins and Gehring 2005). Another barrier may be that *Pinus edulis* mast seeds every 3-5 years, but these seeds may only remain viable for < 2 years. Thus for maximal recovery potential it would be nest if a mast seeding year had occurred within 2 years before the mortality event.

In summary, studying the piñon-juniper woodlands within a barometer of change (Gitlin et al. 2006) such as Bandelier National monument, carries considerable potential for revealing the probable futures of the woodland ecosystems which occupy 40 million ha of the western US (Romme et al. 2009). Recent large scale (and quite expensive) experiments are currently being used in Sevilleta experimental range to induce some of the drought impacts observed so prominently in Bandelier without manipulation (Pangle et al. 2012). All of these values of drought effects are enhanced due to that fact that Bandelier’s woodlands have undergone extensive thinning treatments, first using live trees and later dead ones; various canopy reduction approaches are increasingly being applied and information is required about their ecological outcomes (Owen et al. 2009). It will be valuable to carefully watch the ecosystem effects of piñon pine loss on other ecosystem components, and whether saplings survive this selection event to adulthood. It will also be very interesting to see the response of juniper, which, with the exception of some local areas, has survived recent drought conditions. However, this pattern is beginning to reverse as a current drought is resulting in some juniper mortality, a trend that should be monitored carefully at Bandelier.

Major Drivers of Change in Piñon-juniper woodlands-

Climate change: The western US is currently experiencing the warmest temperatures observed in a millennium (IPCC 2007). As stated in previous chapters, global circulation models converge on a scenario where northwestern New Mexico will very likely experience warming, and possibly experience less spring precipitation (Karl et al. 2009). Warming is currently increasing the lethality of drought for trees (Adams et al.2009), and *Pinus edulis* in particular is strongly affected (Breshears et al. 2005).

Drought & Insect Outbreaks: One of the most important expressions of climate change is increasing warm drought periods (see previous chapters; Adams et al. 2009). **Piñon pine is the single most susceptible tree species to mass mortality during drought in the Southwestern US** (1950's Allen and Breshears 1998, Chapter 1, Breshears et al. 2005, Gitlin et al. 2006). In the 2002 drought, over 90% of mature piñon pine died, leading to substantial transformations in vegetation community distribution and canopy cover on the landscape (Chapter 1). *Ips* bark beetles and the fungi they disperse lead to mortality of water stressed trees (Negron et al. 2009) because water stressed trees are less able to invest in defense mechanisms such as resin production (Wallin et al. 2004). *Juniperus monosperma*, the most common co-dominant of *Pinus edulis* in Bandelier, is more drought-tolerant but can experience periods of localized mass mortality (Bowker et al 2012), and has done so at the stand level in Bandelier National Monument (BF Jacobs & CD Allen, personal communication). As with other tree species, we consider drought and bark beetles to be a linked stress syndrome (Gaylord et al. 2013) in piñon pine, and do not attempt to consider them separately. Mass mortality in *Juniperus monosperma*, on the other hand, has not been clearly tied to an insect or fungal pathogen.

Persistent accelerated erosion: Persistent accelerated erosion is perhaps less of a change agent than an “agent of stasis”. Erosion may prevent vegetation from changing in a desired trajectory. It is widely believed that because of grazing in the late 1800’s, an herbaceous understory was diminished, favoring higher density of trees and initiating soil erosion (Miller and Wigand 1994, Brockway et al. 2002). Even with the creation of the Bandelier National Monument and the retirement of grazing, high erosion rates continued, possibly exacerbated by a drought in the 1950s (Davenport et al. 1998). High erosion rates create a positive feedback whereby herbaceous plants cannot colonize the unstable surface, and because there are few plants in between trees, erosion is unchecked.

Restoration treatments: In response to accelerated erosion and greater than desired canopy cover, thousands of hectares of piñon-juniper woodland have been subjected to restoration treatments. These treatments consist usually of the removal of some trees or their branches, and dispersal to this material to interspaces with the hopes of constraining erosion and promoting herbaceous vegetation (Jacobs & Gatewood 1999). Prior to the 2002 drought, application of these treatments at scales from small plots (Jacobs & Gatewood 1999) up to an entire watershed successfully arrested the erosion rate and promoted herbaceous vegetation (Hastings et al. 2003). Prior to this time, freshly harvested branches, with needles, were harvested from both piñon and juniper for redistribution. After the drought, dead standing piñon pine were harvested, most having lost their needles previously. The herbaceous plant and erosion control response to post-drought slash was less effective (BF Jacobs personal communication).

Fire: Piñon-juniper woodlands have been less affected by fire than other ecosystems, but have also experienced loss particularly in the 1996 Dome fire and the 2011 Las Conchas fire. The natural fire regime of these ecosystems is not completely known and not without controversy. While some literature suggests frequent ground fire, evidence is mostly from ponderosa pine stands near piñon-juniper stands. Patchy stand-destroying crown-fires may occur in this ecosystem (see below).

Reference Conditions

The past conditions of the piñon-juniper woodlands can be coarsely reconstructed based on historical or archaeological evidence, and are characterized by change. Significant portions of piñon-juniper woodlands may have been cleared on Bandelier's mesa tops (Allen 2004) during the period of ancestral Puebloan settlement. The period of abandonment coincided with a megadrought in the 1500's. Both settlement and mega-drought could be viewed as likely triggers of ecological state transition. Recovery and regrowth from this event must have occurred during the cool period known as the Little Ice Age. It is known that the ecosystems that grew back from these earlier perturbations were woodlands with a substantial grass cover, as noted in the Ramon Vigil Land Grant of 1877 and later adjacent Land Grants surveyed by the General Land Office (Allen 2004). This land grant and the resettling of nomadic Native Americans onto reservations around the 1860's marked the introduction of large scale livestock grazing (Allen 2004), since the Navajo, Apache and Utes served as a deterrent to Euro-American use of the area. Although livestock had certainly been introduced earlier, their numbers are not documented and cannot be estimated well, but are thought to be much lower than after the 1860s. Thus, the period around 1860 - 1870 is one of the better periods to use as a reference to compare current conditions to because it is over 300 years after occupation by ancestral Puebloans, yet just before the introduction of large livestock herds. The introduction of livestock quickly compromised herbaceous plant cover, as stocking rates were an order of magnitude greater than what would be considered carrying capacity today (Foxx and Tierney 1984) By 1913, grass cover was described by the General Land Office as “scant”, compared with descriptions from 30-40 years before as “excellent” or “fine” growth of grass (Allen 2004).

Reduction of the herbaceous layer had one major outcome in an emergent erosion problem. The creation of the National Monument in 1916, and the transition to management by the National Park Service in 1932 led to the termination of commercial ranching, but trespassing or feral livestock have remained a problem (Allen 2004). Erosion and hydrological impacts lingered after the cessation of commercial grazing. As early as 1938 the inability of the soil to retain moisture was noted by the General Land Office (Allen 2004). A 1948 memorandum within the Monument described the large extent of the problem and suggested means to correct it. High erosion rates may have been further accelerated by the 1950’s drought which resulted vegetation mortality. Accelerated erosion has persisted until now as possibly the foremost management issue in the piñon-juniper woodlands.

The sharp decrease in herbaceous vegetation corresponds with a cessation of fires in the general area. An oft-stated belief is that because this major herbaceous component had previously existed in the understories of piñon-juniper communities in Bandelier and nearby that ground fire may have preserved a relatively open canopied configuration (Gottfried et al. 1995, Jacobs & Gatewood 1999). The fire history is difficult to know with certainty because piñon-juniper woodlands are poor preservers of fire scars from which to reconstruct past fire cycles. The evidence for frequent ground fire is indirect but comes from 2 lines of reasoning: 1. Most individual trees are young, whereas older trees are more widely scattered (Jacobs et al. 2008). Since woodlands generally become thick without maintenance by fire, this implies the presence of fires, which did not destroy whole stands, i.e. low intensity ground fire. 2. A frequent fire cycle is well established for directly adjacent ponderosa pine savannas (Allen et al. 1995), and it is reasonable to believe that these fires carried into piñon-juniper stands with herbaceous understories. This series of events may have led to thickening tree canopies, which reinforce a more closed woodland rather than an open grassy woodland. Jacobs et al. (2008) use a predictive model to conclude only about a third of Bandelier's current piñon-juniper woodlands were recently encroached savannas. The other two-thirds likely were persistent woodlands that may have thickened in the past century and which did not have a frequent-fire regime.

Staff at Bandelier National Monument have developed a specific set of *desired future conditions* (Bandelier National Monument 2006), which reflect the findings of Jacobs et al. (2008). First, they distinguish between multiple distinct types of communities, which might commonly be lumped together as piñon-juniper woodlands:

juniper-shrub-grasslands, piñon-juniper savannas, piñon-juniper woodlands. True persistent woodlands are confined mostly to shallow rocky sites.

In the past century, the juniper-shrub-grasslands are thought to have been invaded by juniper and the piñon-juniper savannas are thought to have thickened. These are two systems where more frequent fire is desired: <10 year return in juniper-shrub-grasslands, 10 – 25 year return in piñon-juniper savannas. Low canopy cover (< 5% in juniper-shrub-grasslands, <15% in piñon-juniper savannas) is a part of the desired conditions for each of these systems. Both are expected to support enough herbaceous or shrub understory both to carry fire and to stabilize soils against erosion.

Desired conditions for piñon-juniper woodlands are fundamentally different. These may have a piñon, juniper or mixed canopy exceeding 30% cover. They grow on rocky soils which naturally resist spread of fire, and do not support much herbaceous biomass. Crown fires of small spatial extent (< a few hectares) in intervals from 25 – 100 years would create heterogeneity, and stand age diversity.

Overall, Monument scientist Brian Jacobs envisions a reference mosaic of productive swales supporting *Pinus ponderosa* savannas, persistent piñon-juniper woodlands on rockier uplands, and grading in between these locations would be more open piñon-juniper woodlands supporting herbaceous ground cover (B. Jacobs, pers. comm.). Fires would be primarily ground-based and frequent in swales, often but not always carrying into adjacent grassy piñon-juniper woodlands. Patchy crown fire in persistent woodlands would be an infrequent event, either ignited by lighting or ground fire form adjacent open, grassy woodlands.

Methods & Data

Watershed-scale thinning study

This 15-year dataset documents a landscape-scale experiment (Jacobs et al. 2002; Hastings et al. 2003). The “paired watershed study” examines two adjacent watersheds draining the low elevation mesa tops in Bandelier National Monument. One watershed was selected for a treatment that consisted of removal of about 70% of the trees and dispersing the slash throughout the interspaces whereas the other was left as an untreated control. Each mesa was monitored along twenty 100-m line transects that ran from mesa tops downslope; each transect can be subdivided into upper and lower 50-m segments, and have been analyzed in this way in the past (Jacobs et al. 2002). This results in a sample size of 80 transect segments in total. The data include pre-treatment data from 1996, and sample before and after the 2002 extreme drought that led to mass tree mortality. One-half of the thinned samples were subjected to an experimental prescribed fire in spring 2010. All transects were monitored annually from 1996 to 2000, and again in 2008 and 2010. Data collected include plant percent cover by species (separated into live and dead fractions) and ground cover. The strong spatial replication, temporal richness, and combination of imposed and unplanned ecosystem change make this an extremely high value dataset to understand both different ecosystem states within these piñon-juniper woodland ecosystems, and the particular vegetation changes that may occur due to different drivers such as thinning, drought, and low intensity fire.

Analyses

Using general procedures developed in Bowker et al. (2013), we used the watershed-scale thinning study to develop a state-and-transition model. A state-and-transition model

(STM) is a conceptual summary of dynamics within a particular ecosystem type, and is a also a succession model, which takes into account multiple successional pathways (Bestelmeyer et al. 2003, Briske et al. 2008). A state is comprised of a distinct set of functional and possibly compositional elements, and tends to be stable unless a perturbation triggers a transition to another state. Within states there may exist one or more phases. Phases may exhibit different functional or structural characteristics than other phases within a state, but transitions can occur from phases to phases relatively easily and without external triggers (for example year-to-year changes in the relative abundance of plant species). Briefly, based on literature on known ecosystem dynamics, expert knowledge and ecological principles, an *a priori* model is developed. This model proposes multiple hypothetical and distinctive states, which may exist in space and time within the piñon-juniper ecosystem. For example, one might hypothesize an open grassy woodland or savanna and thickened woodlands as two distinct states. Our *a priori* model is more fully described in Bowker et al. (2013).

To validate the existence of the states proposed in *a priori* STMs, we used a hierarchical cluster analysis, using Ward's method with a flexible beta (McCune & Grace 2002). Our cluster analysis was conducted using live plant functional group cover (e.g. grass cover, forb cover, etc.; both live and dead *J. monosperma* and *P. edulis* were treated at the species level rather than using a tree functional group to elucidate differences in mortality among the species), rock cover, litter cover, downed wood cover, and bare ground cover. For the purposes of our analysis, we treated upper and lower sections of transects as unique observations (as previously done in Jacobs et al. 2002), and treated transect readings of different years as independent observations. The consequences of these simplifications are that it should be more difficult to ascribe the lower and upper segment of a transect, or transect observations in different years to separate clusters. This is because upper and lower segments will tend to resemble each other more than will different transects, and a single transect at a given time point will tend to be similar to itself at a different time point. Thus, from the standpoint of clustering data, this can be considered a conservative analysis. Nevertheless, in practice we had no difficulty ascribing different cluster membership to the same transect through time, or transect segments linked in space. Distance among samples (transect segments at a particular time point) was defined as Bray-Curtis distance.

Hierarchical cluster analysis groups together similar transects segments in a particular time point. Cluster analyses are subjective descriptive tools and should not be viewed as strict hypothesis tests. We considered a cluster to be roughly equivalent to an ecosystem state or phase. Based on the expected number of clusters in our *a priori* models (6), we examined results for 2–10 cluster solutions. We used the following guidelines to select the best number of clusters: (1) acknowledging that we may not observe all of the clusters in our *a priori* model (and that their absence does not prove they do not exist), and that additional clusters may exist that we did not anticipate, we selected a solution with a number of clusters reasonably close to our *a priori* expectations; (2) we accepted clusters which were a good match with our *a priori expectations,* if they existed; (3) we accepted unanticipated clusters when they were consistent with a mechanistic explanation as to how they could emerge (e.g. dictated by abiotic factors or a likely outcome of a given disturbance); (4) we aimed for the number of clusters which explained about one-half the variation in the data. We selected the solution that *best* satisfied *all* of the above criteria. To help us define the characteristics of our clusters we applied indicator species analysis (Dufrene and Legendre, 1997) and viewed NMDS ordinations.

Similar clusters were grouped as phases of states as an interpretive aid. As an additional aid to the development of a final state and transition model, we tabulated cluster membership for each transect segment over time. This assisted us in determining which transitions were possible and which triggers (treatment, drought) were likely involved.

Pooled small-scale restoration studies

Between 1992 and 2004, 3 distinct but similar studies were conducted which manipulated piñon-juniper canopy cover, and in two of the studies, applied additional treatments of the ground surface in an effort to promote herbaceous biomass such as seeding, mulching or raking. Second only to drought mortality in the overstory, the condition of the understory is the most important character of piñon-juniper woodlands because it influences erosion which in turn influences possible vegetation dynamics. The understory also distinguishes among some ecosystem states and phases (see *Condition & Trend*, below). We wished to determine the relative importance of canopy treatments, understory treatments, and other factors on the relative abundance of understory plant functional groups. Two of the datasets capture conditions before and after the severe 2002-2003 drought, adding another driver of understory vegetation change. Finally, two of the studies also had plots which experienced unplanned wildfire. Pooling of these data allows a wider variety of time periods and treatment combinations to be analyzed, alongside climate conditions, age of treatment, and fire.

Chong study - In 1992 a study by Geneva Chong (Chong 1993) evaluated revegetation methods for degraded piñon-juniper woodland. The blocked study design consisted of three types of tree overstory treatments crossed with various understory treatments. The Chong study was located just up-mesa of the Frijoles Mesa small scale study area. The tree overstory treatments included: cut and scatter slash or girdling, and control (no treatment). The modification of overstory canopy was minor, only enough canopy was removed to provide slash for redistribution. The understory treatments were: control, raking, seeding, seeding + straw mulch, seeding + fertilizer, seeding + fertilizer + straw mulch. Each block was assigned to one of the three overstory treatments for a total of 9 “blocks.” A block was 26m by 39m in size and was subdivided into six 13m X 13m plots. Each plot had twelve data collection points located on a grid for a total of 72 points/block. To collect vegetative cover data, a 0.5m^2 rectangular frame was centered over each point and the percent cover by species or growth form within the frame was visually estimated. The plots were resampled by Bandelier National Park staff in 1995 in the cut and scatter and control blocks. Our analysis used data at the 13m x 13m plot level, omitting the plots with girdled trees because they were not comparable to the following studies.

Jacobs study

Jacobs and Gatewood (1998) in 1994 established two sites on Garcia Mesa and Frijoles Mesa. Within each site, 15 contiguous 15m^2 plots (representing three five-plot replicates) were established with canopy removal and slash redistribution (applied in 1995) along with 3 control plots (one per replicate), which were near to but not contiguous to the other plots. One site received a 100% canopy reduction, and one received a 70% canopy reduction (similar to Loftin 1998). Each of the treatment replicates were sub-divided into 5 blocks in which understory manipulations were applied at random: thinning-slash only, imprinting (the creation of impressions in the soil surface to act as passive collection points of seeds, organic matter and water), seeding, imprinting + seeding, and raking + seeding. In 1995 a third site was established, and treated in 1996, however the understory seeding-soil preparation treatments were never applied. This site used a 100% canopy removal treatment comparable to Frijoles Mesa (BF Jacobs pers. comm.). At least some sites were sampled from 1994 – 2001, and again in 2004. All of one site, and portions of another site burned in the Cerro Grande fire, thus all data from 2000 and later reflect post-fire recovery.

Loftin study

Loftin (1998) established two treatments in 10-15 ha blocks, and overstory thinning & slash redistribution treatment, and a control. The thinning treatment was an aggressive one simulating a fuelwood harvest. All *Juniperus* were removed, and all *Pinus* < 20 cm diameter were removed resulting in a nearly complete canopy reduction. In practice it was approximately a 70% canopy removal and identical to the co-located small scale

Garcia Mesa study site (Jacobs and Gatewood 1998). The treatment and control areas were divided into 5 contiguous plots, and in each plot, a 100m transect was used to monitor vegetation with regularly-spaced quadrats. These transects were sampled from 1994 – 2003, except for 1998. We used the average of quadrats along individual transects as independent observations for two reasons: 1. This makes the area sampled more similar (though still larger) to the Chong (1993) and Jacobs and Gatewood (1998) studies, 2. Although the 1-2 ha plots they sampled were contiguous, the 100m transects used to monitor vegetation were discrete in space. This entire study was burned in the Cerro Grande fire, and all data from 2000 and after reflects post-fire recovery.

Analysis

All of these studies have previously been analyzed individually and have been associated with some form of publication in the past (Chong 1993, Jacobs and Gatewood 1998, Loftin 1998), thus to repeat this was not useful. Rather the goal of our analysis was to develop a synthetic approach which determines the relative importance of interannual climate variability (including drought), overstory reduction and slash redistribution, ground surface manipulations, age of treatments, and fire on the relative functional group composition of the understory. This set of possible drivers has not been evaluated together before in these studies.

Because all three studies used different techniques and personnel to evaluate understory vegetation, we felt that an analysis of absolute abundance of different vegetation components could not be conducted with confidence. Instead we calculated the relative abundance of live total cover different plant functional groups (grasses, forbs, total herbaceous, total understory), bare ground, and litter, which does not assume that all methods would have produced similar covers estimates. Rather it assumes only that there is no bias in over- or underestimating any functional groups among studies. Second, we pooled all observations and coded them based on experimental manipulations. First we coded four different types of overstory reduction and slash redistribution treatments: no thinning, Chong (1993) protocol, Jacobs and Gatewood (1999) Frijoles and Alamo Mesa sites100% thinning protocol, Jacobs and Gatewood (1999) Garcia Mesa site/Loftin (1999) lighter thinning protocol (~70%). Ground surface manipulations were largely coded with binary “dummy variables” (i.e. yes/no) coding whether or not the ground surface had been mulched with straw, imprinted, raked or fertilized. Because different mixes and application rates were applied, seeding was coded with 3 levels: control, Chong (1993) protocol, and Jacobs and Gatewood (1998) protocol. Finally, a fire dummy variable was created indicating those plots which burned in the Cerro Grande fire which included some portions of the Jacobs and Gatewood (1998) study, and the entirety of the Loftin (1998) study

Because these are repeated measure studies, time must be accounted for in some way. We chose to treat observations through time as discrete samples and introduce multiple predictor variables that would account for difference in samples from year to year. Time is accounted for explicitly in 2 climate variables: current year precipitation (sourced from a fire tower in the piñon-juniper woodlands, supplied by BF Jacobs), and previous 2 years + current year precipitation. Number of years since treatment was introduced as an “age of treatment” variable.

We analyzed the pooled dataset using boosted regression trees (Elith et al. 2008). This method is an extension of the regression tree method employed in *Chapter 2*. Rather than seeking to develop one predictive tree which models means of a response variable based on dichotomous rules based on predictors, “boosting” uses a large number of simple trees together. First, a simple tree, usually with < 3 splits is fit to the data. Then another simple tree is fit to the residuals. This process is repeated until no new information is explained or until the procedure reaches user-set stopping rules. We limited our tree size to 3 splits, and our number of trees to 50. We made no attempt at editing data to create a more balanced meta-study, nor is a balanced design needed to apply this technique. Rather we treated data as a large set of individual observations dispersed in space and time, and used all available data. The benefit of this approach is that it ensures that a broader set of manipulations is addressed, and that the period of time in which we have observations is extended.

Like first generation regression trees, these models can be used as predictive models, although they are more complicated and difficult to visualize. Our interest in the method was its ability to estimate the relative influence of different predictors on a response variable.

NPS Inventory & Monitoring (SCPN) monitoring dataset

The NPS Inventory & Monitoring network (SCPN) has established a set of 46 monitoring plots in the piñon-juniper woodlands on mesa tops at Bandelier. Fifteen were established in 2008, 25 were established in 2009, and 6 were established in 2010. Though these will continue to be monitored in the future, we have access only to one sampling per plot at this time. Plots were thinned from 2007 – 2009 and surveyed between 6 months and 2 years after treatment. Plots are square and 0.50 ha in size (71m x 71m). Three 50m transects, paced 25 m apart, were used to sample shrub, herbaceous vegetation, ground cover (biological soil crusts, rock, litter, etc.) and soil stability. Percent cover of shrubs and herbs, and ground cover, was estimated within quadrats sampled along the transects (Decoster and Swan 2011).

Soil aggregate stability was estimated using a field-based test developed by Herrick et al. (2001). This method focuses on resistance to water erosion but is also informative regarding resistance to wind erosion. This method subjects dry soil aggregates to a timed water immersion and wet sieving regimen and assesses cohesion using an ordinal scale ranging from 0 (too unstable to sample) to 6 (greatest aggregate stability). The test simultaneously takes into account slaking, and effects of shear stresses, and is being widely used by land management agencies as an easy yet informative monitoring method. Soil aggregate stability was measured 18 times per plot, with each of the 18 measurements taken across three 50m transects in each plot.

Measurements of overstory trees and saplings were collected in subplots within the larger plot. Canopy closure was measured with a hemispherical densiometer in early plots. The methodology was later switched to a line intercept measurement of canopy cover. There were 25 plots in which both types of measurements were taken, and the two methods were highly linearly related. We used these points to generate a regression which we used to interpolate missing canopy closure variables.

Analysis

Response Variable **–** Because soil erosion is a major factor that can promote or retard vegetation change in the piñon-juniper woodlands, we focused our analysis on soil aggregate stability. Erosion is influenced by two primary factors: 1. Erosivity, the ability of erosive forces to move sediment, and 2. Erodibility, the propensity for sediment to be moved by erosive forces. Soil aggregate stability is a negative index of erodibility (Bowker et al. 2008), such that low scores indicate high erodibility. Prior to analysis, we squared the aggregate stability values, then calculated an average value for the plot. We used the squared values because this field test is curvilinearly related to % water stable aggregates, a more time-consuming and more precise laboratory method. Based on data presented in the original paper by Herrick et al. (2001), squaring the field kit data improves the linearity of the relationship between these two values. Another issue with the raw data is that it is expressed on an ordinal scale; thus although it is commonly done, taking averages of multiple measurements is questionable. A square transformation makes the data more similar to a continuous linear scale, so that averages may be taken more confidently. In practice, this transformation also improves the linearity of the relationship between soil aggregate stability ratings and some predictors, such as chlorophyll a, an indicator of biological crust biomass (Bowker et al. 2008). In all of our analyses, we exclude measurements taken on pumice, as pumice surfaces generally are not susceptible to erosion and thus do not contribute to our knowledge about the relationship between biotic characteristics and soil stability.

Choosing Explanatory Factors: Before running our model, we chose variables based on *a priori* reasoning that we thought would contribute to soil stability. Specifically, we chose calendar year of sampling, mechanical thinning (Yes or No), canopy closure, total biocrust cover (which was a composite of cyanobacteria, undifferentiated crust and lichen), total litter cover (a composite of duff/litter and dead woody debris), forb cover, annual grass cover, perennial grass cover, shrub cover and the canopy closure-by-mechanical thinning interaction. We included year because a concurrent analysis of another data set (LTER transects, below) showed that erosion varies by year. Unlike in wetter environments (Tisdale and Oades 1982), dryland soil aggregate stability is largely determined by live biota (biocrusts, plants and their fungal symbionts), rather than soil organic matter (Chaudhary et al. 2008), thus we included cover of biocrusts and different plant functional groups and canopy closure in our models. Total litter cover was included because it physically shields soil aggregates from erosive forces, possibly allowing aggregates to retain greater stability. Mechanical thinning was expected to influence soil aggregate stability directly because disturbance associated with recent thinning activity could destroy soil aggregation to some degree. Also mechanical thinning was expected to exert multiple indirect effects due to modifying canopy closure, herbaceous vegetation, and the amount of litter and coarse woody debris on the soil surface. Consequently, we chose to include the interaction term (canopy closure-by-mechanical thinning) because we thought mechanical thinning could affect canopy closure, which could in turn affect soil stability. Next, we used the stepwise function in JMP Pro 9 (SAS Institute Inc.; Carey, NC), specifying minimum AICc to select a subset of the above variables. This method used the minimum corrected Akaike Information Criterion to select the best model. This process selected year, mechanical thinning, forb cover, canopy closure and canopy closure-by-mechanical thinning interactions, which we then used as explanatory factors in our final ANOVA model explaining variation in soil stability.

“LTER” / “JRM” long-term transects dataset

This section examines two data sets associated with a set of long-term transects and their interrelationships (Chong et al. 1993). The first dataset contains long-term vegetation data monitored using a line-intercept method. This data was collected on 9 transects (JRM 1,2, 6, 7, 8, 9, 10, 11 and 12) surveyed in 1999 and 2003. The collection of these data is fortuitous as is allows for the monitoring before and after the 2002 drought. A second dataset includes erosion measurements from 11 transects (9 vegetation transects plus JRM 3 and 4), measured in each year from 1999 to 2007. Erosion was monitored using erosion bridges at fixed points along the vegetation transect. We combined these two data sets to examine possible relationships between vegetation structure and erosion.

Analysis

Vegetation -We conducted several analyses to examine how the vegetative community is changing across time. Like other data sets, we focused our analyses on the live canopy cover of the vegetative community. First, using year and transect as explanatory variables, we used perMANOVA (Anderson 2001) to examine how the composition of species and functional groups may have changed according to these factors. To complement the perMANOVA, we used non-metric multidimensional scaling (NMDS) to graphically depict the composition of functional groups across these same factors. NMDS is an ordination technique that condenses multivariate data (such as vegetation community data) into a single point and places it in ordination space (McCune and Grace 2002). In NMDS plots, it is not the position of individual points that is relevant, but rather the relative position of multiple points. Specifically in this case, two points that are close together represent vegetation communities that relatively similar and points that are far apart represent communities that are relatively different. Thus, looking across NMDS axes, one can get a sense of a relative gradient in community composition as well as possible differences in community composition across levels of factors of interest (such as year and transect; see below). The perMANOVA and NMDS were conducted using the adonis and metaMDS functions, respectively, in the vegan package of R (Oksanen et al 2011).

While the perMANOVA and NMDS can detect and graphically depict overall differences in community composition, they do not specify which components of the community may be changing. Thus, in addition to the perMANOVA and NMDS, we also tested for differences in total vegetative abundance, species richness and the abundances of individual functional groups (forbs, grasses, cacti, shrubs, trees and subshrubs). In our analysis of individual functional groups, we analyzed both absolute abundances and the proportion of each functional group in the community (i.e. the abundance of each functional group / sum of all functional groups). Analyzing the proportion of each functional group is useful for revealing shifts in the dominance of particular functional groups, if present. To test for differences in these response variables, we used an ANOVA with year and transect as explanatory factors in JMP Pro 9 (SAS Institute Inc.; Carey, NC).

Erosion -We also examined the effect of transect and year on the change in soil surface elevation from the previous year. This response variable produces an annual sediment loss or gain rate, and a good relative index for comparison among erosion bridges. In this analysis, negative numbers connote soil erosion and positive numbers connote soil deposition. For the data collection, the same 300m transects were used as in the vegetation surveys (above). However, instead of reading erosion along the entirety of the transect, four positions, with 50 – 200 m between positions, were chosen along the transect. In our analyses of the factors affecting erosion and the link between erosion and vegetation (see below) we used the average change in soil surface elevation per transect. We did this because the multiple measurements per plot are not independent from each other. Also, in each plot, vegetation and soil elevation were not measured at the same spatial scale. Thus, we evaluated the vegetation transect and the average erosion bridge measurements as representing the vegetation and erosion for that local area. Like the model for changes in vegetative functional groups, we used ANOVA (JMP Pro 9, SAS Institute Inc.; Carey, NC) with year and transect as explanatory variables.

Linking Vegetation to Erosion - With data on the two variables over time, we conducted analyses to explore the potential relationship between vegetative characteristics and erosion. First, we examined the relationship between community composition and erosion. To do this, we used two methods. Both methods employ NMDS as a data reduction tool. After an ordination has been conducted, the axis scores (coordinates that describe where a sample is mapped in ordination space) can be saved. These scores can then be used as univariate summaries of the key aspects of the entire community composition. Our first method used regression to see if NMDS scores would explain variation in erosion. In one model, we included NMDS axis 1 and NMDS axis 2, as well as year. Knowing that erosion can vary by year, (see results below), we also analyzed each year separately. For these latter analyses, we generated NMDS scores for each year separately then used those new NMDS scores in regressing erosion measurements onto NMDS scores. We conducted all of these analyses twice; using data on species and functional group composition. Our second method, also regression, examined the influence of univariate vegetation aspects of the community (total vegetative cover, species richness and abundance of individual functional groups). It is certainly possible that vegetation characteristics of one year could affect erosion in subsequent years (e.g. vegetation in 2003 affecting erosion in 2004), but to limit the number of hypotheses tested, we focused our analyses only on years in which both vegetation and erosion were measured.

Condition & Trend

Possible ecosystem states and transitions among them

We developed a state-and-transition model based on our analysis of the watershed-level thinning study. Our cluster analysis found eight clusters of samples in time and space which largely confirmed our conception of ecosystem states (Fig. 25), but revealed some common variants within those states. Our ordinations (Fig. 25) plot samples such that closer points are more similar than distant points. “Distance” is based on all of the analyzed variables (plant and soil surface cover), and thus condenses many different pieces of information. The samples are coded with symbols representing the 8 clusters from our cluster analysis (Fig. 25). We display six versions of the same ordination, which differ in that points are rescaled based on the relative abundance of a type of plant or ground cover. From this information we can begin to describe the characteristics of the 8 clusters, and the ways in which they differ from each other. To aid interpretation, the 8 clusters are grouped together in various combinations as variants of ecosystem states. We found evidence of both productive and unproductive woodlands. Unproductive woodlands may be those undergoing accelerated erosion based on low understory cover and low litter accumulation, although erosion is not measured in this dataset. Two clusters indentify productive woodlands, which are characterized by high litter. Within the productive woodlands there are two variants with different higher ratios of *J. monosperma* to *P. edulis*. Unproductive woodlands are characterized by bare ground. There are also two variants of unproductive woodlands, one characterized by more shrubs and *J. monosperma*. Open woodlands exhibit some degree of herbaceous vegetation and high cover of dead *P. edulis*. Savannas are represented by 3 variants which differ in the amounts of litter, and live and dead herbaceous cover. We believe that the desired state of the Bandelier piñon-juniper woodlands best matches the characteristics of open woodlands and savannas in the areas sampled, although productive woodlands may not conflict with the desired state of the ecosystem. Unproductive woodlands are most consistent with degraded states, but they could also represent persistent woodlands that happen to grow on naturally steeper, rocky, or shallow-soiled sites.

An examination of cluster assignments of transect segments over time, in relation to known perturbations or manipulations, reveals much about possible transitions (Table 3). Overall, productive and unproductive woodlands prevail in similar numbers prior to

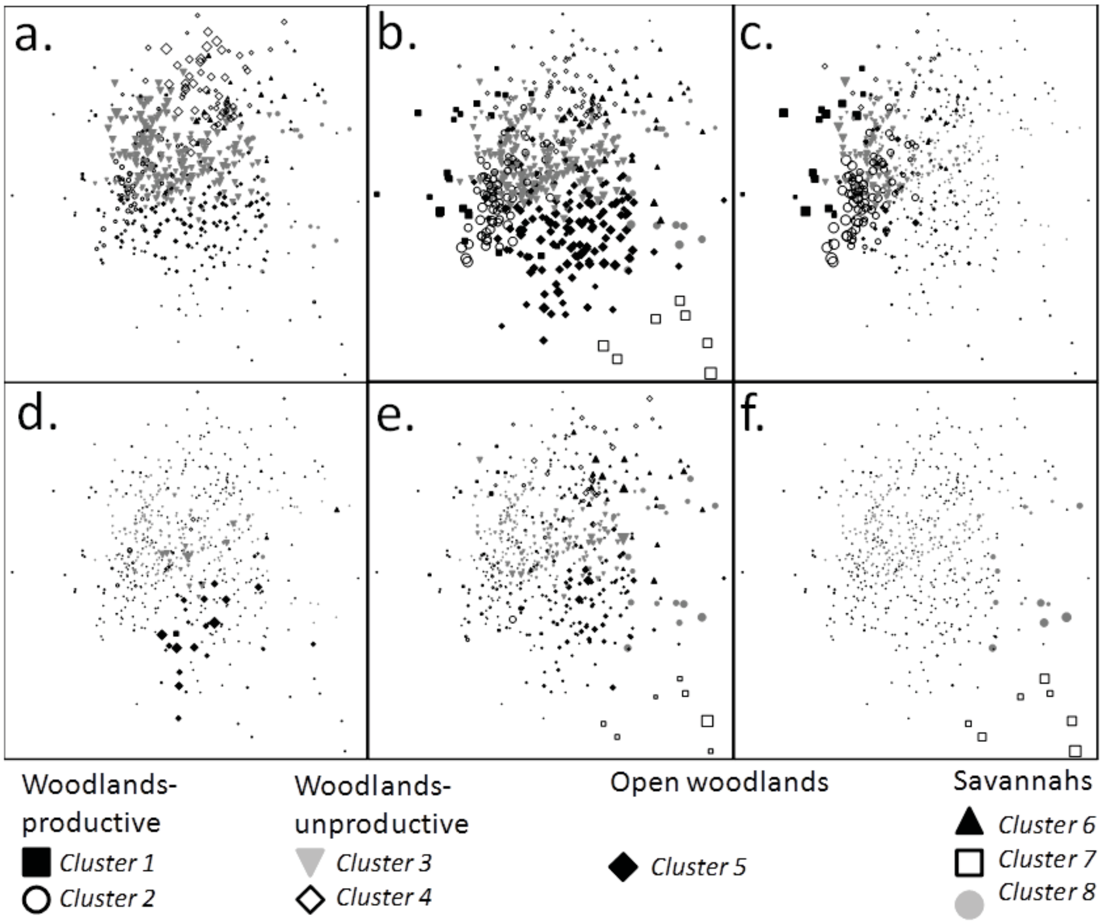


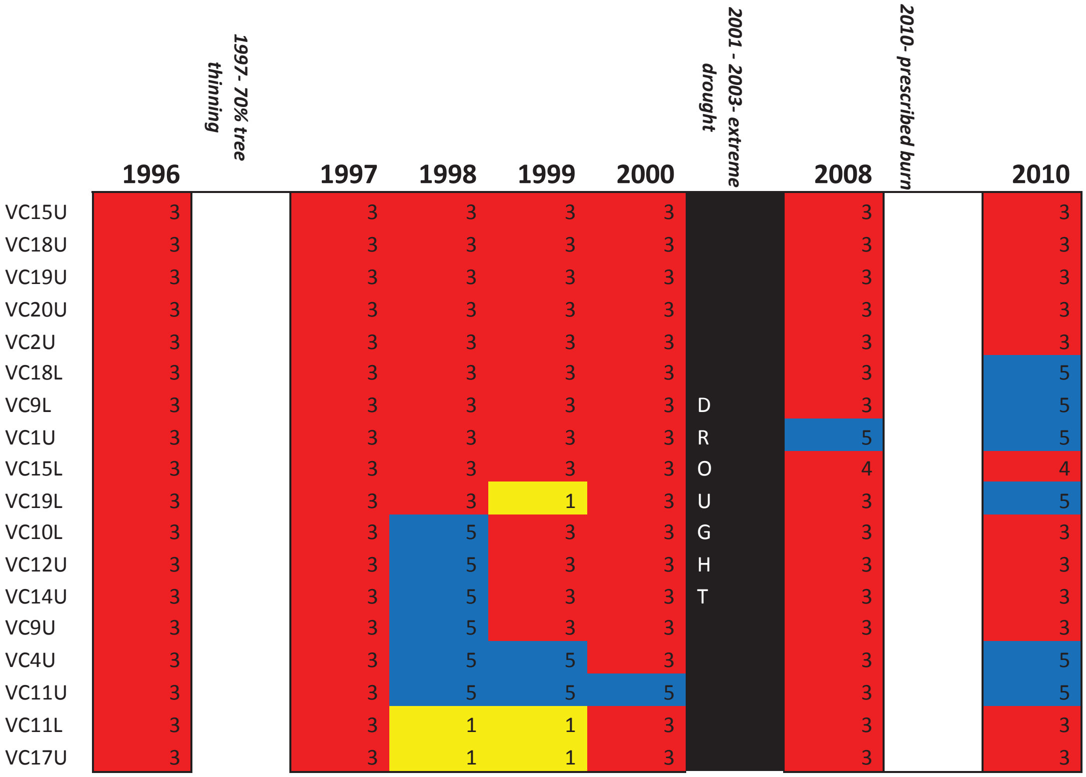
Figure 25. Diagrams showing six versions of the same non-metric multidimensional ordination with symbols coding for the eight clusters found in our cluster analysis. In each panel a point is a transect in a given time point; the position of each point is identical in all panels. The panels differ in that the size of symbols is scaled proportionally to a particular element of the community that is characteristic of one or more clusters. (a.) Symbols are scaled according to percent cover of bare ground, which is especially high in clusters 3 and 4. (b.) Symbols are scaled according to percent cover of litter, which is high in clusters 1, 2, 5 and 7. (c.) Symbols are scaled according to live *P.edulis* cover, which is high in clusters 1, 2, and to a lesser extent 3. (d.) Symbols are scaled according to dead *P. edulis* cover, which is high in cluster 5. (e.) Symbols are scaled according to live forb cover, which is high in clusters 6, 7 and 8. (f.) Symbols are scaled according to dead grass cover, which is high in cluster 7.

the overstory thinning treatment (far left column of Table 2), and constitute the majority of samples. Both the overstory thinning treatment and drought lead to a greater prevalence of open woodlands over the course of monitoring. Open woodlands appear to be a relatively unstable state and can transition to savannas, especially under thinning, stay the same, or regress to unproductive woodlands. Transitions from unproductive woodlands to other states are observed, but the most frequent outcome was for unproductive woodlands to remain unchanged, or move toward another state only to later regress. In contrast, transitions from productive woodlands to open woodlands or savannas in response to thinning treatment or drought seem much more common and persistent (Table 2, note the change from yellow to blue or green).

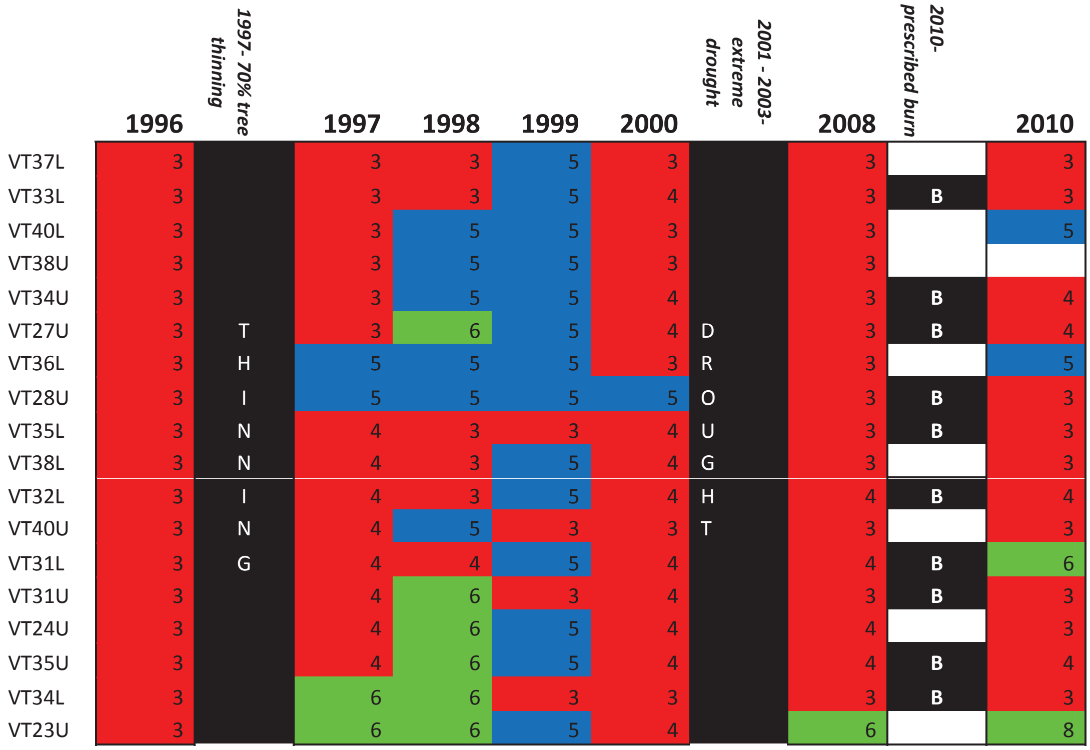
The thinning treatment produced variable results. Commonly, thinning led to the conversion of productive woodlands to open woodlands or savannas, and the conversion of open woodlands to savannas. Many thinning-induced transitions to savanna-like vegetation where transitory, reverting back to other states after 1998.

Drought can suddenly transition a productive woodland into an open woodland, and does so in almost every case (Table 2: note the change from yellow to blue after the drought). Previously thinned transects are less consistently affected by subsequent drought. This is probably due to the removal of drought susceptible piñon pine trees prior to the drought, leaving a less drought-susceptible plant community.

Compared to the other two major change agents, effects of burning are unclear; in 50% of cases no state transition is observed, however this was a very low intensity burn.

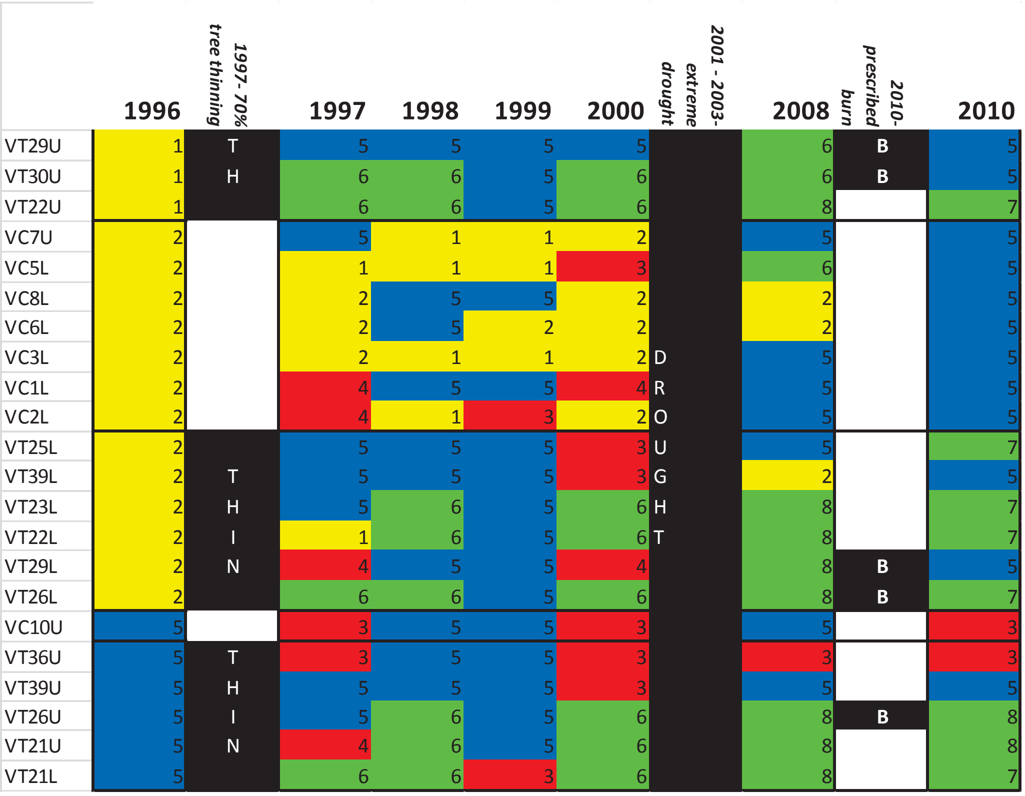


	1996	*1997- 70% tree thinning*	1997	1998	1999	2000	*2001 - 2003- extreme drought*	2008	*2010- prescribed burn*	2010
VC15U	3		3	3	3	3		3		3
VC18U	3		3	3	3	3		3		3
VC19U	3		3	3	3	3		3		3
VC20U	3		3	3	3	3		3		3
VC2U	3		3	3	3	3		3		3
VC18L	3		3	3	3	3		3		5
VC9L	3		3	3	3	3	D	3		5
VC1U	3		3	3	3	3	R	5		5
VC15L	3		3	3	3	3	O	4		4
VC19L	3		3	3	1	3	U	3		5
VC10L	3		3	5	3	3	G	3		3
VC12U	3		3	5	3	3	H	3		3
VC14U	3		3	5	3	3	T	3		3
VC9U	3		3	5	3	3		3		3
VC4U	3		3	5	5	3		3		5
VC11U	3		3	5	5	5		3		5
VC11L	3		3	1	1	3		3		3
VC17U	3		3	1	1	3		3		3



	1996	*1997- 70% tree thinning*	1997	1998	1999	2000	*2001 - 2003- extreme drought*	2008	*2010- prescribed burn*	2010
VT37L	3		3	3	5	3		3		3
VT33L	3		3	3	5	4		3	**B**	3
VT40L	3		3	5	5	3		3		5
VT38U	3		3	5	5	3		3		
VT34U	3		3	5	5	4		3	**B**	4
VT27U	3	T	3	6	5	4	D	3	**B**	4
VT36L	3	H	5	5	5	3	R	3		5
VT28U	3	I	5	5	5	5	O	3	**B**	3
VT35L	3	N	4	3	3	4	U	3	**B**	3
VT38L	3	N	4	3	5	4	G	3		3
VT32L	3	I	4	3	5	4	H	4	**B**	4
VT40U	3	N	4	5	3	3	T	3		3
VT31L	3	G	4	4	5	4		4	**B**	6
VT31U	3		4	6	3	4		3	**B**	3
VT24U	3		4	6	5	4		4		3
VT35U	3		4	6	5	4		4	**B**	4
VT34L	3		6	6	3	3		3	**B**	3
VT23U	3		6	6	5	4		6		8

	1996	*1997- 70% tree thinning*	1997	1998	1999	2000	*2001 - 2003- extreme drought*	2008	*2010- prescribed burn*	2010
VC7L	4		4	1	6	4		8		7
VT33U	4		3	5	5	4		3	**B**	4
VT37U	4	T	4	3	3	4		4		
VT32U	4	H	4	4	3	4		4	**B**	4
VT25U	4	I	4	4	5	4		8		8
VT27L	4	N	4	6	3	4		3	**B**	8
VT28L	4		4	6	3	6		8	**B**	8
VT30L	4		4	6	5	4		8	**B**	8
VC20L	1		3	5	5	3		3		3
VC17L	1		1	1	5	1	D	5		5
VC12L	1		1	1	1	3	R	3		5
VC3U	1		1	1	1	3	O	5		5
VC13L	1		1	1	1	1	U	5		3
VC14L	1		1	1	1	1	G	5		5
VC16L	1		1	1	1	1	H	5		5
VC16U	1		1	1	1	1	T	5		5
VC4L	1		1	1	1	1		5		5
VC6U	1		1	1	1	1		5		5
VC8U	1		1	1	1	1		5		5
VC13U	1		1	1	1	1		5		
VC5U	1			1	1	1		5		5
VT24L	1		5	5	5	5		5		5



	1996	1997 - 70% tree thinning	1997	1998	1999	2000	2001 - 2003- extreme drought	2008	2010- prescribed burn	2010
VT29U	1	T	5	5	5	5		6	B	5
VT30U	1	H	6	6	5	6		6	B	5
VT22U	1		6	6	5	6		8		7
VC7U	2		5	1	1	2		5		5
VC5L	2		1	1	1	3		6		5
VC8L	2		2	5	5	2		2		5
VC6L	2		2	5	2	2		2		5
VC3L	2		2	1	1	2	D	5		5
VC1L	2		4	5	5	4	R	5		5
VC2L	2		4	1	3	2	O	5		5
VT25L	2		5	5	5	3	U	5		7
VT39L	2	T	5	5	5	3	G	2		5
VT23L	2	H	5	6	5	6	H	8		7
VT22L	2	I	1	6	5	6	T	8		7
VT29L	2	N	4	5	5	4		8	B	5
VT26L	2		6	6	5	6		8	B	7
VC10U	5		3	5	5	3		5		3
VT36U	5	T	3	5	5	3		3		3
VT39U	5	H	5	5	5	3		5		5
VT26U	5	I	5	6	5	6		8	B	8
VT21U	5	N	4	6	5	6		8		8
VT21L	5		6	6	3	6		8		7

Table 2. Cluster membership of samples through time. Clusters are color coded according to final state-and-transition model: red, unproductive woodlands; yellow, productive woodlands; blue, open woodlands; green, savannas.

We summarized the above observations in a state-and-transition model (Fig. 26), which provides a useful framework for conceptualizing the variation in piñon-juniper landscapes, the vegetation changes that are possible, and the drivers of that change. A more complete catalog of states, phases and transitions can be found in Bowker et al. 2013. In some ways, productive woodlands appear to be moving in a trajectory consistent with desired conditions in that the tree canopy has opened, and herbaceous vegetation has increased. This can be induced by thinning treatments which add fresh slash along with attached leaves. It can also be induced by drought mortality of the overstory. Despite these positive developments it cannot be ignored that a major departure from desired trajectories is the nearly complete loss of mature piñon-pine, and current mortality of juniper (*not observed in data*). Finally, unproductive woodlands, which were prevalent on the landscape appear relatively static, neither degrading further or undergoing desired transitions.

We sought to further understand the role of thinning and slash redistribution on promotion of herbaceous vegetation and reduction of erosion using the small-scale restoration studies, and the SCPN monitoring plots and the LTER/JRM transects.

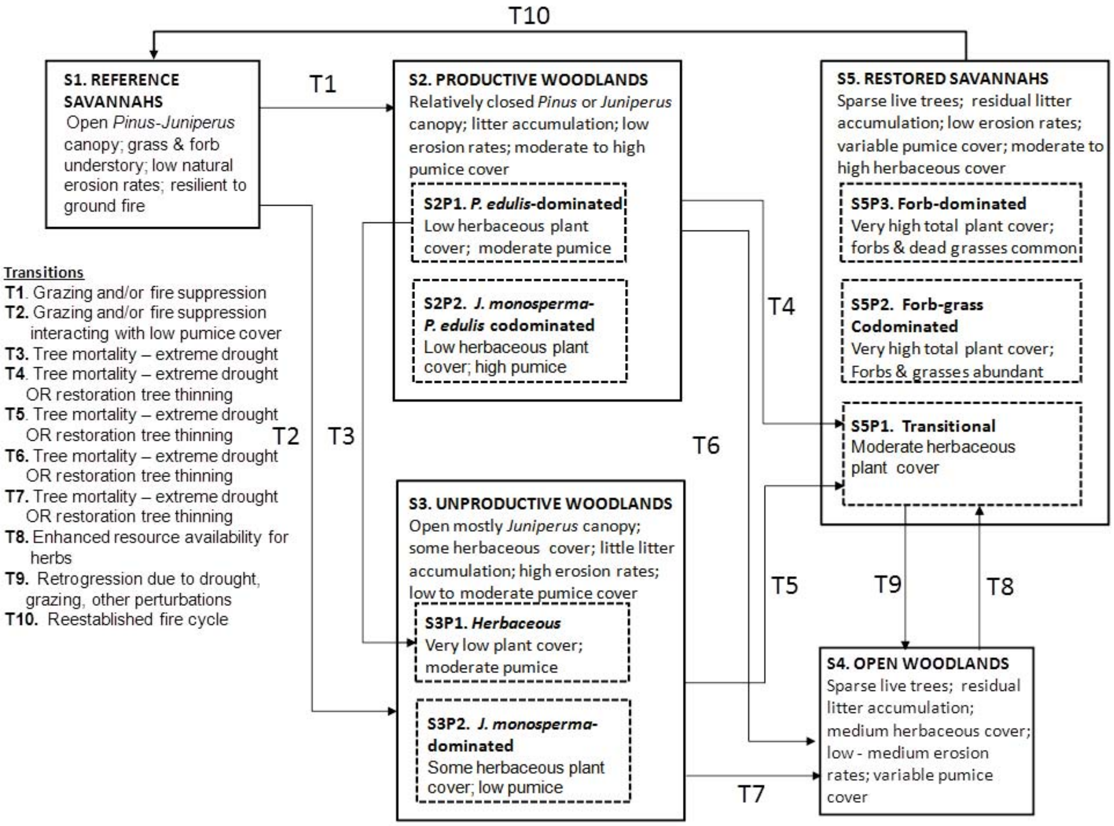


Figure 26. State-and-transition diagram for Pinyon-juniper Mesa Tops. Solid boxes represent ecosystem states. Dashed boxes indicate phases within states. Arrows indicate transitions. In some cases, phases within the reference state are not connected to any others by arrows; this is our method of representing spatial variants of the reference state that are determined by abiotic factors, or cases where we simply do not have a strong hypothesis for the relationship among phases.

Effects of thinning and slash redistribution on herbaceous understory and erosion

Pooled small-scale datasets-

The various live understory components were most strongly regulated by the type of overstory reduction applied and years since treatment (Fig. 27). In the case of total understory, total herbaceous cover, and total forb cover, years since treatment was the single most important predictor. In the case of total grass cover, overstory reduction was the most influential variable. Current year rain was another strong influence, especially on herbaceous understory components but not on litter or bare ground. Rainfall of course influences current year production, previous year establishment or death, and the seed bank. The current year + previous 2 years of rainfall was a comparatively weak predictor. Whether or not the plot had burned was a small but non-negligible influence on the live understory vegetation, but was the strongest determinant of litter cover and bare ground cover (Fig. 27). Interestingly, direct manipulations of the understory were universally uninfluential regarding the eventual development of the understory. The age of the treatment was modestly influential, but in particular affected forbs.

These boosted regression tree results tell us the relative importance of the various factors but not about *how* these factors influence cover of understory, bare ground or litter. To determine the most important patterns we also viewed simple regression trees limited to only 2-3 splits.

In the case of total understory, cover was nearly double (0.16) if overstory thinning methods used by Jacobs & Gatewood (1999) or Loftin (1999) were applied, compared to either no thinning or the much lighter Chong (1993) protocol (0.09). In the more aggressively thinned stands, cover was higher if current year rain was ≥ 340 mm (0.20), than if rainfall was lower (0.12). Finally, within aggressively thinned stands in years receiving > 340 mm of rainfall, cover was more than double if the treatment was at least 3 years old (0.23) compared to younger treatments (0.11).

In the case of total herbaceous cover, the strongest determinant of cover was current year rainfall; cover was approximately double if rainfall was ≥ 290 mm (0.11),

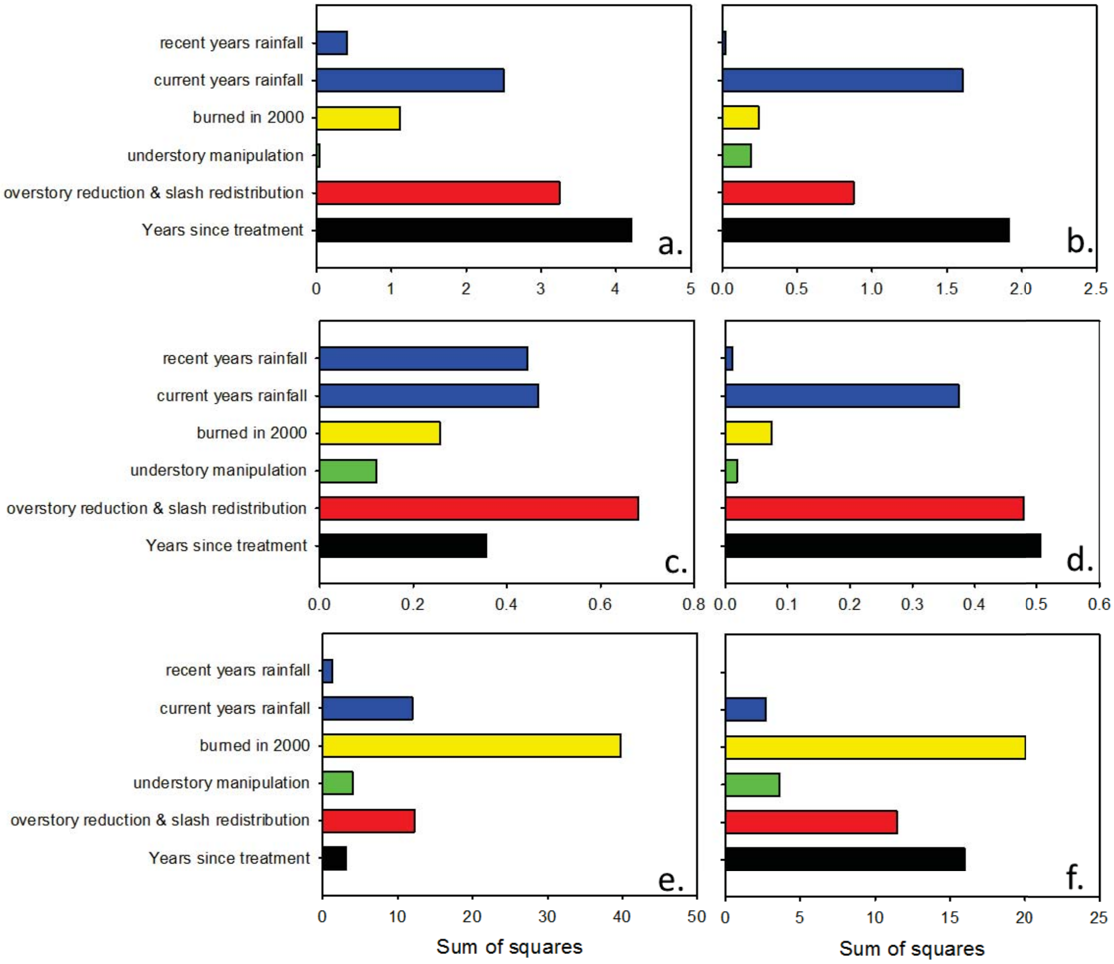


Figure 27. Relative influence of several determinants of understory, bare ground, and litter relative cover in pooled small-scale restoration studies. "Recent years rainfall" reflects current + previous two years precipitation. "Understory manipulation" pools together effects of seeding, raking, straw mulching, and imprinting. a. Total understory relative cover $R^2 = 0.68$, b. Total herbaceous cover $R^2 = 0.52$, c. Total grass cover $R^2 = 0.39$, d. Total forb cover $R^2 = 0.53$, e. Bare ground cover $R^2 = 0.69$, f. Litter cover $R^2 = 0.67$.

compared to drier years (0.06). During years receiving > 290 mm of rainfall, 3+ year old treatments supported higher cover (0.13) compared to younger treatments (0.08). Finally, in 3+ year old treatments in years with > 290 mm of rainfall, there was a second effect of current year rainfall: cover was much higher if total rainfall fell between 290-373 mm (0.22) than if it equalled or exceeded 374 mm (0.13).

Grass cover exhibited some similar patterns to those above. Greater reduction of overstory promoted grass cover (0.8 if 70% or more of overstory was thinned, 0.05 if less overstory thinning had occurred). Within the aggressively thinned treatments, recent precipitation (current year total + previous 2 years total) had a negative effect on grass cover; if the past three years rainfall met or exceeded 752 mm cover was only 0.08, compared to 0.18 if rainfall was lower. This result seems to suggest that prolonged relatively dry periods (this dry period was 2002-2004) do not necessarily cause major declines in grass cover; we note that this is based only on 7 observations, thus we do not have high confidence in this result. If an aggressively thinned plot did receive ≥ 725 mm of rain in the past three years (true of every three year period except 2002-2004), current year rain > 290mm promoted cover (0.09) compared to drier current years (0.05).

Forbs also showed a dependency on time since treatment: cover was lower if treatments had been applied within 2 years (0.015) compared to older treatments (0.035). In these 2+ year old treatments current year rain was a strong influence. If current year rain equaled or exceeded 290 mm, forb cover was quadrupled (0.04) compared to drier years (0.01). Finally, observable only in 2+ year old plots in years receiving ≥ 290mm of rain, the most aggressive overstory treatment, applied by Jacobs and Gatewood (1999), resulted in twice the cover of forbs (0.06) compared to other overstory treatments (0.03).

Bare ground was most strongly influenced by the Cerro Grande fire, if a plot had been burned, the average relative cover of bare ground was 0.58 compared to 0.25. In those plots that were unburned, the less aggressive overstory reductions (controls and

Chong 2003 protocol) resulted in almost twice as much bare ground (0.34) compared to more aggressive reductions (0.19). In unburned, aggressively thinned plots treatments that were younger than 1 year exhibited greater bare ground (0.27 vs. 0.17).

Litter was also most strongly affected by the Cerro Grande fire, as burned plots had about half the litter cover (0.25) and unburned plots (0.49). In unburned plots, there was an age of treatment effect whereby plots older than a year had much more litter (0.53) than young plots (0.39). Finally, in these unburned, older plots, more aggressive overstory reductions promoted litter (0.59 vs. 0.42).

SCPN dataset - In the SCPN data, none of the considered factors explained a significant amount of variation ($p \leq 0.05$) in soil aggregate stability. However, several suggested a trend ($0.05 > p \geq 0.10$). Calendar year, forb cover, canopy closure and the canopy closure-by-thinning interaction all showed a trend, suggesting they could influence aggregate soil stability (Table 3, Fig. 28A, C and D). The trend of the interaction effect (Table 3) suggests that the regression coefficient of soil stability on canopy closure is different between the thinned and not-thinned plots. Thus, we conducted two post-hoc regressions of soil stability on canopy closure for the thinned and not-thinned plots. We found a significant positive relationship in the not-thinned plots, but no significant relationship in the thinned plots (Table 3, Fig. 28D), suggesting that thinning disrupts the positive effect of canopy closure on soil stability. Finally and somewhat surprisingly, we did not find any evidence that thinning influenced soil stability, other than by decreasing canopy cover (Table 3; Fig. 28B).

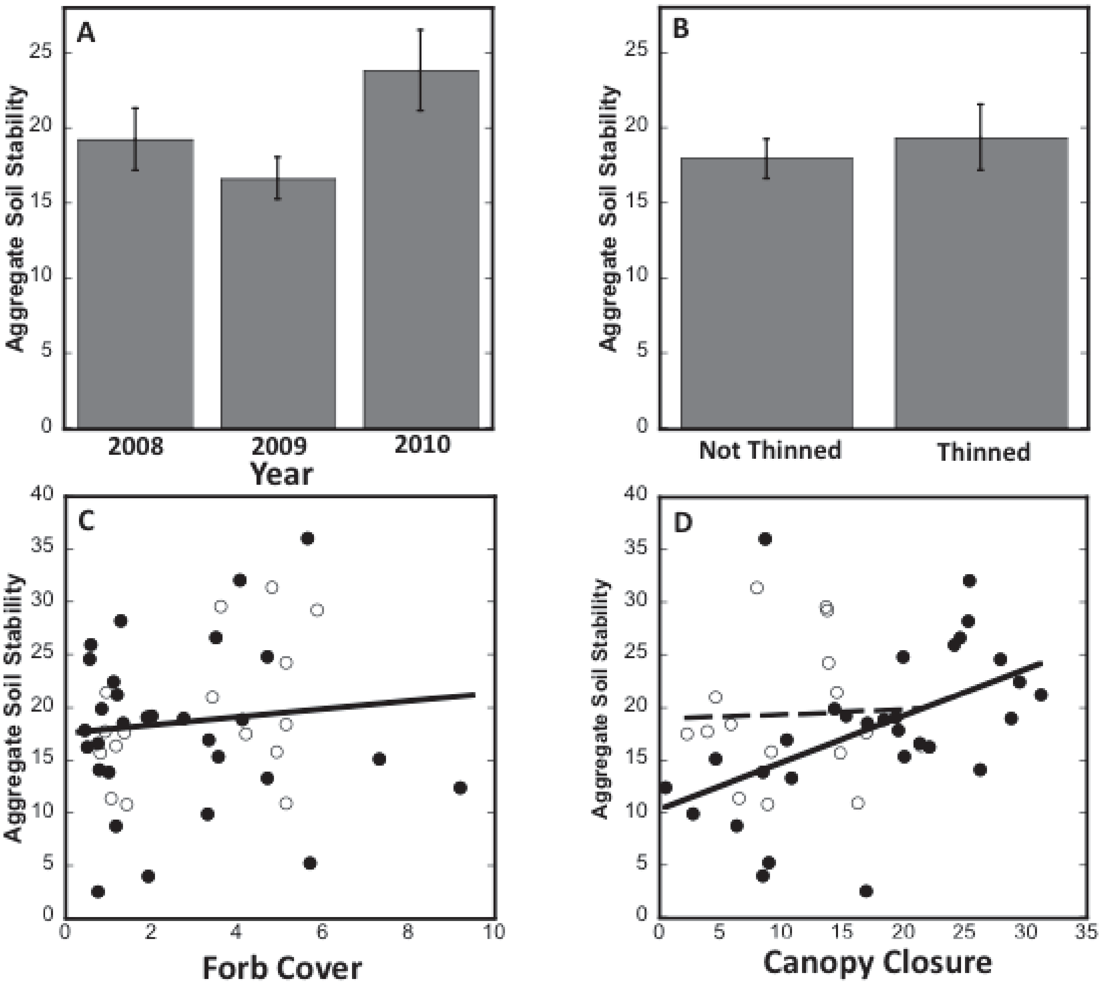


Figure 28 Bar graphs and regressions showing the influence of year (A), thinning (B), forb cover (C) and canopy closure (D) on aggregate soil stability. Because of the canopy closure x thinning interactions, we showed regression lines on the thinned and unthinned plots. The solid line and solid circles are the regression on not-thinned plots and the dashed line is the regression in the thinned plots.

This lack of direct influence of thinning and slash redistribution on soil stability may be due to a number of factors. First, it is possible that thinning has counter-acting indirect effects. For example, thinning could decrease canopy cover, which further decreases soil stability. At the same time, thinning could increase forb cover (via disturbance and/or opening of the canopy), which could increase soil stability. It is also possible that thinning has no measurable effect on stability. The small size of measured soil aggregates (<1cm^3) may make measurements at larger plot scales difficult, especially if canopy and application of treatments is heterogeneous. In any 0.5 ha plot, it is possible that thinning which predominantly occurs in one portion of the plot would not affect soil stability in other portions of the plot. It is also important to note that this study was conducted over just three years, thus a longer-term study may reveal patterns not seen here.

Factor	MS	MSE	D.F.	F	p	R^2
Year	102.2	37.2	2, 39	2.7487	.0764	.08
Thinning	61.3	37.2	1, 39	1.6506	.2065	
Forb Cover	115.3	37.2	1, 39	3.1021	.0860	.05
Canopy Closure	125.5	37.2	1, 39	3.3760	.0738	.05
C. Closure-by-Thinning Interaction	132.1	37.2	1, 39	3.5557	.0668	.05
C. Closure (Thinned)	1.0	45.2	1, 14	0.0212	.8862	
C. Closure (NotThinned)	422.6	47.6	1, 28	8.8844	.0059	.24

Table 3. The F-table for all the factors in our final statistical model as well as the post hoc regressions in the thinned and not-thinned plots. R^2's were reported for trends ($0.05 < p \leq 0.10$) and significant outcomes ($p \leq 0.05$).

Finally, our results do not necessarily suggest that thinning does not influence erosion. Soil aggregate stability is not a direct measurement of realized erosion or sediment loss. Soil aggregate stability is related to the ability of soil aggregates to resist erosive forces. However another contributing factor to actual erosion rates is erosivity, the ability of wind or water to move sediment which is not accounted for by measures of soil aggregate stability. Slash on the soil surface will have a strong tendency to slow overland flow, decreasing the erosivity of water. We also examined the effects of thinning and slash redistribution on sediment loss or gain using multi-year data from the LTER/JRM transects.

LTER/JRM transects - In the LTER/JRM transects, we found both year and transect explained a significant amount of variation in the composition of species (Fig. 29A) and functional groups (Fig. 29B). The NMDS plot is used to illustrate our results obtained using perMANOVA. Specifically, the NMDS plot shows: 1. A similar trajectory through time in all transects, i.e., all transects changed in similar ways, and 2. All transects maintain a similar ecological distance relative to other transects through time, i.e., if a pair of transects started very different in 1999 they remained different in 2003. This indicates that both year and transect identity explain variation in vegetative community composition.

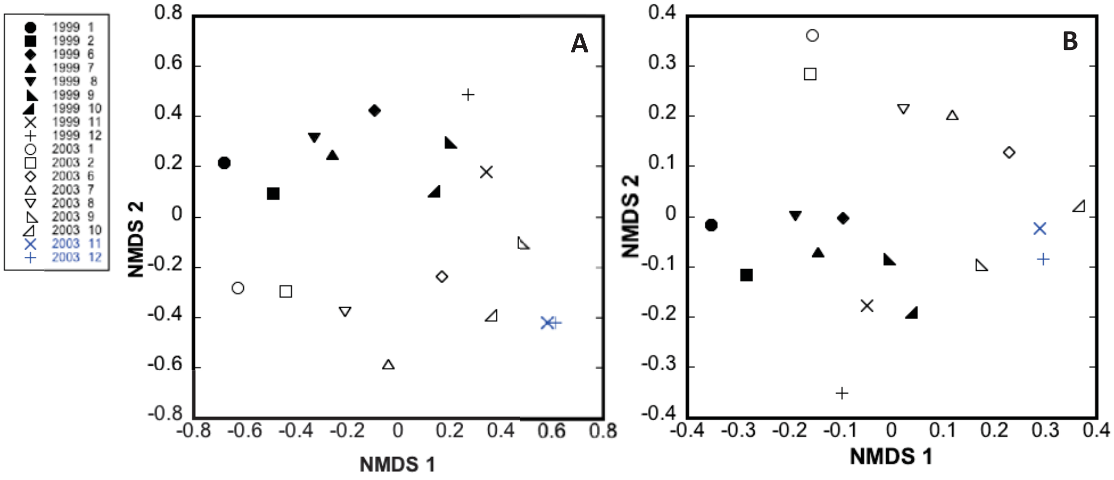


Figure 29. NMDS plots of the composition of live species (A) and live functional groups (B). The solid shapes are communities from 1999 and the hollow shapes are communities from 2003. Each shape is a transect. The NMDS thus shows how community composition changes across years and transects.

When we examine how the communities changed from 1999 – 2003, several univariate vegetative metrics changed from pre- to post 2002 drought. Total vegetative abundance decreased by ~22% after the drought (abundances for ’99 and ’03 were 96.6, and 75.1, respectively; $F_{1,8} = 52.7062$, p < .0001). Examining absolute abundance of functional groups, we saw a 62% decrease in grass ($F_{1,8} = 10.4405$, p = .0120), a 36% reduction in trees ($F_{1,8} = 11.5915$, p = .0093) and an 81% reduction in sub-shrubs ($F_{1,8} = 20.3794$, p = .0020; Fig. 30A). There was also a trend suggesting a 33% reduction in shrubs ($F_{1,8} = 4.3119$, p = .0715). We did not detect a difference in total forbs or cacti (Fig. 30A).

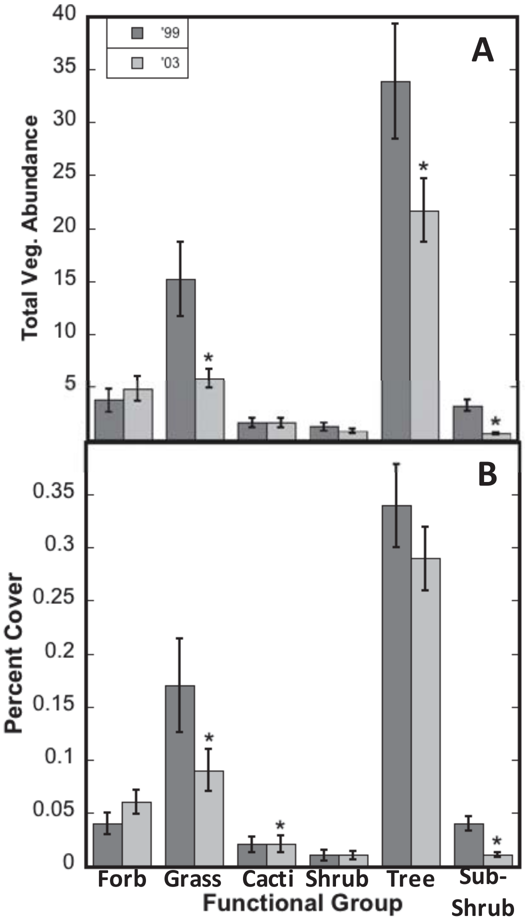


Figure 30. Bar graphs showing the mean and standard error for total (A) and percent (B) abundance of individual functional groups. Asterisks denote significant differences between years. Even though it is difficult to tell from the graph, the percent cacti is different in the two years (means are .0186 and .0249 for 1999 and 2003, respectively).

In examining the percent abundance of functional groups between years, we found multiple significant differences. There was a 49% decrease in grass cover and 74% decrease in sub-shrubs from 1999 to 2003 (Fig. 30B). We also found a trend suggesting a 25% increase in the cover of cacti (Fig. 30B). Based on the drought of 2002, the decrease in total vegetative cover from 1999 to 2003 is not surprising. Interestingly, there seem to be differential mortality among functional groups, such that the proportion of grasses and sub-shrubs decreased, while cacti increased (Fig. 30B). These results provide evidence to support the idea that although drought may induce transitions from productive woodlands to open woodlands or savannas, it can compromise herbaceous understories as well.

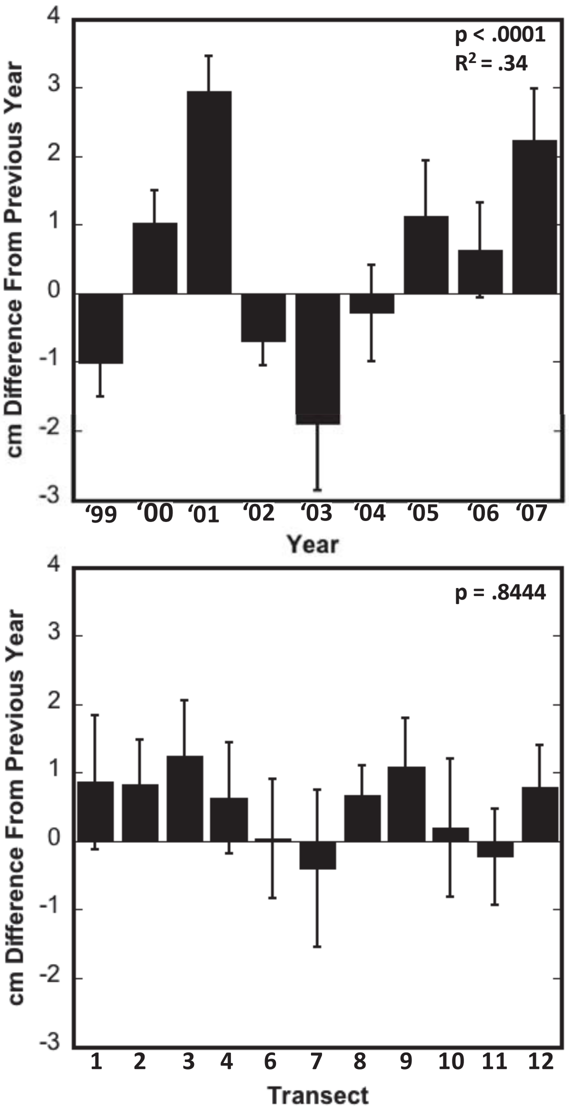


Figure 31. Bar graphs showing how erosion differed across years (A) and transects (B). Bar heights and error bars represent mean change in soil level from the previous year ± 1SE. Negative and positive values indicate erosion and deposition, respectively.

We found a significant effect of year ($F_{8,80} = 5.5940$, $p < .0001$, $R^2 = .34$), but not transect ($F_{10,\ 80} = 0.5563$, $p=.8444$, $R^2 = .04$) on erosion (Fig. 31 A, B). The two years with the greatest erosion (i.e. the largest negative change in soil level from the previous year) were years 1999 and 2003 (Fig. 31 A), which happen to be the same years in which vegetation was measured. This is somewhat challenging in studying the link between vegetation and erosion. To examine the relationship between variables, it is ideal to have variation in both the explanatory and response variables. An unfortunate coincidence minimized the variation in erosion in our analyses below.

We found mixed results when examining the relationship between the community composition (NMDS scores) and erosion. In our model, which included both year and NMDS scores (NMDS 1_{comp} & NMDS 2_{comp} = the first two axes in an ordination that included vegetation data from 1999 and 2003 data) as summary variables of community composition species, we found a significant effect of NMDS 2_{comp} ($F_{1, 14} = 4.85$, p = .0448, $R^2 = .25$) and a moderate effect of year ($F_{1, 14} = 3.4307$, p = .0852) in explaining variation in erosion. In analyzing the '99 data ($NMDS1_{99}$ & $NMDS2_{99}$ = NMDS scores generated using just vegetation data from 1999), we found a (weakly) significant relationship with $NMDS2_{99}$ ($F_{1,6} = 5.36$, p = .0599, $R^2 = .42$). Similarly, in '03 (NMDS 1_{03} & NMDS 2_{03} = NMDS scores generated using just vegetation data from 2003), we found a significant regression of $NMDS2_{03}$ on erosion ($F_{1,6} = 13.34$, p = .0107, $R^2 = .68$). It is important to note that because they were generated for three distinct data sets, NMDS 2_{comp}, NMDS 2_{99}, NMDS 2_{03} may or may not indicate similar community compositions in 1999, 2003 and the composite data set (1999 and 2003 together). Regardless, in examining how individual species may relate to the NMDS axes, we found that *Bouteloua gracilis* abundance consistently correlated with all three NMDS 2 variables. Interestingly, in all three analyses, we found a consistent relationship between species composition (NMDS scores) and erosion, suggesting that plant community composition contributes to variation in erosion across space and time.

In addition to species composition, we also analyzed the community at the level of functional groups and examined how this community metric may relate to patterns in erosion. Across all analyses, we found no relationship between plant functional group and erosion. This was somewhat surprising given the above results on species composition. We are not sure why there were significant relationships with species composition, but not with functional group composition or individual functional groups. Perhaps there are a select few species within a functional group that influence erosion, but the other members of the functional group do not influence erosion. If this were the case, looking at species composition would show patterns, but analyzing the functional group as a whole would not. Another possibility is that there could be particular combinations of species that influence erosion, which would be diluted and hence not perceptible in an analysis of functional groups.

In addition to vegetative characteristics water availability may influence erosion. Using the NOAA website (http://vlb.ncdc.noaa.gov/temp-and-precip/time-series/index.php?parameter=pdsi&month=12&year=2012&filter=1&state=29&div=6) to obtain 3-month average (June, July and August) Palmer drought severity indices (PDSI's) for New Mexico region six, for each year of this study, we examined the potential correlation between PDSI and erosion. We chose three-month averages as this time frame would encompass both the measurement of both vegetative characteristics and erosion. 2003, which saw the greatest decrease in soil elevation (i.e. the greatest erosion) also had the highest drought severity (-4.32). However, 1999, which had the second greatest loss of soil, had a normal PDSI (0.05). Further, 2001, which had the highest soil aggregation, also had a normal PDSI (0.01). Thus, at least for this data set, there does not seem to be a clear relationship between drought severity and erosion.

Summary & Projections

The trajectory of piñon-juniper woodlands in Bandelier National Monument is divergent from stated desired conditions. However, it is important to note that these woodlands are heterogeneous, and not all sites have the potential to support the same desired outcome. Woodlands would have interfingered with ponderosa pine savannahs which may have played a large role in spreading frequent fire to grassier open piñon-juniper woodlands and savannas. Highly productive persistent woodlands would have been subject to low frequency crown fire. Generally speaking, many of the woodlands and savannas recently supported too much tree canopy compared to desired conditions, and some of them support much less herbaceous cover than desired. The current woodlands can be thought of as a mosaic of persistent, unproductive woodlands (averaging 21% or less tree canopy cover, up to 45% bare ground cover, and < 5% herbaceous cover), persistent productive woodlands (averaging 38% or more tree canopy cover, 17% or less bare ground cover, and < 5% herbaceous cover, open woodlands (20% tree cover, 55% litter cover), and thinning-induced savannas which have potential to support ground fire (averaging 7% or less tree cover, and up to 32% herbaceous cover).

The first characteristic, overgrown canopy, is reversing due to three drivers: 1. intentional canopy reductions (thinning), 2. fire and 3. drought mortality of trees. Thinning may induce ecosystem state transitions which are favorable, but are probably best applied to situations where the large majority of trees are younger rather than older. Stands that are primarily young are likely stands that have thickened relatively recently, and would change in a desired trajectory if thinning was applied. These areas are more likely to be former savannas or former open grassy woodlands (Jacobs et al. 2008). There are also situations where thinning is not highly likely to promote a transition, i.e. unproductive woodlands. This was actually expected to some degree. The watershed-scale thinning study attempted to avoid such areas by selecting mesatops with favorable soil types, gentle slopes, and little exposed bedrock. However, individual transects were heterogeneous mosaics of productive and unproductive types. Treatments were applied uniformly, regardless of individual transect characteristics, in the process treating some unproductive woodlands. This allows us to experimentally demonstrate that less productive woodlands are less responsive to canopy treatment. There is comparatively little data on the effects of fire in piñon-juniper woodlands of Bandelier, though there were some impacts in the Dome fire of 1996 and the Las Conchas fire of 2011. Thinning and drought mortality make stand destroying crown fires less likely in the future because affected canopies are less continuous. The 1996 Dome fire encompassed thousands of hectares of piñon-juniper woodlands, apparently creating some conversions to shrublands, grasslands or ruderal herbs (Fig. 18). However most fire-impacted stands within the Dome fire boundary in the Bandelier fire atlas remained as woodlands, and apparently did not experience crown fire, though that would be the expected normal outcome of a fire in a piñon-juniper woodland. Another dataset mapping fire severity does not include most of the woodlands in question suggesting that burning was low severity or patchy, allowing the woodlands to persist. Another reason why there may have been relatively little conversion of woodlands to other vegetation types is that a large proportion of these woodlands in question are codominated by *Juniperus deppeana* and *Quercus undulata,* both resprouters post-fire. These dominant species are atypical of most of the woodlands in the Monument. Virtually all of these stands lost mature piñon pine in between 1981 (before the fire) and 2004 (after the fire), but these transformations are almost certainly due to the drought rather than the fire. Drought mortality in the tree canopy is the most problematic of these three drivers. Despite that canopy has been reduced overall, we cannot confidently say that the woodlands are moving closer to desired conditions because mature individuals of the previously co-dominant *Pinus edulis* have been lost. The future of this species is uncertain. Currently it persists in the understory as saplings, but because the future climate is likely to bring additional and increasingly warm droughts (Williams et al. 2012), it is not clear that this species will rebound (Adams et al. 2009). Just as troubling, at least in some stands, is the possibility of juniper mortality. While juniper mortality is not addressed in any existing dataset, it is a troubling possibility in the current drought that is currently being observed in some parts of New Mexico. If both species are lost, affected stands will effectively undergo a shift in growth form dominance to grasslands and/or shrublands. Transitions such as these are clear departures from desired conditions, and may force managers to reevaluate desired or possible future trajectories.

The second major element of these ecosystems is the understory and its control over the accelerated erosion problem (Davenport et al. 1996). The effect, at the scale of a whole watershed, of a thinning and slash redistribution study on promotion of understory herbs is well documented (Chong 1993, Hastings et al. 2003, Jacobs 2002, Loftin 1999). We interpret successful promotion of understory using slash redistribution as an ecosystem state change – a transition to a structural and functionally distinct ecosystem. This state change is very much in line with the desired conditions of more open woodlands with herbaceous understories. Our examination of long-term transects confirms that greater development of a grassy understory is associated with less emitted sediment. In our analysis of the SCPN monitoring plots, greater soil aggregate stability seems to be promoted by greater canopy cover, despite that thicker canopied woodlands tend to yield more sediment in erosion events (Hastings et al. 2003). Though this was not the expected result, it likely indicates that most sediment yield in Bandelier is regulated more by the connectivity of patches which can intercept and store runoff (Davenport et al. 1996) than by the erodibility of soil *per se*. In general, the literature and our analyses of existing data indicate that: 1. herbaceous understories can be promoted by slash redistribution, though this does not always happen, 2. Erosion is reduced by herbaceous understories and slash, 3. Promotion of herbaceous understories and reduction of erosion are consistent with desired trajectories for the piñon juniper ecosystem.

Since a large proportion of Bandelier's piñon-juniper woodlands have undergone thinning and slash redistribution, can the understory stratum of these woodlands be said to be on a desired trajectory? There are reasons to be optimistic, but recent drought conditions cast some doubt on the permanence of older treatments, and whether similarly successful results will continue to be replicable. Our new contribution here is to demonstrate that understory productivity is just as tightly controlled by climate as by overstory reduction and slash redistribution. A fortuitous redistribution of slash during a wetter than average period is most likely to promote understory production because two constraints (adequate moisture, and unstable soil surfaces) are simultaneously omitted. However for reasons poorly understood, wet years in the recent past seem to lead to lesser understory production. Long-term transects suggest that established herbaceous understories can be compromised by extreme drought, and that proportionately grasses may experience more cover loss than the tree canopy (Fig. 30). Other results from the small-scale thinning plots of Jacobs and Gatewood (1999) seem to directly conflict with this observation. Most of the grass species would be able to rebound from a drought die-back due to vegetative growth from surviving patches, but it is not known how resilient these herbs are to repeated drought events which may be more common in the future (Karl et al. 2009). Some short-lived forbs produce large quantities of long-lived seed in favorable years, thus may retain considerable resiliency. As drought eliminates more and more canopy, the understory will become increasingly important. Since herbaceous plants are less long-lived, and leave less recalcitrant residues, we can expect an emerging dynamic ecosystem that varies strongly from wet years to dry years. We would conjecture that very high herbaceous biomass would be possible in wet years, whereas much reduced cover and accelerated erosion could characterize drought years.

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Appendix: Dataset assessment and shapefiles

List of datasets identified by Bandelier NM staff, and status in vegetation change assessment

All vegetation data identified in Statement of Work were reviewed, in addition to some other vegetation datasets identified in communications with Monument scientists. Some ancillary data were reviewed if deemed potentially useful to the vegetation change assessment.

Datasets reviewed for NRCA by Bowker & Smith 2014.

1. Allen (1989) Vegetation Map (1981 imagery) a.k.a. "Ecopatch" - Analyzed in vegetation change assessment.

2. Frijoles watershed "Fieldpoint" data (Allen 1989) – Analyzed in vegetation change assessment.

3. Muldavin et al. (2011) NMNHP Vegetation map & associated ground data - Analyzed in vegetation change assessment.

4. Fire effects monitoring plots (1991-present) – Plots pertaining to Ponderosa pine ecosystems analyzed in vegetation change assessment.

5. Oertel NMNHP validation of vegetation map (2002) – Assessed and rejected for analysis in vegetation change assessment.

6. *Various elk herbivory photosets* – Data assessed and rejected for further analysis.

7. *Elk exclosure data (2000~2005) & associated ancillary data* – Vegetation data analyzed in vegetation change assessment. Ancillary data assessed and not analyzed.

8. Valles Caldera National Preserve browsing study (Kranz 2001) – Dataset reviewed and rejected for further analysis because it is primarily conducted outside of Bandelier.

9. Valles Caldera National Preserve elk observations (Kranz 2001) - Dataset reviewed and rejected for further analysis because it is primarily conducted outside of Bandelier.

10. Random surveys of elk browsing on shrubs (2002 & 2005) – Data analyzed in vegetation change assessment.

11. Targeted survey of refuged shrubs (2006) – Data analyzed in vegetation change assessment..

12. Cross-country ski area post fire aspen study – Data assessed as lower priority, not analyzed in vegetation change assessment.

13. Post-Dome fire transects – Data analyzed in vegetation change assessment.

14. La Mesa Fire vegetation plots (Foxx, 1977-present) – Original author is analyzing and preparing for publication; was not used in vegetation change assessment.

15. Chong small-scale piñon-juniper restoration study – Data analyzed in vegetation change assessment.

16. Jacobs small-scale piñon-juniper restoration study -Data analyzed in vegetation change assessment.

17. Loftin small-scale piñon-juniper restoration study -Data analyzed in vegetation change assessment.

18. Piñon-juniper paired watershed restoration study (1996-2010) & Archaeology site subset– Analyzed in vegetation change assessment.

19. LTER/JRM transects & ancillary erosion data – Vegetation and erosion data analyzed in vegetation change assessment.

20. Southern Colorado Plateau Network piñon-juniper plots – Analyzed in vegetation change assessment.

Datasets identified by Bandelier National Monument, but not reviewed for NRCA

21. Frijolito watershed studies – Analysts not supplied with this data, it was not reviewed

22. Piñon-juniper demography plots – Analysts not supplied with this data, it was not reviewed.

23. Burro exclosures – Data assessed as low priority because grazing is a minor driver of change currently.

Dataset Rating System

Each dataset is evaluated on a scale of 1-5 in the categories: Sampling design & statistical robustness, Temporal resolution, Spatial reference, Management relevance, Uniqueness. In addition, a summary score is provided.

Sampling design & statistical robustness: Rates the degree to which a dataset is compatible with a robust statistical analysis capable of testing a hypothesis or enabling modeling or prediction of a phenomenon.

1. Data strictly qualitative and incompatible with statistics; or data are semi- quantitative or quantitative but suffer multiple major departures from the conditions listed under rating 5.

2. Semi-quantitative or quantitative data which suffer one major departure, or 3 or more minor departures from the conditions listed under rating 5.

3. Semi-quantitative or quantitative data which suffer two minor departures from the conditions listed under rating 5.

4. Semi-quantitative or quantitative data which suffer one minor departure from the conditions listed under rating 5.

5. Semi-quantitative or quantitative data which satisfies all of the following conditions: a. Study design optimized to test one or more hypotheses and discriminate among competing hypotheses, b. Adequate replication to allow sufficient statistical power; no serious pseudo-replication, c. If an experiment, the study employs a proper unbiased control; if an observational study adequate variability in independent variables or predictors is attained, d. Chosen measurements are well-matched to the hypotheses, precise, and accurate.

Temporal resolution: Independent of experimental design quality and statistical robustness, the rating assesses the ability of a dataset to establish trends over time. Takes into account the period of time sampled, the frequency of sampling, and the consistency and comparability of methods used in different years. The following terms are used below: Study duration - Multiyear study (2 – 5 years), Long term study (> 5 years); Sampling interval - Seasonal (> 1 per year), Annual (1 per year), Frequent (every 2-3 years), Infrequent (> 3 years between samples).

1. No temporal resolution; sampling was conducted once only.

2. Repeatedly measured data which suffer one major departure, or 3 minor departures from the optimal conditions listed under rating 5.

3. Repeatedly measured data which suffer two minor departures from the optimal conditions listed under rating 5.

4. Repeatedly measured data which suffer one minor departure from the optimal conditions listed under rating 5.

5. Repeatedly measured data which satisfies all of the following conditions: a. Long-term study, b. At least frequent sampling, as deemed appropriate for study objectives, c. Methods used do not vary over time.

Spatial extent and reference: Independent of experimental design quality and statistical robustness, this rating assesses the quality and completeness of the documentation of the spatial location of samples or studies and the degree of generality afforded by the spatial distribution of samples. Data with no location data at all are not considered.

The following definitions are used: location data quality: high (Each unique sample completely documented with representative GPS coordinates such as plot center or transect end points), intermediate (General locations identified with GPS coordinates, but not individual samples), low (lacking precise coordinates); spatial extent: high (samples are dispersed in a reasonable, unbiased fashion across a large area such as a vegetation type or jurisdiction), intermediate (samples are largely concentrated in a local area such as a watershed or mesa OR samples are dispersed across a large area with distinct unjustified clumping or other possible biases), low (entire study concentrated in a single location such as a 1 ha split-plot).

1. Location data quality and spatial extent both low
2. Both location data quality and spatial extent of intermediate or worse quality; one of these properties is of low quality
3. Both location data quality and spatial extent of intermediate quality
4. Either location data quality or spatial extent of intermediate quality
5. Both location data quality and spatial extent of high quality

Uniqueness: Encompassing both relative dataset quality and content, this rating evaluates the degree to which the study contains a unique perspective or design or information not found in other datasets from the area. For example, many studies document vegetation change in response to stressors such as fire or herbivory, but relatively few studies document both this change and the outcome in terms of faunal communities or ecosystem processes. Studies which do both would score highly.

1. Multiple datasets of equal or better quality exist in the area of interest addressing similar questions using similar techniques
2. A dataset equal or better quality exists in the area of interest addressing similar questions using similar techniques
3. Other datasets exist in the area of interest addressing similar questions, but the present study is of superior quality
4. There is up to 50% overlap with other studies in the area of interest in terms of questions and techniques, but there is significant new subject matter or application of new techniques or questions which go beyond preceding work.
5. There is < 25% overlap with other studies in the area of interest in terms of questions and techniques, and the present study is composed primarily of new subject matter or application of new techniques or questions which go beyond preceding work.

Summary Score: Takes into account all of the above with the realization that it is unrealistic to expect a dataset which maximizes all components at once. Based on the assumption that the value of a dataset is limited by its most severe weakness, a summary score will reflect the lowest category score among: 1. sampling design & statistical robustness, 2. Maximum (temporal resolution **OR** spatial extent and reference), 3. Uniqueness

1. Allen (1989) Vegetation Map a.k.a. "Ecopatch"

Date: Polygon data based on 1981 aerial imagery.

Location: Entire park.

Data type: Quantitative and Qualitative; Canopy cover and community type.

Data Source: Allen CD. 1989. Changes in the landscape of the Jemez Mountains, New Mexico. Ph.D. Dissertation, University of California, Berkeley.

Description: This study mapped plant communities in Bandelier National Monument and surrounding areas. One small portion of the Monument in the Northwest is excluded, as is the Tsankawi unit.

Known Problems & Caveats: Mapped using stereoscope methods and cellophane tracing of aerial imagery which was state-of-the-art at that time.

Rating: Sampling design & statistical robustness: ***5 (Does not apply; this is a map product)***; Temporal resolution: ***1,*** Spatial extent & reference: ***5***, *Uniqueness:* ***5 (It is rare for a single location to have vegetation mapped more than once. This data provides a snapshot during the wet 1980s)***. Summary: 5

2. Frijoles watershed "Fieldpoint" data (Allen 1989)

Date: 1987

Location: Frijoles watershed

Data type: Quantitative and Qualitative; Ground measurements of slope, aspect, canopy cover, plant cover by species, bare rock cover, ecopatch type (a community classification) and several other incomplete fields referring to fire history and other site characteristics.

Data Source: Allen CD. 1989. Changes in the landscape of the Jemez Mountains, New Mexico. Ph.D. Dissertation, University of California, Berkeley.

Description: This study sampled 969 ground plots distributed in a spatially balanced way across the entire Frijoles watershed. Data was partly meant to be a validation of Allen 1996 vegetation map, but contain unique information not relevant to the map.

Known Problems & Caveats: Some data fields are sparsely filled.

Rating: Sampling design & statistical robustness: *5* ***(Excellent sample size and well-dispersed plots capturing multiple gradients)***; Temporal resolution: ***1,*** Spatial extent & reference: ***5,*** ***Uniqueness : 4***, Summary: 4

3. Muldavin et al. (2011) NMNHP Vegetation map & associated ground data

Date: Field surveys initiated 2003; validation data spanned 2007 – 2008; mapping based primarily on 2004 aerial imagery.

Location: Entire park.

Data type: Quantitative and Qualitative; Cover class data by species, species lists, site attributes, DBH for trees < 20cm, fuel transects.

Data Source: Available on SCPN website; Muldavin E, Kennedy A, Jackson C, Neville P, Neville T, Schulz K, and Reid M. 2011. Vegetation classification and map: Bandelier National Monument. Natural Resource Technical Report NPS/SCPN/NRTR— 2011/438. National Park Service, Fort Collins, Colorado.
Description: A total of 711 plots were collected for the classification and mapping portion of the project. Of these, 210 were standard plots, 467 were quick plots, and 34 were observation points. Standard plots: vascular plant species list, cover estimated by species using a modified Domin-Krajina Scale, DBH of trees. We measured diameter at breast height (DBH) for trees greater than 20 in in diameter. Trees below 2 in (5 cm) in diameter were broken down further into two height classes: those taller than breast height (4.5 ft; 1.37 m), and those shorter. Of the 210 standard plots, 151 were monumented fuel plots (Brown 1974; Lutes et al. 2006). Quick plots record only the dominant and most common species in each function al group. Observation points contain mostly qualitative data on an occurrence, including: location and community type, which may or may not include photos. The accuracy assessment consisted of 211 quick plots.

Muldavin also supplied dead tree canopy cover which was not part of his original data products.

Known Problems & Caveats: NA

Rating: Sampling design & statistical robustness: ***5 (Very large number of stratified random points)***; Temporal resolution: ***1,*** Spatial extent & reference: ***5,*** *Uniqueness:* ***5 (The sheer volume of data is matched only by Allen (1989). This dataset, however is post 2002-2003 drought adding to its value)***. Summary: 5

4. Fire effects monitoring plots (1991-present)

Date: mid 1990's - present

Location: Throughout park, multiple vegetation types.

Data type: Quantitative; Shrub density, Density and cover of understory, density of overstory trees by size class.

Data Source: Laura Trader

Description: These are a series of fire monitoring plots containing data on vegetative characteristics. There are 174 plots in total, spanning many ecosystem types, most of which have experienced burning. In general, the plan for each plot (in the case of planned fire events) is to measure the plot before the burn, then again in year 1, year 2, and year 6. In the case of unplanned events, pre-burn data may not exist. Some plots reburned in which case this cycle repeats. This dataset can be divided into sets. For example, there are four different sets of Ponderosa Pine plots. In the La Mesa fire there are 19 plots, each sampled 3 – 5 different years. There are 10 upper elevation plots, each sampled 3 – 6 times. There are 19 low elevation plots, each sampled 1 – 7 times, Finally, there are 10 mechanically thinned plots, each sampled between 2 -4 times. A subset of plots in each of the four sets were not disturbed, which helps to serve as a control in measuring effects of disturbance. Disturbed plots were surveyed just before distsurbance, which also contributes to the control.

Known Problems & Caveats: The main issue with this dataset, which substantially constrains clear conclusions, is the uneven sampling across years. These data consist of several plots surveyed across a span of ~15 years. Some plots were never disturbed (i.e. they were not burned or thinned). These plots are somewhat informative as serving as a control, but their usefulness is limited because in most cases, they were not surveyed in the same years as the disturbed plots.

Another issue is the spatial heterogeneity among the burned and thinned plots, which makes it difficult to tease apart the effects of these different disturbance types and geographic variation. Burned plots were surveyed in three different areas (low elevation, upper elevation ponderosa pine and in the area of the La Mesa fire), while the thinning plots were surveyed in a separate area. Again, because of the known impact of geographic variation in shaping plant communities, this spatial clumping of disturbance plots makes it difficult to discern the effects of different disturbance types compared to geographic variation.

Rating: Sampling design & statistical robustness: ***3 (Not designed with statistical analysis in mind)***;Temporal resolution: ***3 (long-term dataset, infrequently sampled, 2002 data are partial)***; Spatial extent & reference: ***5 (the large array of plots capturing elevational, fire-induced, and thinning-induced variation is highly valuable)****; Uniqueness :* ***5 (Repeated sampling along an intensity gradient, temporal scale captures a major drought period, and study adds a human manipulation - seeding)***. Summary: 3

Fire effects monitoring plots (1991-present)

Date: mid 1990's - present

Location: Throughout park, multiple vegetation types.

Data type: Quantitative; Shrub density, Density and cover of understory, density of overstory trees by size class.

Data Source: Laura Trader

Description: These are a series of fire monitoring plots containing data on vegetative characteristics. There are 174 plots in total, spanning many ecosystem types, all of which have experienced burning. In general, the plan for each plot (in the case of planned fire events) is to measure the plot before the burn, then again in year 1, year 2, and year 6. In the case of unplanned events, pre-burn data may not exist. Some plots reburned in which case this cycle repeats.

Known Problems & Caveats: Since these are monitoring plots, many of which follow wildfire or prescribed fire events, there is no statistical experimental design. Initial year of monitoring differs by plot.

Rating: Sampling design & statistical robustness: ***5 (Selection is not random, but the large array of plots capturing elevational, fire-induced, and thinning-induced variation is highly valuable)***;Temporal resolution: ***3 (long-term dataset, infrequently sampled, 2002 data are partial)***; Spatial extent & reference: ***5****; Uniqueness :* ***5 (Repeated sampling along an intensity gradient, temporal scale captures a major drought period, and study adds a human manipulation - seeding)***. Summary: 5

5. Oertel NMNHP validation of vegetation map (2002)-

Date: 2002

Location: Throughout park, multiple vegetation types.

Data type: Quantitative, Canopy cover by major tree species

Data Source: Laura Trader; collected and complied by Rebecca Oertel. Not published or used to validate Muldavin et al. (2011) model.

Description: These are a subset of the FMP plots. The new data are canopy cover estimates of overstory trees and selected shrubs, in addition to brief verbal notes, e.g., “2 PIPO with red needles”. These were originally intended as validation for the Muldavin vegetation map (below). This snapshot dataset coincides with a severe drought in progress.

Known problems or caveats: Canopy cover data do not distinguish among dying overstory trees and living ones. Many trees would have died within the year. This dataset on its own does not contain enough points for spatial analyses such as interpolation.

Rating: Sampling design & statistical robustness: ***5***; Temporal resolution: ***1 ,*** Spatial extent & reference: ***5****, Uniqueness :* ***4 (This data adds a new measurement to an existing database, and is***

one of three large sets of point data describing canopy cover in different time periods. The unique timing of the measurement during a drought mortality event gives the data a potential that can be built upon). Summary: 2

6. Various elk herbivory photosets

a. Dome-area shrub photos

Date: November 27 2000

Location: Dome Diversity Unit (St. Peter's Dome), Santa Fe National Forest, NM; GPS coordinates for each photo supplied in excel file "Dome-Aspen-Photos-2000Nov28.xls"

Data type: Qualitative; Annotated Photos

Data Source: Steve Fettig, accompanied by Sandy Palmer

Description: set of 44 photos resulting from a 1 day reconnaissance trip, documenting various forms of elk herbivory on aspens. A datapoint consists of a location, a photo, and a short note describing the photo contents (e.g. debarking close-up, fallen over, etc.). A subset of 9 photos are associated with somewhat more developed captions. Does not seem to use a standardized sampling protocol, if so it is undocumented. May be useful for mapping instances of herbivory or spatial extent of herbivory, especially if pooled with other similar data, but value is limited without a standardized survey which allows the magnitude of the documented herbivory to be compared to some standard or control.

Known Problems & Caveats:

Rating: Sampling or experiment design: ***1***; Statistical robustness: ***1***;Temporal resolution: ***1***; Spatial reference: *5;* Summary: 1 provides supplementary documentation of herbivory, low value in and of itself to documents trends or patterns.

b. American Springs shrub photos

Date: December 15 2000

Location: American Springs, northeast of Bandelier National Park on Los Alamos National Laboratory side; GPS coordinates for each photo supplied in excel file "American-Springs-aspen-photos-2000Dec15.xls"

Data type: Qualitative; Annotated Photos

Data Source: Steve Fettig

Description: set of 36 photos, documenting various forms of elk herbivory mostly on aspens. A datapoint consists of a location, a photo, and a short note describing the photo contents (e.g.

debarking close-up, fallen over, etc.). Does not seem to use a standardized sampling protocol, if so it is undocumented. May be useful for mapping instances of herbivory or spatial extent of herbivory, especially if pooled with other similar data, but value is limited without a standardized survey which allows the magnitude of the documented herbivory to be compared to some standard or control.

Known Problems & Caveats:

Rating: Sampling design & statistical robustness: ***1***;Temporal resolution: ***1***; Spatial reference: *5;* Summary: ***1*** provides supplementary documentation of herbivory, low value in and of itself to document trends or patterns.

c. Jemez west side photos, vegetation and elk pellet and browsing data

Date: photos date from 2001, possibly September, quantitative field data date from September 18 2002.

Location: Western Jemez Mountains, Santa Fe National Forest New Mexico; There is some uncertainty whether the 7 sites documented in the 2001 photos are part of the 8 for which quantitative data are collected. They use inconsistent naming conventions. The photos are captioned as sites 1-3 FR376, sites 4-6 FR 144, and site 7 marked “Hot Rocks”. The quantitative plots are named 2, 376A, 144A, 7, 6, 144B, 378A, 3. GPS coordinates are available for the quantitative plots but not for the 2001 photos.

Data type: Annotated Photos and some quantitative including slope, aspect, pre-fire overstory, pellet counts and aspen dbh.

Data Source: Steve Fettig

Description: A set of 34 photos from 2001, documenting various forms of elk herbivory on aspens. A datapoint consists of a photo, and a short note in its title describing the photo contents (e.g. debarking, sprouts, etc.). Some photos are grouped with captions in figure format. Does not seem to use a standardized sampling protocol, if so it is undocumented. Coordinates of photo points undocumented.

There are also quantitative data gathered from 8 sites, 50+ associated photos, a methods summary, brief report outline. The linkage between the photographed sites in 2001 and these is not clear. A datapoint consists of a $30m^2$ plot in which physiography, cover of aspen sprouts and shrubs, number of pellet groups and percentage browsing by shrub species were recorded. Some photos are grouped with captions in figure format. Sampling protocol was half random & GIS assisted and half haphazard selection in field.

Known Problems & Caveats:

2001 photos rating: Sampling design & statistical robustness: ***1***;Temporal resolution: ***1***; Spatial extent & reference: *1;* Summary: 1; low value in and of itself to document trends or patterns.

2002 Quantitative data rating: Sampling design & statistical robustness: ***2***;Temporal resolution: ***1***; Spatial extent & reference: *5;* Summary: 2; Reasonable regression-type study design limited by relatively few samples; may help support more robust datasets.

7**.** *Elk exclosure data (2000~2005) & associated ancillary data*

Date: 2001-2006

Location: Bandelier's Mixed conifer exclosures

Data type: Quantitative

	2001	2002	2003	2005	2006
Aspen live count	X	X	X	X	X
Aspen dead count	X				
Canopy cover (densiometer)	X	X			
Grass & forb biomass	X	X	X		
Grass & forb count & height		X	X		
Butterfly counts		X	X		
Pellet counts		X			
Chokecherry counts		X			
Potentilla flower count		X	X		

Data Source: Steve Fettig

Description: A set of elk exclosures in mixed conifer ecosystems paired with non-exclosed controls are used in what appears to be several different studies. They are grouped here for convenience but data are rated separately. Basic data are counts of aspen although several addition types of data were included describing vegetation, butterfly use, and butterfly habitat quality. Formats are not always consistent among years, and some data were only collected in one year. A map documents their locations, but precise GPS coordinates are not documented. The exclosures are dispersed in a relatively small portion of the Cerro Grande fire. There are some statistical shortcomings. Least problematic is that the number of pairs (5) is small; this is a limitation that is difficult to avoid with larger scale manipulations like exclosures. More problematic is apparent spatial clumping of exclosures, one could question whether they are true replicates. Temporal depth of various variables differs; up to 5 samplings exist for some data.

Known Problems & Caveats: Spatial clumping in locations; unplanned burns are challenging analytically but ecologically interesting

Aspen live count rating: Sampling design & statistical robustness: ***3***; Temporal resolution: ***4*** ; Spatial extent & reference: ***2 (location data surely exist but are not in my possession, this rating cannot be finalized without this information)****; Uniqueness:* **5 (appears to be the only set of elk exclosures in the area)**; Summary: ***3***

Aspen dead count, pellet group count, and chokecherry count rating: Sampling design & statistical robustness: ***3***; Temporal resolution: ***1*** ; Spatial extent & reference: ***2 (location data surely exist but are not in my possession, this rating cannot be finalized without this information)****; Uniqueness:* **5 (appears to be the only set of elk exclosures in the area)**; Summary: ***3***

Canopy cover, grass & forb count and height, and Potentilla flower count rating: Sampling design & statistical robustness: ***3***; Temporal resolution: ***3*** ; Spatial extent & reference: ***2 (location data surely exist but are not in my possession, this rating cannot be finalized without this information))****; Uniqueness:* **5 (appears to be the only set of elk exclosures in the area)**; Summary: ***3***

Grass & forb biomass rating: Sampling design & statistical robustness: ***3***; Temporal resolution: ***4***; Spatial extent & reference: ***2 (location data surely exist but are not in my possession, this rating cannot be finalized without this information)****; Uniqueness:* **5 (appears to be the only set of elk exclosures in the area)**; Summary: ***3***

Butterfly count rating: Sampling design & statistical robustness:***3***; Temporal resolution: ***3***; Spatial extent & reference: ***2 (location data surely exist but are not in my possession, this rating cannot be finalized without this information)*****;** *Uniqueness:* **5 (appears to be the only set of elk exclosures in the area)**; Summary: ***3***

8. Valles Caldera National Preserve browsing study (Kranz 2001)

Date: July – October 2001

Location: Valles Caldera National Preserve

Data type: Quantitative, some qualitative: 1. Number of individuals browsed and unbrowsed of several shrubs eaten by elk, including aspen, and estimate of biomass consumed 2. Number of conifer saplings scarred by antler rubbing, and sapling mortality rates due to rubbing, 3. Spatial coordinates and elevation. 4.

Data Source: Douglas Krantz, supplied by Steve Fettig

Description: set of 211 transects based on a stratified sampling strategy and spanning the entire preserve. Sampling strategy is not well-documented but sample seems to be near roads and trails, and are reasonably spatially balanced across the preserve with a large number of replicates dispersed among valle/forest ecotones, forest interior, and riparian areas. The above plant data were collected at each location.

Known Problems & Caveats:

Rating: Sampling design and statistical robustness: ***4 (excellent sample size, but potential bias due to so many samples near roads)***; Temporal resolution: ***1***; Spatial reference: ***5;*** *Uniqueness:* ***4 (spatial extent and replication sets this data apart although some similar data exists elsewhere)*** Summary: ***4***.

9. Valles Caldera National Preserve elk observations (Kranz 2001)

Date: July – October 2001

Location: Valles Caldera National Preserve

Data type: Quantitative: Over 80 dated observations of elk numbers and demographics; high quality spatial data may be lacking, although an arcview file appears to map them manually and the report references hand drawn maps.

Data Source: Douglas Krantz, supplied by Steve Fettig

Description: The elk observation data was collected using automobile transects. A quadrant was surveyed in a “run” on a particular date. Different runs of the same quadrant may have used different routes. About 80 different runs are documented. Spatial location of observed animals was marked on maps. Apparently hand-drawn maps reside at the Valles Caldera National Preserve headquarters.

Known Problems & Caveats: These have not been developed as GIS layer data or converted to spatial coordinates which would greatly enhance their utility.

Rating: Sampling design and statistical robustness: ***3 (good sample size, but automobile survey creates biases, and runs were not conducted using the same protocol every time)***; Temporal resolution: ***2 (repeated sampling within one season, methods vary somewhat among sampling runs)***; Spatial reference: ***2 (not immediately useful without additional data extraction and processing)****; Uniqueness:* ***5;*** Summary: ***2***.

10. Random surveys of elk browsing on shrubs (2002 & 2005)

Special notes: Although sampling also occurred in 2006, only some data overlaps, and the two different blocks of data exhibit a different sampling strategy focusing on a different question. Thus they are evaluated separately.

Date: August 26 – September 29 2001, June 1 – July 7 2005

Location: Cerro Grande burn, Northwest Bandelier National Monument & surrounding areas

Data type: Quantitative: 76 observations in 2001, 119 in 2005. Data documents herbivory and minimum DBH and maximum height of several shrubs, in addition to pellet counts (2001 only), pre-fire overstory, slope, aspect, and canopy cover. These two datasets have a different but overlapping spatial extent.

Data Source: Steve Fettig, not published

Description: These were random points. A point consists of a 10-meter by 10-meter plot at each point in which maximum height, number of growing tips of each species (up to a maximum of approximately 200 tips) and, frequency of browsing of shrubs was recorded. This is a well-replicated study with excellent spatial extent across the Cerro Grande burn. It will be especially useful in determining the predictors of browsing intensity, which could form the basis of some sort of spatial model. Due to the random plot location, it is possible for a plot to have none of the target shrubs and therefore no browsing. Some shrubs have very sparse data because they did not occur frequently.

Known Problems & Caveats:

Rating: Sampling design & statistical robustness: ***5*** Temporal resolution: ***1(sampling was conducted in multiple years, but the same sites were never resurveyed)***; Spatial extent & reference: ***5****; Uniqueness:* ***4 (other datasets document herbivory, but this one has a far superior replications and spatial extent)****;* Summary: ***4***.

11. Targeted survey of refuged shrubs (2006)

Special notes: Although sampling also occurred in 2002 & 2005, only some data overlaps, and the two different blocks of data exhibit a different sampling strategy focusing on a different question. Thus they are evaluated separately.

Date: June 14 – July 5 2006

Location: Cerro Grande burn, Northwest Bandelier National Monument & surrounding areas

Data type: Quantitative: 109 observations. Data documents herbivory and minimum DBH and maximum height of several shrubs in refuged and un-refuged locations. Much data also exists describing binary characteristics of sites and quantitative measurements (height, diameter, etc.) of refuge lelements such as downed trees and rocks.

Data Source: Steve Fettig, not published

Description: These were a targeted sampling of refuges. A point consists of a 10-meter by 10-meter plot at each point in which maximum height, and frequency of browsing of shrubs both in the refuge and outside of it (if present) was recorded. This is a well-replicated study with excellent spatial extent across the Cerro Grande burn. Because it describes many characteristics of refuge sites that will not be available as spatial data, it has little promise for spatial modeling;

however, it could help establish a non-spatial predictive model of the characteristics of refuge sites. will be especially useful in determining the predictors of browsing intensity, which could form the basis of some sort of spatial model. All sites have at least one of the target shrubs, but many lack unrefuged individuals measured in the same year. The 2001 – 2005 data may be able to serve as reference data for unrefuged individuals.

Known Problems & Caveats:

Rating: Sampling design & statistical robustness: ***5*** Temporal resolution: ***1(sampling was conducted in multiple years, but the same sites were never resurveyed)***; Spatial extent & reference: ***5****; Uniqueness:* ***5 (other datasets document herbivory, but this one has a far superior replications and spatial extent)****;* Summary: ***5***.

12. Cross-country ski area post fire aspen study

Date: 27 July – 9 August 2006

Location: Cross country ski area, Bandelier NM.

Data type: Quantitative, Semi-quantitative and qualitative: 144 mature trees sampled, their location marked with GPS coordinates, dbh measured (minimum = 7.3), circumference debarked by elk was estimated in classes, and whether or not they had burn scars was recorded.

Data Source: Steve Fettig

Description: Appears to be linked to the shrub resprout from Cerro Grande, but documentation of the original intent of the study is lacking. It is potentially useful for determining the relationship of tree size and burn status to elk herbivory.

Known Problems & Caveats:

Rating: Sampling design and statistical robustness: ***5 (difficult to rate without knowing the hypothesis being addressed, but could be used for the above purposes)*** Temporal resolution: ***1***; Spatial reference: ***5****; Uniqueness:* ***5****;* Summary: ***2***.

13. Post-Dome fire transects

Date: 1997, 1998, 2002, 2008

Location: Dome fire area in park and in adjacent National Forest.

Data type: Quantitative; Basal and canopy cover data by species.

Data Source: Original study by A. Barclay et al.; 2000s collected and complied by Rebecca Oertel and Kay Beeley.

Description: This study established 48 transects after the Dome fire. Sites represented a gradient in burn intensity (unburned, low, medium, and high intensity). Some sites on the National Forest side were seeded with annual grasses including *Lolium*. The original study ran for 2 years. The 10 transects in Bandelier were resampled in 2002 in a drought year. The plots outside of Bandelier were resampled in 2008. Future samplings should synchronize sampling inside and outside of park if possible. The original study ran for 2 years. The 10 transects in Bandelier were resampled in 2002 in a drought year. The Whole array was resampled in 2008.

Known Problems & Caveats: Unseeded plots on National Forest side are strongly clustered on single mesa top. 2002 & 2008 sampling is disjunct across administrative boundaries.

Rating: Sampling design & statistical robustness: ***3 (Good sample size and well-dispersed plots but weak separation of administrative boundaries, fire intensity, and seeding; factors incompletely crossed; some transects are missing at times; most notably recent samplings within and outside of park boundaries are disjunct, compromising analysis options)***; Temporal resolution: ***2 (Long-term study infrequently sampled, inconsistent sampling),*** Spatial extent & reference: ***5****, Uniqueness :* ***5 (Incorporates an otherwise undocumented human manipulation and induced ecosystem state)***. Summary: 3

14. La Mesa Fire vegetation plots (Foxx, 1977-present)

Date: 1979 – 2010 (including 1998 and possibly other time points)

Location: Burnt Mesa, Escobas Mesa, Frijoles Mesa.

Data type: Quantitative; Cover data by species using a quadrat-based method.

Data Source: Original study by T. Foxx; recent monitoring by Collin Haffey & Kay Beeley.

Description: This study established 10 transects after the La Mesa fire. Design is unknown; controls may or may not exist. Only 2010 data is available. The ~30 year span in sampling is unusually rich, but there is only a small number of transects. Preparation of a final report by the original author is ongoing. Thus, there is probably little value in reanalyzing this dataset and since it is not completely available, it is not rated. Sites represented a gradient in burn intensity (unburned, low, medium, and high intensity).

Known Problems & Caveats:

Rating: Not rated.

15. Chong small-scale piñon-juniper restoration study

Date: Vegetative cover data were collected in 1992 for all blocks and in 1995 for only cut and control blocks. The control data was collected in early June 1992 and mid-July 1995. The cut treatment data was collected in April-May 1992 and late June 1995

Location: Frijoles Mesa; GPS coordinated not provided to analysts

Data type: Quantitative; cover and density of understory species and ground cover.

Data Source: Provided by Brian Jacobs. Some data published in Chong GW. 1993. Revegetation of piñon- juniper woodlands with native grasses. Pages 34-41 In: Aldon EF, Shaw DW (tech.coords.) Managing piñon-juniper ecosystems for sustainability and social needs; proceedings of the symposium 1993 April 26-30; Sante Fe, New Mexico. Gen. Tech. Rep. RM-236. Fort Collins, CO: U.S. Department of Agriculture, Forest Service.

Description: In 1992 a blocked study design was established by Geneva Chong, consisting of three types of tree overstory treatments crossed with various understory treatments. The Chong study was located just up-mesa of the Frijoles Mesa small scale study area. The tree overstory treatments included: cut and scatter slash or girdling, and control (no treatment). The understory treatments were: control, raking, seeding, seeding + straw mulch, seeding + fertilizer, seeding + fertilizer + straw mulch. Each block was assigned to one of the three overstory treatments for a total of 9 “blocks.” A block was 26m by 39m in size and was subdivided into six 13m X 13m plots. Each plot had twelve data collection points located on a grid for a total of 72 points/block. To collect vegetative cover data, a 0.5m^2 rectangular frame was centered over each point and the percent cover by species or growth form within the frame was visually estimated. The plots were resampled by Bandelier National Park staff in 1995 in the cut and scatter and control blocks.

Known Problems & Caveats: The modification of overstory canopy was minor, only enough canopy was removed to provide slash for redistribution.

Rating: Sampling design & statistical robustness: ***4 (Sound experimental design limited by low statistical power to detect interaction effects; some previous analyses have used sampling points within plots which is pseudoreplication)***;Temporal resolution: ***2 (Two thirds of blocks were resampled once, after three years)***; Spatial extent & reference: ***5 (Lacking data on the location of plots)****; Uniqueness :* ***3 (Some overlap with small-scale restoration plot, but unique treatment combinations exist)***. Summary: 3

16. Jacobs small-scale piñon-juniper restoration study

Date: Vegetative cover data were collected in 1992 for all blocks and in 1995 for only cut and control blocks. The control data was collected in early June 1992 and mid-July 1995. The cut treatment data was collected in April-May 1992 and late June 1995

Location: Garcia Mesa, Frijoles Mesa

Data type: Quantitative plant cover data

Data Source: Brian Jacobs; some data published in Jacobs BF, Gatewood RG. 1999. Restoration studies in degraded pinyon-juniper woodlands of north-central New Mexico. Pages 294-298 *in* Proceedings of Proceedings of the U.S. Forest Service RMRS-P-9.294-298.

Jacobs and Gatewood (1998) in 1994 established two sites on Garcia Mesa and Frijoles Mesa. Within each site, 15 contiguous 15m^2 plots (representing three five-plot replicates) were established with canopy removal and slash redistribution (applied in 1995) along with 3 control plots (one per replicate), which were near to but not contiguous to the other plots. One site received a 100% canopy reduction, and one received a 70% canopy reduction (similar to Loftin 1998). Each of the treatment replicates were sub-divided into 5 blocks in which understory manipulations were applied at random: thinning-slash only, imprinting (the creation of impressions in the soil surface to act as passive collection points of seeds, organic matter and water), seeding, imprinting + seeding, and raking + seeding. In 1995 a third site was established, and treated in 1996, however the understory seeding-soil preparation treatments were never applied. This site used a 100% canopy removal treatment comparable to Frijoles Mesa (BF Jacobs pers. comm.). At least some sites were sampled from 1994 – 2001, and again in 2004. All of one site, and portions of another site burned in the Cerro Grande fire, thus all data from 2000 and later reflect post-fire recovery.

Known Problems & Caveats: Replicate plots are contiguous, not true spatial replicates. Understory treatments not applied to third site. Cerro Grande burned some but not all sites.

Rating: Sampling design & statistical robustness: 3 (Contiguousness of treated plots may lead to spatial autocorrelation), Temporal resolution: 3 (Different sampling schedules make a repeated measures analysis challenging) Spatial extent & reference: *3; Uniqueness : 4 (only one of “small scale thnning studies” to employ mulitiple sites).* Summary: 3

17. Loftin small-scale piñon-juniper restoration study

Date: 1994 – 2003, except 1998

Location: Garcia Mesa

Data type: Quantitative plant cover.

Data Source: Sam Loftin, some data published in Loftin SR. 1999. Initial response of soil and understory vegetation to a simulated fuelwood cut of a pinyon-juniper woodland in the Santa Fe National Forest. Pages 311-314 *in* Proceedings of Proceedings of the U.S. Forest Service RMRS-P-9.294-298.

Description: Loftin (1998) established two treatments in 10-15 ha blocks, and overstory thinning & slash redistribution treatment, and a control. The thinning treatment was an aggressive one simulating a fuelwood harvest. All *Juniperus* were removed, and all *Pinus* < 20 cm diameter were removed resulting in a nearly complete canopy reduction. In practice it was approximately a 70% canopy removal and identical to the co-located small scale Garcia Mesa study site (Jacobs and Gatewood 1998). The treatment and control areas were divided into 5 contiguous plots, and in each plot, a 100m transect was used to monitor vegetation with regularly-spaced quadrats. These transects were sampled from 1994 – 2003, except for 1998. We used the average of quadrats along individual transects as independent observations for two reasons: 1. This makes the area sampled more similar (though still larger) to the Chong (1993) and Jacobs and Gatewood (1998) studies, 2. Although the 1-2 ha plots they sampled were contiguous, the 100m transects used to monitor vegetation were discrete in space. This entire study was burned in the Cerro Grande fire, and all data from 2000 and after reflects post-fire recovery.

Known Problems & Caveats: Design is pseudoreplicated.

Rating: Sampling design & statistical robustness: ***2 (Pseudoreplication limits inference that can be drawn from this study alone)***;Temporal resolution: ***5 (Two thirds of blocks were resampled once, after three years)***; Spatial extent & reference: *3* ***(Single site)****; Uniqueness : 2* ***(Does not significantly add to Chong or Jacobs & Gatewoods studies)***. Summary: 2

18. Piñon-juniper paired watershed restoration study (1996-2010) & Archaeology site subset

a. overall dataset

Data type: Quantitative plant cover data

Location: Frijoles Mesa

Data Source: Brian Jacobs.

Description: The "paired watershed study" examines two adjacent watersheds draining the low elevation mesa tops in Bandelier National Monument. One watershed was selected for a treatment which consisted of removal of about 70% of the trees, and dispersing the slash throughout the interspaces (Frijoles Mesa), while the other was left as an untreated control (Garcia Mesa). Each mesa was monitored along 20 100- m line transects which ran from mesa tops downslope; each can be subdivided into upper and lower 50 m segments, and have been analyzed in this way in the past (Jacobs et al. 2002). This results in a sample size of 80 transects in total. The data include pre-treatment data from 1996, and capture an extreme drought which led to mass tree mortality. Half of the thinned samples were subjected to an experimental prescribed fire in spring 2010. The transects were monitored annually from 1996 – 2000, and again in 2008 and 2010. Data collected include plant percent cover by species (broken into live and dead fractions) and ground cover. The strong spatial replication, temporal richness, and combination of imposed and unplanned ecosystem change make this an extremely high value dataset for this purpose.

Known Problems & Caveats: Using upper and lower segments as replicates might be considered pseudoreplication, also there are some good reasons for doing so.

Rating: Sampling design & statistical robustness: ***5***;Temporal resolution: ***5 (Two thirds of blocks were resampled once, after three years)***; Spatial extent & reference: *5; Uniqueness : 5 (This is an exceptionally rare large spatial scale, long-term experiment with multiple planned events [thinning & burning], and one significant unplanned event [drought]* . Summary: 5

b. archaeology subset

Date: late 1990s – early 2000s

Location: Frijoles Mesa

Data type: Quantitative; Basal and canopy cover of plants by species. Ground cover.

Data Source: Brian Jacobs; some data summarized in Jacobs BF, Gatewood RG, Allen CD. 2002. Watershed restoration in degraded piñon-juniper woodlands—A paired Watershed study 1996-1998

Description: These are a set of 20 (needs verification) transects centered on known archaeological sites within the watershed restoration study. They were sampled in 1997 - 1998, and 2000 – 2001, apparently inconsistently. Approximately half of the transects were in areas receiving a canopy reduction and slash scattering treatment, while the other half were located on the untreated control side of the watershed.

Known Problems & Caveats: In raw data made available to the analyst, the transects are not all sampled in every year.

Rating: Sampling design & statistical robustness: ***5 (Selection is not random, but the large array of plots capturing elevational, fire-induced, and thinning-induced variation is highly valuable)***;Temporal resolution: ***2 (long-term dataset, infrequently sampled, incoongruence in sampling)***; Spatial extent & reference: ***5****; Uniqueness :* ***5 (Cultural site focus is unique in thinning studies)***. Summary: 5

c. erosion bridges

Location: Frijoles Mesa

Data type: Quantitative, sediment loss & accumulation in cm elevation change

Data Source: Brian Jacobs

Description: Permanent erosion bridges were installed in specific settings in between transects in the paired watershed study. Data were collected again in 1995 for the cut and scatter and control blocks.

Known Problems & Caveats: Difficult to clearly tie to vegetation data; scale mismatch with vegetation transects.

Rating: Because there was no clear way to link these data to vegetation changes, this dataset was determined to be low priority for inclusion in vegetation change assessment. Not rated.

19. LTER/JRM transects & ancillary erosion data

Date: Vegetation data from 1991 – present; Erosion bridge data from 1999 – present.

Location: In clusters throughout park, multiple vegetation types.

Data type: Quantitative. Basal and canopy cover of plants by species. Erosion data is the product of erosion bridges, which measure soil heights from the soil bridges. By comparing current year's height to a previous height, these data provide a measure of net soil gain (aggradation/deposition) or loss (erosion).

Data Source: Kay Beely, some data published in Chong GW. 1993. Revegetation of piñon-juniper woodlands with native grasses. Pages 34-41 In: Aldon EF, Shaw DW (tech.coords.) Managing piñon-juniper ecosystems for sustainability and social needs; proceedings of the symposium 1993 April 26-30; Sante Fe, New Mexico. Gen. Tech. Rep. RM-236. Fort Collins, CO: U.S. Department of Agriculture, Forest Service.

Description: These are a set of 11 transects designed to monitor vegetation change and erosion rates, based on monitoring techniques proposed by the Jornada Experimental Range. A transect is a 300m line intercept transect on which plant basal and canopy cover is recorded. Ground cover including pumice and biocrusts is also recorded. Erosion bridges were installed at 4 locations along the transects. JRM 1,2, 5,6,7,8,9,10,11, 12 appear to have been frequently monitored for vegetation. While additional transects (JRM 3 & 4) are also used for erosion monitoring.

Known Problems & Caveats: Spatial clumping of transects might result in spatial autocorrelation. The full set of transects is seldom sampled in the same year. Sampling dates of the various transects are incongruous which makes them difficult to analyze as a set. Vegetation and erosion were not sampled in the same spatial scale. Vegetation was sampled continuously along a 300m transect, while erosion bridges were set up at four discrete locations along the same 300m transect.

Rating: Sampling design & statistical robustness: ***3 (Possibility for spatial autocorrelation, incongruous sampling)***; Temporal resolution: ***3 (Excellent temporal depth, frequent sampling spanning over 20 years, weakened by incongruous sampling times)***; Spatial extent & reference: ***3 (Plots are clumped, but clumps are well-dispersed)****; Uniqueness :* ***5 (Temporal depth and pairing of vegetation and erosion data is unique)****.* Summary: 3

20. Southern Colorado Plateau Network piñon-juniper plots

Date: 2008, 2009 and 2010

Location: Mesa top Piñon-Juniper woodlands

Data type: Quantitative, over 200 metrics of biotic and abiotic characteristics including plant cover by species and diversity, ground cover, and soil aggregate stability.

Data Source: Southern Colorado Plateau Network of Inventory & Monitoring Program. Some summary data is presented in DeCoster JK, Swan MC. 2011. Integrated upland vegetation and soils monitoring for Bandelier National Monument: 2009 summary report. Natural Resource Data Series NPS/SCPN/NRDS—2011/168. National Park Service, Fort Collins, Colorado.

Description: This is an array of 46 monitoring plots dispersed across the mesa-top pinon juniper woodlands (an ad-hoc ecological site equivalent). Plots were established in three years. Because canopy treatment is so prevalent at Bandelier, half of the plots represent treated areas. Sampling methods are well documented in DeCoster and Swan 2009.

Known Problems & Caveats: These plots are simply too young and have not been resampled frequently enough yet to be highly useful. This will change over time.

Rating: Sampling design & statistical robustness: ***5 (The more than 200 scored variables makes it difficult to form a statistically testatable a priori hypothesis of the relationship between the environment and soil stability)***;Temporal resolution: ***2 (These data were collected over a 3-year period)***; Spatial extent & reference: ***5 (These data were collected from 46 plots across the Piñon-Juniper forest type)***; *Uniqueness :* ***5 (Lots of data attempting to understand the relationship between vegetation and erosion)***. Summary: 5 ***(As this dataset accumulates temporal depth, it will be one of the highest quality in the Monument. Currently it is just a snapshot with limited value).***

Shapefiles

Location of shapefiles used for the Bandelier National Monument Natural Resource Condition Assessment.

Maldavin vegetation (see the last 3 columns of the attribute table)	3. Muldavin(2011)Vege_map_ground_data\Modified data by Bowker _ Smith 2014\Reclassification of Muldavin et al. to Ecopatch categories
Allen vegetation	1. Allen (1989) Vegetation Map a.k.a. Ecopatch\data files supplied by BAND
Major vegetation group comparisons and other analyses	Bowker _ Smith 2014 generated data\Arundell Canopy cover\Bandelier_MBowker.gdb
Fire boundaries	Accessory Spatial Data\Fire boundaries _ severity
Park boundary	Accessory Spatial Data\Park Boundary
DEM	Accessory Spatial Data\Digital Elevation Model
Elk browsing	Bowker _ Smith 2014 generated data\Aspen browsing and height surfaces\Spatial data

Appendix C: Scope of Work for Assessing the Vegetation Community Condition at Bandelier National Monument

Vegetation Community Condition at Bandelier National Monument

Vegetation change over large areas can occur naturally, for example in response to environmental disturbance or as a result of natural successional processes. However, in recent times human activities such as livestock grazing, fire suppression, predator removal and climate change have resulted in environmental changes so rapid that species and populations often cannot adapt; species may become locally or regionally extinct, non-native species may increase in abundance and distribution, and plant populations may become increasingly susceptible to disease and other disturbances.

Protecting communities undergoing rapid and often unpredictable change is a difficult challenge for natural area managers. By integrating the numerous ecological datasets available, managers at Bandelier National Monument (BAND) have a unique opportunity to document recent landscape-scale changes occurring during the last several decades and to relate those changes to human activities. Understanding the degree to which human activities are altering vegetation communities and in what manner will allow managers to act proactively in anticipation of future impacts.

This project will be coordinated between NPS-BAND, NPS-Southern Colorado Plateau Network (SCPN), and USGS to assess vegetation changes in three community types - pinyon-juniper woodland, ponderosa pine forest, and mixed conifer-aspen forest - that have occurred within the park and surrounding landscapes over the last 20 (approximately) years. The investigator will also determine, when possible, the primary drivers of observed change, including interactions between vegetation and herbivores. In addition, future discussions between the investigator and NPS will determine the reference conditions to which current vegetation communities should be compared, giving particular attention to an approach which provides the most value to NPS managers.

Pinyon-Juniper Woodland

Pinyon-juniper woodland is the most common vegetation type within Bandelier, though the ecology of this community and the distribution of pinyon and juniper species have changed greatly during the last century. Pinyon (*Pinus monophylla, P. edulis*) and juniper (*Juniperus monosperma*) trees have become substantially more dense and the community, though altered, has expanded into areas that were previously grasslands, shrublands or ponderosa pine forests. In addition, the herbaceous understory that historically supported ground fires has been greatly reduced or eliminated. Primary drivers of the increase in pinon-juniper abundance since the late 1800s have been: 1) grazing by domestic livestock, feral burros and deer that selectively impacted herbaceous understory plant species, and 2) human suppression of fires that historically limited woody species recruitment.

The resulting denser and more widely distributed pinyon-juniper forests have most recently been impacted by drought and associated tree mortality due to pine beetle infestations. Between 2000 and 2004, over 90% of the mature pinyon trees in Bandelier were killed, resulting in both the conversion of woodlands to more open savanna-type communities, and a critical increase in woody debris that could support very hot fires in communities not adapted to this level of fire intensity. While short-term (3-5 year) droughts have occurred historically in the region, it is projected that human-mediated climate change will result in consistently higher temperatures and drier conditions for the foreseeable future.

Managers and scientists have developed a set of desired future conditions (DFC) for the pinyon-juniper woodlands that would characterize a naturally functioning and disturbance-resilient community. These conditions include a matrix of open grass and shrublands with some denser woodland patches, and a much more developed understory. Low intensity surface fires would occur at intervals of 5–30 years depending on ground cover and substrate, and crown fires would be uncommon. DFC for pinyon-juniper woodlands in Bandelier are based on inferences about the nature and status of these communities following prehistoric land use activities (ending around ca. 1600) and prior to historical land use patterns (beginning around 1880).

Ponderosa Pine

Ponderosa pine (*Pinus ponderosa*) forests occur at moderate elevations in Bandelier. Primarily due to the suppression of fire during the last century, individual tree density has greatly increased and the extent of ground cover, particularly grasses, has declined in these communities. Historically, relatively frequent surface fires limited tree recruitment and maintained an open understory. Recruitment pulses for trees occurred only during wet periods when seeds were abundant and fires less frequent. In recent times, high tree densities, drought, and large areas of dead trees resulting from pine beetle infestations have allowed fires that did start to become very intense, devastating large areas of forest and threatening human life and property. Current restoration approaches in ponderosa pine include fire management and thinning, with DFC that include more a more open understory with fewer adult trees, increased herbaceous ground cover, and greater ground fire frequency.

Mixed Conifer/Aspen

Mixed conifer forests occur at the higher elevations in Bandelier, and include a diversity of pine and fir species along with dispersed aspen (*Populus tremuloides*) stands. There are two important ecological changes occurring in these forests with which managers are concerned. As in the pinyon-juniper and ponderosa pine forests, fire suppression has led to increased conifer density and reduction of the herbaceous understory. These changes increase the risk of crown fires and reduce the availability of important wildlife habitat historically found in these communities. Also of concern is a widespread reduction in aspen abundance observed across the western US and in Bandelier. Impacts to aspen populations have resulted from reduced fire frequencies, (fire encourages aspen sprouting), browsing by super-abundant populations of elk, and competition from expanding conifer populations. Specifically, exclosure data and observations show the extent to which browsing substantially reduces aspen density and growth, particularly following fires when elk preferentially consume new aspen sprouts.

Objectives

The primary objectives of this project are to:

1) evaluate the direction of landscape-scale vegetation change that has occurred in recent (20-year) time in Bandelier in these three communities;
2) identify the primary drivers of observed change;
3) suggest (to the extent possible) the direction of change that might be observed in each type in the next 30-50 years.

To achieve the objectives the investigator will first use existing information and expert consultation to describe the current condition of the three community types in the park in regards to species composition, general demographics (population age distribution) of the primary tree species, observed (not quantified) assessment of community structure, e.g. relative cover of canopy and understory species, and current fire and disturbance regimes. (Much of this work has been done previously so it is anticipated that this task will comprise a relatively small portion of the overall effort.)

Secondly, the investigator will utilize existing data to describe recent (20 years) changes in the same four properties (species diversity, age distribution, community structure, disturbance) for each community. To do this the investigator will need to assess the utility of available datasets by examining metadata and communicating with the NPS and USGS scientists most familiar with the data. The investigator will then select the data that are most relevant to the overall assessment for each community type. (The responsibility for data management prior to analysis will be determined in future discussions.)

Further, the investigator will need to communicate with NPS and other scientists to determine appropriate reference conditions with which to compare these three communities as they currently exist at Bandelier. Reference communities may be those described in previously described DFC, or similar communities in other locations that have experienced minimal recent impacts, but they should have relevance to managers, i.e. exhibit community characteristics that could be obtainable with reasonable management within projected disturbance and climate change projections.

Finally, it is likely that completion of one or more of the tasks will be hampered by the absence of sufficient data and/or information. The investigator will provide analysis and conclusions using whatever data, information and expert opinions are available, but will include in the final report a discussion of relevant data gaps that could be addressed with further sampling or research.

All work will include close collaboration between USGS and NPS staff. The investigator will have access to unpublished NPS and BAND natural resource data that may be useful, including the draft vegetation map and associated databases, NPS fire effects monitoring data, and inventory and monitoring data sets. Relevant unpublished reports from NPS/BAND and other applicable data sets will be available.

Tasks

1. *Data Transfer and Meeting with Bandelier NM Personnel.* The investigator will travel to Bandelier in the summer of 2011 to meet with Bandelier personnel to discuss priorities and transfer data. These meetings will also be used to develop a reference or desired condition concept.
2. *Data Screening.* The investigator will view and assess the utility of 38 candidate datasets identified by NPS and Bandelier NM Staff. Each will be assessed based on relevance to vegetation change, completeness and readiness for use. Data which sample long time series, and data that are well-replicated in space will be preferentially rated.
3. *Analysis and Synthesis for Pinyon-Juniper, Ponderosa Pine and Mixed Conifer communities.* Using the best of the available datasets (2-5 datasets for each community type), the investigator will statistically analyze to the degree possible the rate and direction of vegetation change in species composition, population age distribution, and community structure, identify key drivers responsible for observed changes, and describe any important changes that have or are anticipated to occur in relevant disturbance regimes. If possible the analysis will also attempt to define alternative ecosystem states.
4. *Analysis and Synthesis for Aspen populations*. Given the concern over browsing impacts on aspen, the investigator will pay particular attention to data sets that describe observed and measured changes in aspen populations in the park. The same conditions of species composition, age distribution, and

community structure will be assessed and compared over time, and the primary drivers of observed change identified. Projected future conditions for aspen may be made separately, or included as part of the mixed conifer community assessment.

5. *Identification of data gaps*. The investigator will identify data gaps and suggest future areas of research that would provide park managers with critical information on vegetation community dynamics.
6. *Reporting*. A final synthesis report will be prepared which integrates information from the literature with the results obtained above. Chapters of the report will be devoted to each of the three target ecosystems. The report will include attention to the requirements of the NPS NRCA report guidelines, including a description of the resources assessed (vegetation communities), the measures used to conduct the assessment, the data and methods applied, the reference conditions to which the state of the resources were compared, the resource conditions and trends (if possible), and an evaluation of the level of confidence of the final assessment. Appendices will summarize the investigator's assessment of the candidate datasets and identify data gaps.

Appendix D: Species of Management Concern at Bandelier National Monument

(See attached file: Appendix D Species of Management Concern at Bandelier National Monument.xlsx)

Appendix E: Analysis of Water Quality Core Parameters and Constituents at Bandelier National Monument

Bandelier National Monument

Natural Resource Condition Assessment

Natural Resource Report NPS/BAND/NRR—2015/1000

Sample type	Bandelier	Comparative Studies – Fire/Flood	Other potential impacts
NPS CORE WATER QUALITY PARAMETERS			
Turbidity	Historic exceedances (Brown 2008) and a primary reason Frijoles remains impaired (State of NM 2004); extreme sediment loads in Frijoles following La Mesa Fire (Veenhuis 2002); low to moderate levels in both streams in mid-2000s (Macy and Monroe 2006) but increased dramatically in both streams following Las Conchas Fire flooding, declining in subsequent months but generally remaining above pre-fire levels (Dyer and Monroe 2013);	Generally exceeds drinking water standards in the first year following a fire (Neary et al. 2005); found variation in mobile sediment related to burn severity in the stream catchment; sediment transport from the Cerro Grande Fire from LANL remained high four years post fire (Gallaher and Koch 2004);	Ongoing upland erosion will likely keep sediment levels in streams above desired conditions for the foreseeable future (NPS 2007);
Water Temperature	State identified high temperatures as cause for impairment for Frijoles (State of NM 2004); increased temps recorded in Frijoles following La Mesa Fire and in Capulin following Dome Fire (Brown 2008); many exceedances following Las Conchas Fire and floods in 2011 (Dyer and Monroe 2013);	Immediate increases in temps due to direct impacts from a heated environment; average annual temps may increase due to absence of vegetation that blocks solar radiation on the water; (Neary et al. 2005); Hall and Lombardozzi (2008) found higher water temperatures in burned areas in first year but no detectable differences in later years;	Climate change impacts that will result in increased environmental temperatures in the Southwest;

Sample type	Bandelier	Comparative Studies – Fire/Flood	Other potential impacts
Dissolved Oxygen	Brown (2008) found numerous conditions of low DO in Bandelier since 1988; Macy and Monroe (2006) found no impairment; Frijoles was occasionally slightly below standards following Las Conchas Fire (Dyer and Monroe 2013);	Higher water temps, both short and long-term, cause decreased levels of DO resulting in increased algal growth (Mott, 1999, Neary et al. 2005); O_2 concentrations lower in burned catchments (Hall and Lombardozzi 2008);	
pH	Several historic instances in Frijoles below state minimums (Brown 2008); sites generally within limits in mid-2000s (Macy and Monroe 2006); increased slightly after Las Conchas Fire but within standards (Dyer and Monroe 2013)	Generally increases after fires but rarely above EPA drinking water standards (Neary et al 2005); Hall and Lombardozzi (2008) found no change;	
Conductivity	Some high levels historically (Brown 2008); increased substantially after Las Conchas Fire, declining in subsequent months but not returning to pre-fire levels (Dyer and Monroe 2013);	Hall & Lombardozzi (2008) found greater but statistically insignificant levels of conductivity in burned compared to unburned streams;	

Sample type	Bandelier	Comparative Studies – Fire/Flood	Other potential impacts
Discharge	Streamflow increased dramatically in Capulin following the Dome Fire and in Frijoles following the La Mesa Fire (Veenhuis 2002); flow in both streams was low in 2001 until the August floods when cfs in Frijoles rose to over twice the previously recorded maximum (>7,000 cfs, Dyer and Monroe 2013);	Post-fire floods and increases in discharge are often the most damaging hydrologic impacts of wildfire; post-fire flows can be orders of magnitude greater than pre-fire flow amounts (Neary et al. 2005); increased runoff amounts on LANL continued up to 3-4 years following the Cerro Grande Fire (Gallaher and Koch 2004);	Models and recent trends suggest that while average precipitation in the Southwest may decline due to climate change summertime monsoon storm intensity may increase (Anderson et al. 2010), with associated impacts on soil stability;
MAJOR IONS			
e.g. calcium, chloride, fluoride, magnesium, potassium, silica, sodium, and sulfate, acid neutralizing capacity [CaCO3], total dissolved solids	In mid-90s chloride levels in Frijoles were moderate and silica was elevated (Levings et al. 1998); elevated levels of several elements in Capulin creek followed the Dome Fire (Brown 2008); Dyer and Monroe (2013) discuss altered water chemistry likely resulting from Las Conchas Fire impacts;	Chloride increases slightly in response to fire but generally remains below drinking water standards, while bicarbonates often increase (Neary et al. 2005); most mineral salts are higher in burned areas (Hall and Lombardozzi 2008);	
NUTRIENTS			

Sample type	Bandelier	Comparative Studies – Fire/Flood	Other potential impacts
Ammonia, nitrite, nitrate, organic nitrogen, phosphorus	Several instances of elevated nutrients, particularly phosphorous, in Frijoles since 1988 (Macy and Monroe 2006, Brown 2008); Macy and Monroe (2013) found no exceedances in either stream in 2011 though orthophosphate and N levels increased after the fire and floods;	Short-term (1-year) increases, particularly in nitrates, often occur due to loss of vegetation (Neary et al. 2005); Hall and Lombardozzi (2008) found higher levels of nitrates and orthophosphates in burned areas;	
TRACE ELEMENTS			
Trace Elements (e.g. aluminum, arsenic, barium, chromium, lead, mercury, selenium, silver, uranium, vanadium, zinc)	In the mid-90s dissolved iron and beryllium were high (Levings et al. 1998); State historically found Frijoles impaired due to DDT, and Capulin impaired due to stream bottom deposits (State of NM 2004); levels of several trace elements increased after the Las Conchas Fire, particularly manganese (Dyer and Monroe 2013), but none exceeded NM standards (some trace metals have no published standards);	Heavy metals presence post-fire comes largely from sediments (Neary et al 2005);	High levels may reflect natural conditions (Levings et al. 1998); possible inputs from past land use (Weeks 2007);
MICROORGANISMS			

Sample type	**Bandelier**	**Comparative Studies – Fire/Flood**	**Other potential impacts**
Microorganisms (fecal coliform, *E. coli*)	Brown reported no data collected since 1996; State reported exceedences of fecal coliform for Frijoles as reason for impairment (State of NM 2004); Dyer et al. (2012) and Dyer and Monroe (2013) found exceedences for *E. coli* at all Frijoles sites in 2010 and 2011;		Livestock and human inputs including visitor activities in and around water sources, grazing and other agricultural activities, wildlife (Mott 1999); current inputs of *E. coli* unknown?
OTHER ELEMENTS			
Radionuclides	After the Las Conchas Fire, SCPN (Dyer and Monroe 2013) conducted two sampling events for gross alpha, beta, and gamma radiation as well as isotopic uranium radionuclides	Storm events following the Cerro Grande Fire transported elevated levels of nucleotides from LANL lands (Gallaher and Koch 2004);	
MACROINVERTEBRATES			
Total Abundance	Total abundance declined to nearly zero in Capulin following post-Dome Fire flooding, recovered by the following spring but declined again after floods in following summer, recovered by year 2 (Vieira et al. 2004); declined to almost zero in both streams following post Las Conchas Fire floods (Stumpf and Monroe 2012);	Measurable impacts to riparian invertebrates are primarily from post-fire peak flood events (Minshall et al. 2003, Neary et al. 2005); Hall and Lombardozzi (2008) found reduced densities and biomass 1-year post-fire but largely recovered by 2 years; Malison and Baxter (2010) reported greater abundance 5 years post-fire in high severity areas;	

Sample type	Bandelier	Comparative Studies – Fire/Flood	Other potential impacts
Functional Feeding Group Abundance	Shredders and grazers declined in Capulin after the Dome Fire floods while other groups increased (Vieira et al. 2004);	Relative abundance of feeding functional groups may change due to loss of riparian vegetation and/or increase in algae (Vieira et al. 2004); changes may be due to substrate stability (Neary et al. 2005); Oliver et al. (2010) found that shredders and scraper declined and collector/gatherer increased (even without floods);	
Species/Taxon Richness	Reduced to approximately two taxa in Capulin following Dome Fire floods and did not recover until year 3 (Vieira et al. 2004);	Trends to more variable in high fire severity areas (Minshall 2003)	
Disturbance Tolerance	Aquatic biota shifts to disturbance-tolerant invertebrates (Vieira et al. 2004);	Sensitive species declined after fire (Oliver et al. 2010);	

Appendix F: Avian Demographic Information for Bandelier National Monument

Avian Demographic Information Needed for Management Decisions

Stephen Fettig
2014

Collecting avian data using point counts

A common monitoring method is the use of point counts to document all birds seen or heard from a given point. This method is popular because it requires no specialized equipment other than binoculars and can be done by anyone with the skills to identify birds by sight and sound. Thus, point counts have the advantage of being relatively easy to conduct and to teach to field crews. Typically point counts are conducted for a standardized period of time, such as 3-, 5-, 8- or 10-minutes depending on the protocol. Such counts produce a list of birds with the number of individuals detected. In an attempt to get more information out of point counts, the distance to each bird is estimated or measured with a range finder. Such distance-estimation protocols can be used to calculate density estimates for the more common species within a project area.

Point counts with distance estimation yield present/absence information, and for the most commonly detected species, yield a density estimate. Estimating distances to an unseen singing bird is filled with challenges because not all birds are equally detectable; some species sing or remain silent for different periods of time while other birds just vocalize more quietly than others.

Factors affecting the likelihood of detecting birds

Some birds are more detectable than others because of their vocal behaviors. Failing to account for the birds not detected will influence the density estimates from point counts. During the breeding season, counts of singing birds generally underestimate the true bird population, occasionally by an order of magnitude based on work by Rappole et al. (1993). Rappole and his colleagues also found that song counts poorly predict understory species composition at most of their study sites. This problem of detecting birds was experimentally demonstrated by Alldredge et al. (2007a) who simulated 40 point counts consisting of 10 birds per count for five primary species over distances (34–143 m) using songs played at low and high singing rates. The results indicate that species, singing rate, distance, and observer were all significant factors affecting detection of birds. Alldredge and his colleagues also found significant statistical interactions of singing rate with species and singing rate with distance which can introduce biases into the density estimates. Simulations of expected bird counts indicated that observers detected between 19% (for the worst observer, lowest singing rate, and least detectable species) and 65% (for the best observer, highest singing rate, and most detectable species) of the true population.

Just as the likelihood of detecting a bird influences abundance estimates from point counts, the difficulty in estimating the distance to an unseen bird can affect density estimates. Alldredge et al. (2007b) found distance estimation to have substantial errors, although proper training of observers reduced errors and bias in distance estimates by approximately 15%. Errors in distance estimation showed a nonlinear relationship to distance. Specifically, observers had problems estimating distances beyond 65 m. Observers generally doubled their distance estimates for songs played away from them compared with distance estimates for songs played directly toward them. These findings suggest that the uncertainties in distance estimates to an unseen vocal bird are substantially higher than assumed by some workers. Alldredge et al (2007b) suggest that sound-based point count estimates of avian abundance based on distance estimation need careful scrutiny.

The work of Alldredge et al. (2007a, 2007b) and Simons et al. (2007) used the same experimental set-up, a laptop-computer-controlled system of speakers within a mature bottomland and mixed pine–hardwood rural forest setting in North Carolina. Simons et al. (2007) showed that, on average, observers tend to undercount birds on unlimited-radius counts, though the proportion of birds counted by individual observers ranged from 81% to 132% of the actual total. In contrast to the unlimited-radius counts, when data were truncated at a 50-m radius around the point, observers overestimated the total population by 17% to 122%. This work also illustrated how detection distances decline and identification errors increase with increasing levels of ambient noise. Specifically, the proportion of birds heard by observers decreased by 28 ± 4.7% under breezy conditions, 41 ± 5.2% with the presence of additional background birds, and 42 ± 3.4% with the addition of 10 dB of white noise.

The problem of detecting birds when there is background noise or wind shown by Alldredge, Simons and their colleagues, has been demonstrated in other work. Working in both mixed pine-hardwood and deciduous habitats under both leaf-on and leaf-off conditions, Pacifici et al. (2008) found background noise and the presence of leaves on trees decreased bird detection probabilities. Their work indicated that observers detect between 3% (for the worst observer, least detectable species, with leaves on the trees and added background noise in the deciduous forest) and 99% (for the best observer, most detectable species, with no leaves on the trees and no added background noise in the mixed forest) of the total bird songs.

Thus, addressing bird detection probabilities and distance estimation from point count observations can be fraught with difficulties. There are ways to mathematically address the problems of detecting birds across observers and listening conditions. Estimating the distance to unseen singing birds, however, can be difficult to address even with highly skilled observers. These issues can yield potentially large uncertainties in density estimates. A more important question, however, is whether or not monitoring to estimate bird numbers or density will provide the information we need to be good stewards of bird populations.

Demographic monitoring

To adequately evaluate the health of bird populations, monitoring efforts need to focus on demographic information (Noon and Sauer 1992 as cited by DeSante et al. 2005). Sometime this information is also call vital rates. This kind of monitoring measures life-history rates of a population, such as the number of young produced per adult each year, the probability of a young bird surviving to its first breeding season, the probability that a surviving young bird will breed, the number of adults that survive each year, the probability that a surviving adult will return to the same site to breed the next year, and the probability that a surviving adult from a different site will immigrate to the study area. There are several reasons that these rates provide the information needed for management of bird populations.

First, while environmental conditions and management actions act directly on vital rates, they act only indirectly on population size or density (DeSante and Rosenberg 1998 as cited by DeSante et al. 2005), i.e., changes in vital rates affect population size or density. The indirect relationship between environment conditions and population size decreases our ability interpret causation unless vital rates are examined. The indirect relationship between environmental conditions and population size can best be seen in the lag times. Population change often has a relatively long lag time after a vital rate is affected, while environment change often affects a vital rate immediately or with small lag time (Temple and Wiens 1989). Second, monitoring of vital rates allows a better understanding of climate effects on reproduction than with monitoring populations. This is because reproduction can show relatively high variability as a function of annual weather patterns (DeSante and O'Grady 2000). Third, vital rates provide a clear measure of habitat quality without confounding effects such as population sources and sinks (Van Horne 1983) or misleading habitat quality information based on relative abundance or population size (Pulliam 1988). Fourth, monitoring demographic parameters or vital rates provides key information on what is controlling bird populations. Vital rates provide information on when and where in a bird's life cycle there may be problems. This when-and-where information is critical to formulating management actions. Thus without monitoring of vital rates we cannot achieve a clear understanding of the relationship between environmental conditions and population trends.

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Appendix G: Ecological Impacts from Grazing by Livestock in Bandelier National Monument

Ecological impacts from grazing by livestock

Stephen Fettig
2014

There are three primary ways grazing impacts ecosystems: composition, function, and structure (Franklin et al. 1981). The ecological costs of livestock grazing can be summarized as follows: (1) Alternation of species composition of communities, including decrease in density and biomass; (2) Disruption of ecosystem functioning, including interference in nutrient cycling and ecological succession; and (3) Alteration of ecosystem structure, including changing vegetation stratification, contributing to soil erosion, and decreasing availability of water to biotic communities.

Alternation of Species Composition

Grazing affects the species composition of plant communities in two ways: (1) active selection by herbivores for or against a specific plant taxon, and (2) differential vulnerability of plant taxa to grazing (Szaro 1989). Decreases in the density of native plant species and diversity of native plant communities as a result of livestock grazing activity have been observed in a wide variety of western ecosystems. In Arizona, perennial grasses and a palatable shrub showed dramatic density decreases with grazing (Blydenstein et al. 1957). In California, Web and Stielstra (1979) documented a 60% reduction in above-ground biomass of annuals and a 16-29% decrease in cover of perennial shrubs with grazing. In Idaho, a grazed site had 33% of the species richness of an ungrazed site (Reynolds and Trost 1980). In New Mexico, grass density increased by 110% after 30 years of protection form grazing (Gardner 1950). In Arizona, species richness increased, as did canopy cover for midgrass, shortgrass, and shrub, and forb groups after removal of livestock (Brady et al. 1989). Another study in Arizona showed that woody plants were significantly more abundant after the removal of livestock (Bock et al. 1984). In Washington State, Rummel (1951) found decreased species richness on grazed sites within ponderosa pine habitat. In Utah, Cottam and Evans (1945) found the absence or near absence of 10 grass species on grazed sites. In Oregon, species richness increased from 17 to 45 species nine years after removal of livestock (Winegar 1977). Again in Arizona, Szaro and Pase (1983) found the herbaceous cover of a grazed plot was less than half that of an ungrazed plot. In Colorado, shrub canopy coverage increased 5.5 times and willow canopy coverage increased 8 times with the removal of livestock (Schulz and Leininger 1990). Grazing, furthermore, can completely transform communities as is the case in most of southern New Mexico. Specifically, grazing is credited with being the leading cause for transforming most southern New Mexico's grasslands to creosote-bush desert (Whitefield and Anderson 1938, York and Dick-Peddie 1969). More directly applicable to the VCNP is the fact that Kennedy (1997) noted that grazing has thoroughly changed most southwestern riparian zones. He called these changes, "complete type conversions." An example is the fact that grazing can eliminate a willow stand in 30 years (Kovalchik and Elmore 1992).

Grazing also can exert a great impact on animal population via habitat structure and prey availability (Wagner 1978, Jones 1981, Mosconi and Hutto 1982, Szaro et al 1985, Quinn and Walgenback 1990). Reynolds and Trost (1980) showed that the density and diversity of small mammals was reduced on grazed sites in Idaho. In Nevada, small mammal density was over one-third lower and nearly 50% on grazed sites compared to ungrazed areas (Medin and Clary 1989). In Utah, use by songbirds, raptors, and small mammals increased 350% after 8 years of rest from grazing (Duff 1979). In Colorado, Crouch (1982) showed that ducks and terrestrial nongame birds were more abundant in ungrazed habitats. In

North Dakota, upland sandpiper nest density was reduced on grazed sited compared to ungrazed areas (Bowen and Kruse 1993). In Montana, grazing altered the species composition and density of riparian bird communities (Mosconi and Hutto 1982). In Oregon, willow flycatcher abundance increased from 0 to 30 when grazing intensity was reduced by 4 times (Taylor and Littlefield 1986). In California, Busack and Bury (1974) found that lizard abundance was 2 times higher and biomass was 3.7 times higher on an ungrazed site. In Arizona, Jones (1981, 1988) showed that lizard abundance and diversity was higher in ungrazed sites for 4 of 5 habitat types. In New Mexico, Szaro et al (1985) showed that a garter snake was 5 time more abundant in ungrazed sites than in grazed areas. In Colorado, the standing crop of trout was doubled after removal of livestock (Stuber 1985).

Grazing also destabilizes plant communities by aiding the spread and establishment of exotic species, such as cheat grass (Bromus tectorum). Livestock help spread exotic plant species by (1) dispersing seeds in fur and dung, (2) opening up habitat for weedy species such as cheatgrass (Gould 1951, Mack 1981), and (3) reducing competition from native species by eating them. It's important to note that alien grass invasions in North America have been most severe in arid or semiarid parts of the west when associated with grazing (D'Antonio and Vitousek 1992).

Disruption of Ecosystem Functioning

Livestock grazing also can disrupt ecological succession. The cumulative impacts of long-term livestock use has produced and maintained early seral vegetation throughout much of the west (Longhurst et al. 1982). Part of the issue is prevention of seedling establishment. Grazing and trampling by livestock, furthermore, have transformed a variety of southwest riparian systems into even-aged non-reproducing communities (Carothers 1977, Szaro 1989). In Oregon, grazing has retarded success in willow-cottonwood-alder communities (Kauffman et al. 1983). These grazing issues prompted Davis (1977) to conclude that livestock grazing was "probably the major factor contributing to the failure of riparian communities to propagate themselves.

Alteration of Ecosystem Structure

The physical structure of ecosystems is often changed by livestock grazing. In Washington, Rummel (1951) documented that grazing was responsible for changing the physical structure of a ponderosa pine forest from an open, park-like stand with dense grass to a community characterized by dense pine reproduction and lack of grasses. In northern Arizona, Cooper (1960) showed that grazing was at least partially responsible for similar structural changes in ponderosa pine forests to those found by Rummel (1951) in Washington. In a high-altitude willow riparian community in Colorado, grazing influenced the spacing of plants and the width of the riparian zone (Knopf and Cannon 1982). Throughout the intermountain West, extensive willow stands once occurred within rangelands which are now almost completely without willows (Kovalchik and Elmore 1992).

Researchers have long recognized that grazing contributes to the deterioration of soil stability and porosity, and increases erosion and soil compaction. In Arizona, Aldo Leopold (1924) wrote that "grazing is the prime factor in destroying watershed values". Grazing reduces the roughness of watersheds, resulting in more surface runoff and more soil erosion (Ohmart and Anderson 1982). In the upper Rio Grande, Cooperrider and Hendricks (1937) showed that grazing changed plant cover and increased flash flooding and erosion. Several authors have shown that grazing increases soil compaction and decreases water infiltration (Alderfer and Robinson 1949, Orr 1960, Rauzi and Hanson 1966, Bryant et al. 1972, Rauzi and Smith 1973, Kauffman and Krueger 1984, Abdel-Magid et al. 1987, Orodho et al. 1990). In the southwest U.S., several authors suggest that livestock grazing has also contributed to degraded stream-channel structure (Bryan 1925, Leopold 1951, Hereford and Webb 1992, Hastings 1959).

Many authors have noted that livestock spend a disproportionate amount of time in riparian zones (Ames 1977, Kennedy 1977, Thomas et al. 1979, Roath and Krueger 1982, Van Vuren 1982, Gillen et al. 1984). This time in riparian zones is a cause of concern because these habitats are among the biologically richest in the West (Johnson et al. 1977). The degradation of Western riparian habitats began with high grazing levels in the late Nineteenth Century (Chaney et al. 1990). Based on an extensive survey of southwest riparian communities, Szaro (1989) concluded that “livestock may be the major cause of excessive habitat disturbance in most western riparian communities.” This view was supported by an interagency wildlife committee (OWIWC 1979) which concluded that grazing was the most important factor in degrading wildlife and fisheries habitat throughout the 11 western states. Likewise, ecologist in Montana suggested that livestock grazing is the major cause of habitat disturbance in most western riparian communities (Mosconi and Hutto 1982).

Platts (1981) summarized a key stream-bank issue by writing, “Grazing can affect the streamside environment by changing, reducing, or eliminating vegetation bordering the stream. Channel morphology can be changed by accrual of sediment, alteration of channel substrate, disruption of the relation of pools to riffles, and widening of the channel. The water column can be altered by increasing water temperature, nutrients, suspended sediments, bacterial populations, and in the timing and volume of stream flow. Livestock can trample stream banks, causing banks to slough off, creating false setback banks, and exposing banks to accelerated soil erosion.” Livestock grazing can further alter riparian vegetation by removing terminal buds and stimulating lateral branching (Kauffman and Krueger 1984, Szaro 1989).

Livestock grazing can be especially deleterious to fish. Removal of vegetation cover and the trampling of over-handing stream banks alter fish habitats and water temperature (Behnke and Zarn 1976, Van Velson 1979), as well as decrease water quality of streams (Diesch 1970, Buckhouse and Gifford 1976). These changes can completely change aquatic ecosystems (Kennedy 1977, Kauffman and Krueger 1984). Insights such as these led the American Fisheries Society to issue a formal position statement call for improved riparian habitat management (Armour et al. 1991).

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