Chapter 6
Pinyon-Juniper Woodlands

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INTRODUCTION

Pinyon-juniper woodlands are one of the largest ecosystems in the Southwest and in the Middle Rio Grande Basin (Fig. 1). The woodlands have been important to the region's inhabitants since prehistoric times for a variety of natural resources and amenities. The ecosystems have not been static; their distributions, stand characteristics, and site conditions have been altered by changes in climatic patterns and human use and, often, abuse. Management of these lands since European settlement has varied from light exploitation and benign neglect, to attempts to remove the trees in favor of forage for livestock, and then to a realization that these lands contain useful resources and should be managed accordingly.

Land management agencies are committed to ecosystem management. While there are several definitions of ecosystem management, the goal is to use ecological approaches to create and maintain diverse, productive, and healthy ecosystems (Kaufmann et al. 1994). Ecosystem management recognizes that people are an integral part of the system and that their needs must be considered. Ecological approaches are central to the concept, but our understanding of basic woodland ecology is incomplete, and there are different opinions and interpretations of existing information (Gottfried and Severson 1993). There are many questions concerning proper ecosystem management of the pinyon-juniper woodlands and how managers can achieve these goals (Gottfried and Severson 1993). While the broad concept of ecosystem management generally is accepted, the USDA Forest Service, other public land management agencies, American Indian tribes, and private landowners may have differing definitions of what constitutes desired conditions.

Key questions about the pinyon-juniper ecosystems remain unanswered. Some concern the basic dynamics of biological and physical components of the pinyon-juniper ecosystems. Others concern the distribution of woodlands prior to European settlement and changes since the introduction of livestock and fire control. This relates to whether tree densities have been increasing or whether trees are invading grasslands and, to a lesser extent, drier ponderosa pine (*Pinus ponderosa*) forests. In areas where woodlands were heavily used by American Indians for fuelwood prior to European contact, the advance of pinyon and juniper could represent the slow recovery from intensive use (Samuels and Betancourt 1982). There are numerous questions regarding declines in watershed condition related to changes in pinyon-juniper tree stand densities and to the density and composition of understory vegetation.

There are different opinions about proper management of woodland ecosystems. Should these lands be managed for a single resource, such as forage for livestock production, or managed for sustained production of multiple resource products and amenities? Depending on site and stand conditions, the woodlands can produce variable quantities of fuelwood, pinyon nuts, wildlife habitat, forage for livestock, and cover for watershed protection. Management must

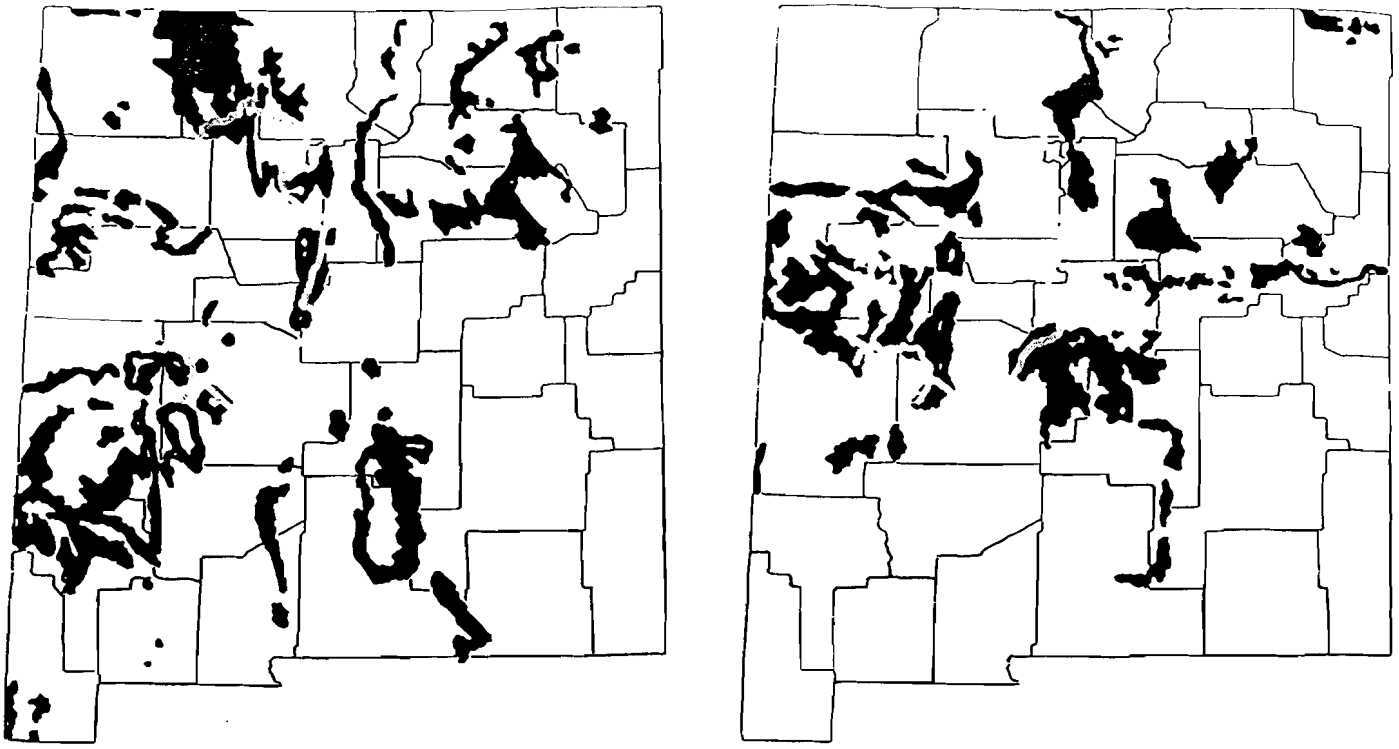

Figure 1.—The distribution of pinyon-juniper woodlands (left) and juniper savannas (right) in New Mexico based on Dick-Peddie (1992). The Middle Rio Grande River Basin is outlined in gray.

also consider increasing recreational demands, threatened and endangered species, and protection of archeological sites. Many pinyon-juniper woodland watersheds in New Mexico have unsatisfactory soil and watershed conditions (USDA Forest Service 1993); managers must develop restoration procedures that recognize the value of woodland ecosystems.

The concerns, questions, and conflicts surrounding management of pinyon-juniper lands, as well as the ecological foundations of ecosystem management, require that all interested parties reevaluate attitudes toward the woodlands. Ecosystem management goals and concepts recognize diversity. Pinyon-juniper woodlands are diverse, and stand characteristics and site productivities vary. Management objectives and prescriptions must evaluate the potential of each site, and decisions must be based on sound scientific information. This information is often unavailable. Therefore, this paper describes what we do know about the characteristics, distribution, and ecology of pinyon-juniper woodlands, including the effects of natural and human factors, within the southwestern United States and particularly the Middle Rio Grande Basin. It also reviews some past and present management options in this widespread and important vegetation type. The review draws on research and management information from the Rio Grande Basin and from similar areas in the Southwest and adjacent regions. It does not attempt to review all of the relevant literature; additional sources can be found within the articles cited in the References.

CHARACTERISTICS

What are pinyon-juniper woodlands? Woodlands are generally characterized by trees that are of small stature but often more than 5 m in height; have relatively open canopies; and have greater than 40 percent crown closure (USDA Forest Service 1993). In the Southwest, relatively open stands where tree crown cover is between 10 and 40 percent are also classified as woodlands. The pinyon-juniper woodlands are variable in species composition, density, and physiographic site characteristics. Some sites contain dense stands of relatively large trees and meet many of the criteria of old growth (USDA Forest Service 1990), while other sites contain open stands of mainly younger trees that appear to be of a more recent origin. Pinyon-juniper and pure juniper woodlands are generally considered together in this review.

The two-needle Colorado pinyon (*Pinus edulis*) is the common species in most pinyon-juniper stands in the Southwest and eastern Utah and Colorado. A one-needle pinyon, *P. californiarum* var. *fallax*, hybridizes with *P. edulis* in low elevations of central Arizona and southwestern New Mexico. Pinyons with a mix of one and two-needle fascicles are found in the low elevations of the Middle Rio Grande Basin, and probably represent Holocene long-distance seed or pollen dispersal from *P. californiarum* var. *fallax* stands in southwestern New Mexico. Pinyons are typically between 3 and 11 m tall and 13 to 46 cm in diameter, although larger individuals can be found on moister sites. The pinyons are slow growing but relatively long-lived trees (Ronco 1990). Trees 300 to 400 years old are common in old-growth stands in both Arizona and New Mexico, but trees over 500 years in age are rare (Swetnam and Brown 1992).

Juniper (*Juniperus* spp.) is the other major tree genus occurring in these woodlands. Junipers are generally small, multi-stemmed trees less than 12 m tall. There are four major juniper species in the Southwest: one-seed juniper (*J. monosperma*); Utah juniper (*J. osteosperma*); alligator juniper (*J. deppeana*); and Rocky Mountain juniper (*J. scopulorum*). Stands can contain one of the juniper species or a combination of species. Junipers can attain great ages, but unfortunately, it is difficult to determine precise ages using tree rings because of the prevalence of false and missing rings, particularly in alligator and one-seed junipers. Rocky Mountain juniper is an exception, and several trees over 1,000 years in age recently have been accurately dated in west-central New Mexico (Grissino-Mayer et al., in press). Less than 50 percent of the Utah juniper stems can be dendrochronologically dated; however, on relatively wetter sites, there is a better chance of cross-dating ring-width patterns within and across populations. Dendrochronology has allowed precise dating of archeological sites that incorporate ancient timber (Bannister and Robinson 1975); dendroclimatic reconstructions spanning a thousand years or more (D'Arrigo and Jacoby 1991; Grissino-Mayer et al., in press); and reconstruction of tree demographies at interannual resolution (Betancourt et al. 1993).

Understory biomass within southwestern pinyon-juniper stands is generally small. However, because of the broad distribution of this vegetation type, the total number of plant species associated with the woodlands is great (Medina 1987; Ronco 1990). Some important representative herbaceous species include blue grama (*Bouteloua gracilis*), sideoats grama (*B. curtipendula*), sand bluestem (*Andropogon hallii*), Arizona fescue (*Festuca arizonica*), and goosefoot (*Chenopodium graveolens*). Representative trees and shrubs include gray oak (*Quercus grisea*), true mountain-mahogany (*Cercocarpus montanus*), sagebrush (*Artemisia* spp.), and Mexican cliffrose (*Cowania mexicana*). More details on understory vegetation are found in Medina (1987), Ronco (1990), and USDA Forest Service (1987).

ECOLOGY

Distribution

Approximately 19 million ha of pinyon-juniper woodland occur in the United States; it is an important vegetation type in seven of the western states (Evans 1988). Pinyon-juniper woodlands constitute the most common vegetation type in Arizona and New Mexico. The literature contains several estimates of the area occupied by woodlands; many of the differences may be attributed to the way marginal juniper savanna lands are defined. West et al. (1975) indicated that conifer woodlands cover approximately 26 percent or about 8.2 million ha of New Mexico, but this value probably included grasslands that have a tree component. A recent survey of New Mexico's forest resources (Van Hooser et al. 1993) reports that approximately 3.4 million ha of pinyon-juniper and pure juniper woodlands have the potential for producing wood products. Fowler et al. (1985) indicated a relatively similar area of about 4 million ha or 14 percent of New Mexico that contain stands that could be considered manageable for tree products because of their site and stand characteristics.

Pinyon-juniper woodlands are an important type within the 64,150 km^2 Middle Rio Grande Valley (Fig. 1), which includes parts of ten counties (Crawford et al. 1993). A major part of the Middle Rio Grande Basin is in Bernalillo, Cibola, McKinley, Sandoval, Socorro, and Valencia counties, and a minor part occurs in Catron, Torrance, Rio Arriba, and Santa Fe counties. Although statistics for the area of pinyon-juniper woodlands within the Basin are not readily available, calculations based on data reported by Van Hooser et al. (1993) indicate that the six counties contain about 1 million ha of woodland or 18 percent of the total county areas. The proportion of area occupied by woodlands ranges from 27 percent for Cibola to 5 percent for Valencia County. Woodlands occur on private, USDA Forest Service, USDI

Bureau of Land Management, American Indian, and State of New Mexico lands.

Available soil moisture and season of precipitation are the most critical factors controlling woodland distribution, composition, density, and stand condition in undisturbed sites. The high variability of woodland habitat types is associated with the variability of climatic and site conditions. Moir and Carleton (1987) recognized at least 70 habitat types and 280 ecological site types in Arizona and New Mexico. Woodlands occur on relatively moist sites that support dense stands of relatively tall trees and on dry sites where trees are scattered and of low stature. Junipers are more drought tolerant than pinyon and tend to dominate drier sites. The proportion of pinyon increases with increased elevation and available moisture. In general, pinyon is dominant above 2,200 m. Competition with ponderosa pine for moisture, or fire regimes associated with ponderosa pine, may determine the upper limit as suggested by the fact that pinyon achieves its greatest abundance or density near its upper limit. Generally species density would taper off near its upper and lower environmental limits.

Topographic and edaphic influences are apparent in southern New Mexico where pure one-seed juniper stands have higher densities on northeastern exposures than on drier southwestern slopes (Pieper and Lymbery 1987). Distribution of Utah juniper in Arizona and northern New Mexico is related to the predominance of winter moisture relative to summer moisture (Springfield 1976). One-seed juniper is most common where winters are cool and dry and where summer precipitation is more important. Alligator juniper is also identified with summer moisture regimes. Rocky Mountain juniper is considered the least drought-tolerant of the common juniper species and generally occurs above 2,100 m in the Middle Rio Grande Basin.

Climate

Southwestern pinyon-juniper woodlands occupy the warmest tree-dominated zone in the region. The climate is usually classified as arid, semiarid, or occasionally, dry subhumid (Ronco 1990). The woodlands grade into juniper savannas, grasslands, oak woodlands, and brush-dominated vegetation zones on drier sites and into ponderosa pine forests at higher, moister elevations. Average annual precipitation ranges from 305 to 560 mm and is influenced by geography, elevation, and topography. Temperature ranges are also variable and may control the upper elevational distribution of the type (Evans 1988).

The seasonal distribution of precipitation in the Middle Rio Grande Basin, like most of the Southwest, varies depending on sea surface temperature regimes in the eastern Pacific Ocean and the seasonal position of the polar and subtropical jet streams, the Pacific anticyclone, and the Bermuda High. The climatic regime is characterized by highly variable frontal precipitation in winter, an arid pre-summer, and summer rains that are predictable in timing and amount at a given station but highly variable from site to site. The importance of monsoonal rains diminishes to the northwest, and the April to June period becomes wetter to the northeast. Summer precipitation is greatest on the east flank of the southern Rocky Mountains, i.e., the Sangre de Cristo, Sandia, Manzano, and Sacramento Mountains.

Interannual and interdecadal variability in cool season precipitation apparently is driven by the sea surface temperatures (SST) and sea surface pressure anomalies in the tropical Pacific and the latitudinal position and sinuosity (meridionality) of the polar jet stream, reflecting expansion and contraction of the circumpolar vortex. Two climatic indices that define these conditions are the Southern Oscillation Index (SOI) and Pacific North American Index (PNA). A negative SOI reflects El Niño (warm conditions) in the tropical Pacific; a positive SOI reflects La Nina (cold conditions). A positive PNA value reflects an intensified Aleutian Low, causing winter storm tracks to shift southward; a negative PNA value indicates more northerly storm tracks. In the Middle Rio Grande Basin, wet winters and springs are associated with positive PNA and La Niña conditions.

Decadal-scale variability in rainfall records from the Line Islands (tropical Pacific islands near the Date Line) indicate precipitation surges during El Niño events. In the 20th century, there has been a general (decadal) association between periods of frequent El Niños and an expanded circumpolar vortex (positive PNA) prior to 1930 and after 1960. The period between 1930 and 1960 was characterized by few El Niño events and a contracted circumpolar vortex (negative PNA). There has been an almost permanent shift to El Niño-like conditions that began in 1976 and continued to 1995 (with the exception of the 1988–89 La Niña) (Fig. 2). Teleconnections (cor-

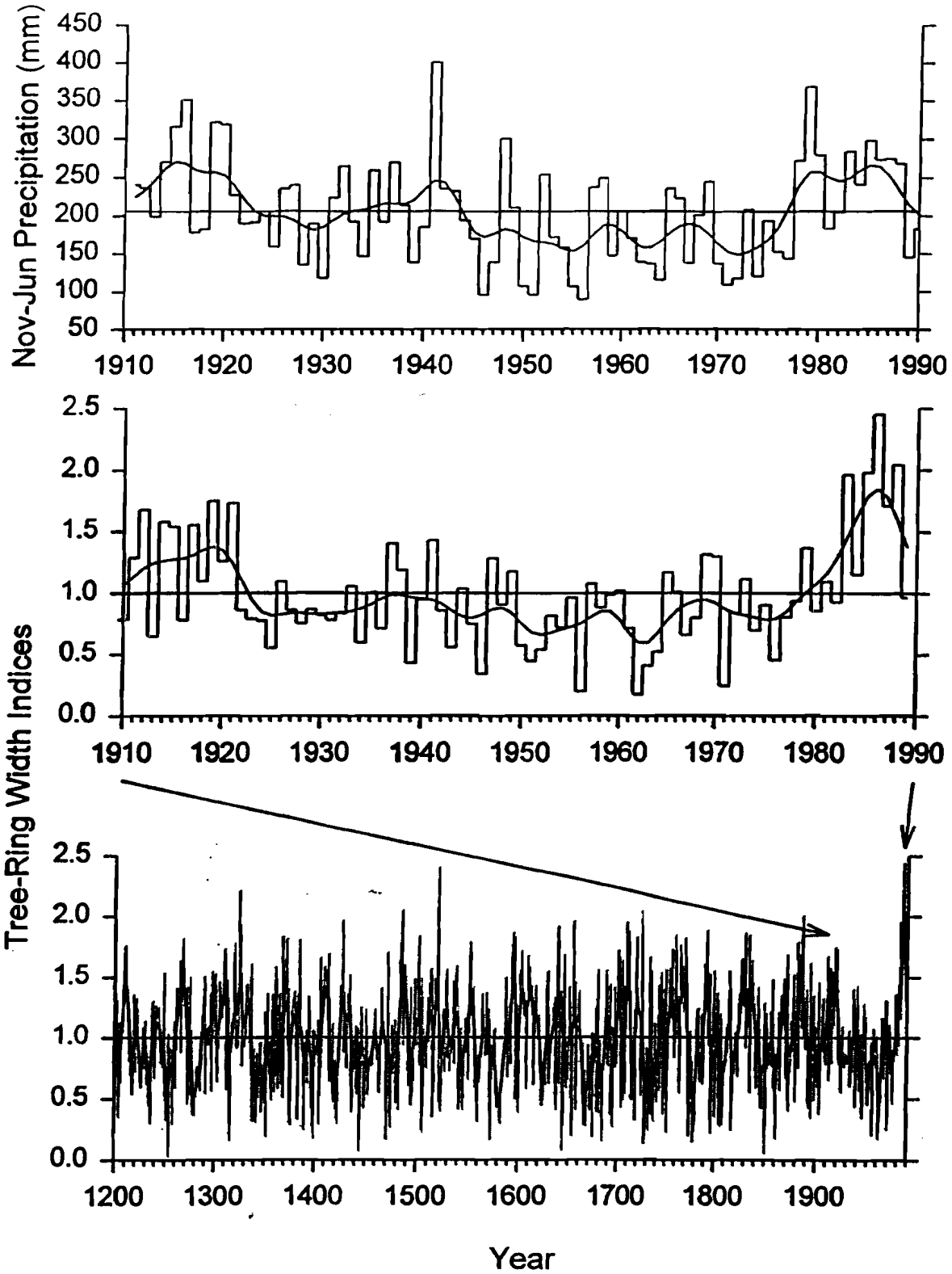

Figure 2.—Winter to spring precipitation variations in northern New Mexico since AD 1200. The upper plot shows total November through June precipitation, 1910-1990, averaged across three recording stations at Chama, Jemez Springs, and Socorro. The middle plot shows annual ring-width variations during the same period from a limber pine (*Pinus flexilis*) tree-ring site near Red River. The Pearson correlation between the precipitation and tree-ring time series is 0.61 ($p < 0.001$). The bottom plot shows the past 700 years of tree-ring variation at this site, suggesting that the late 20th century climatic variation has been very unusual, perhaps unprecedented during the past 8 centuries.

relations between the climate at two distant locations) have shown relationships between the tropical Pacific indices such as SOI, tropical SSTs, Line Island rainfall and southwestern precipitation (Douglas and Engelhardt 1984; Cayan and Webb 1992), streamflow (Molles and Dahm 1990; Cayan and Webb 1992; Webb and Betancourt 1992), and even area burned by wildfires (Swetnam and Betancourt 1990).

Soils and Topography

The woodlands generally occur at elevations of 1,370 to 2,290 m and on all topographic positions. Old-growth stands occur on a variety of physiographic sites. In New Mexico, old stands are often associated with rocky hillslopes where the sparse understory will not carry fire (Swetnam and Brown 1992; Wood and Javed 1992). Woodlands occur on soils that have developed from a variety of parent materials and belong to one of six soil orders (Aridisols, Alfisols, Entisols, Mollisols, Vertisols, and Inceptisols) (Evans 1988). Soils are generally classified as being shallow and well-drained. Woodlands are associated with soils having low fertility (Pieper 1977; Evans 1988), but recent data (USDA Forest Service 1993) indicate that they also can occur on relatively productive soils.

Seed Production

Pinyon is generally monoecious, although dioecious trees are found when the trees are under insect attack or growing under stressed conditions (Gottfried 1987). Seed crops occur every 4 to 7 years depending upon the weather, site conditions, and insect herbivory. Crops are more frequent on the better sites. Trees can start bearing cones at 25 years, but production peaks when trees are between 75 and 100 years old. Cones require three growing seasons to mature (Little 1938) and will contain about 20 relatively large, wingless seeds. A productive tree can produce about 9 kg of seed, and a hectare can yield about 336 kg of seed (Ronco 1990). Mature seed release starts in mid-September and can continue for a 50-day period.

Reproductive success of pinyon is largely a measure of the abundance of episodic bumper seed crops, which are probably linked to favorable climatic events initiating cone primordia and enhancing cone and seed development over a period of several years. Pinyon, during mast years, typically produces large crops simultaneously across large geographic areas. In the 1940s, seed crops in Arizona and New Mexico were monitored annually (Little 1940). The specific influence of climatic variability on flowering, fruiting, and seed germination over a three-year reproductive cycle remains unexplored. Equally unknown is the effect of mast years on long-term fluctuations in stand structure. The climatic trigger must inhibit vegetative growth and induce formation of ovulate cone primordia in late summer, must recur irregularly once every few years, and must be synoptic in scale. By definition, the trigger must be embedded in interannual climatic variability over the region, and ultimately, in the global-scale climatologies that affect the Southwest.

Some of the southwestern junipers are monoecious, such as Utah and alligator junipers, and some are predominately dioecious, such as one-seed and Rocky Mountain junipers (Johnsen and Alexander 1974). Seed-bearing age varies by species and also can be affected by moisture conditions and competition (Gottfried 1989; McPherson and Wright 1987). One-seed and Rocky Mountain junipers begin bearing seed at between 10 and 20 years, while Utah juniper begins at about 33 years (Johnsen and Alexander 1974). Juniper berries, which can contain one to four seeds depending on the species, can be dry and leathery, as in Utah juniper, or thin and resinous, as in one-seed juniper. The main southwestern junipers flower in the spring with berries ripening in the fall; however, some species require two years for the berries to mature. Mature berries can stay on the tree for several years before dispersal.

Seed Dispersal

Wind and gravity are not important dispersal agents for the heavy seeds or berries. Most pinyon seed will fall directly below the canopy and very few seeds will land in adjacent interspace areas (Gottfried 1992a). Birds are considered to be the most important dispersal agents. Balda (1987) found that four species of corvid birds—scrub jay (*Aphelocoma coerulescens*), pinyon jay (*Gymnorhinus cyanocephalus*), Clark's nutcracker (*Nucifraga columbiana*), and Steller's jay (*Cyanocitta stelleri*)—are responsible for caching hundreds of thousands of pinyon seeds during years with large cone crops. Some of these species have special anatomical adaptations which allow them to transport large quantities of seed over considerable distances. Scrub jays are the only one

of the four species that spends most of its time in the woodlands and contributes more to pinyon regeneration in the woodlands than Steller's jay and Clark's nutcracker, which spend much of their time in higher elevation forests.

Hall and Balda (1988) studied caching behavior of scrub jays, and found that they prefer to cache seeds in the soil under pinyon trees even when other locations were available. The birds, as well as mice, will recover 92 percent of the caches by the following spring. The surviving caches are the source of future pinyon regeneration.

Birds also are important for juniper dispersal. Townsend's solitaires (*Myadestes townsendi*) are important dispersal agents for one-seed juniper (Salomonson 1978) and Bohemian waxwings (*Bombycilla garrula*) are important for Rocky Mountain juniper (Noble 1990). In addition to birds, other animals such as coyotes (*Canis latrans*), mice, rabbits, and livestock are considered major dispersal agents for junipers (Johnsen 1962; Noble 1990).

Germination, Establishment, and Early Growth

The environmental and physiological requirements for successful pinyon germination have not been evaluated fully. Preliminary data from an ongoing study have shown that pinyon seeds will germinate in the spring, but if conditions are not satisfactory, they may not germinate until the summer monsoon rain season (Gottfried [unpublished data]). Pinyon germination is between 83 and 96 percent (Gottfried and Heidmann 1986; Ronco 1990). Most juniper seeds will also germinate in the spring; however, germination can be delayed for up to two years because of embryo dormancy, the impermeable seed coat, or chemical inhibitors (Gottfried 1989; Johnsen and Alexander 1974). In general, successful germination is low, ranging from 8 to 49 percent in Utah juniper and from 20 to 75 percent in one-seed juniper.

Although pinyon and juniper are considered shade-intolerant, most new seedlings are often found in the shade of mature trees, shrubs, and slash; overstory foliage generally is not dense enough to reduce light intensities below tolerance levels for survival (Meeuwig and Bassett 1983). Seeds do germinate in the open, but establishment and survival are less certain. Shade moderates the microclimate and, therefore, enhances survival. Seedlings growing under shrubs, which eventually can be overtopped, have the best chance of survival (Meeuwig and Bassett 1983); seedlings growing under mature trees must be released in order to accelerate growth. Seedlings that survive the first year in the open are larger than those that are growing in shaded micro-sites (Harrington 1987).

Although several climatic factors influence germination and establishment, moisture is probably the most critical factor. Meagher (1943) found that supplemental watering did not influence percent germination, but it did improve the speed of germination and survival of pinyon seedlings over a 2-year period. Rapid germination would favor a seedling by allowing it to become established prior to the summer or fall drought periods (Gottfried 1987). Meagher (1943) also determined that shade and/or watering had a similar effect on Utah and one-seed juniper germination and initial survival. Johnsen (1962) reported that one-seed juniper seedling survival under drought is directly related to age. Competition from grasses will limit regeneration but the impact will decline once the tree roots have grown below the zone of highest grass root concentration.

Vegetative regeneration commonly has been found in alligator juniper which sprouts from stems, roots, or the root crown after the removal or death of the main trunk. Jameson and Johnsen (1964) indicated that the ability to sprout declined as stump diameter increased. Some sprouting has been reported for one-seed juniper (Gottfried 1989).

Both pinyon and junipers exhibit slow early growth under natural conditions. Colorado pinyon seedlings can put on between 2.5 and 5.0 cm of top growth annually (Ronco 1990), while taproot growth is about 18 to 28 cm during the first year (Harrington 1987). One-seed juniper can grow approximately 3 cm in the first 20 months in the field (Meagher 1943) while roots can be 23 cm long after 3 months (Johnsen 1962). Top growth of 4 cm in the first year has been reported for Utah juniper (Meagher 1943) and 30 cm in 8 years has been reported for Rocky Mountain juniper (Noble 1990). Actual growth rates will depend largely upon site conditions with the best growth occurring on the moister sites (Gottfried 1987).

Growth of Older Trees

Growth of older pinyon and the junipers also is relatively slow, with best growth occurring on the more moist sites. Pinyon saplings will grow about 10 to 15 cm in height annually and mature trees will

grow 5 to 10 cm annually (Ronco 1990). Little (1987) followed the growth of pinyon trees near Santa Fe over a 47-year period beginning in 1938, when the trees were between 5 and 6 m tall. He reported average annual height growth of 3 cm and average annual growth at b.h. of 0.15 cm or 2.5 cm every 16.8 years. On better sites, pinyon can grow to 30 cm in diameter within 150 years (Ronco 1990).

Junipers generally grow slower than pinyon (Conner et al. 1990). Junipers have the ability to grow when conditions are favorable and to stop growing when conditions are unsatisfactory (Johnsen 1962). This is probably the reason that numerous false and missing rings are characteristic of these species. Growth rates for the main juniper species vary by species and generally decline with age.

Root growth varies by species. Pinyon has both lateral and vertical root systems. Lateral roots usually are found at depths of 15 to 41 cm and can extend away from the bole by a factor of two times the crown radius (Ronco 1990). Junipers have both tap and lateral roots; tap roots can be from 0.5 to 3.7 m in depth, while laterals usually are concentrated in the top 90 cm of soil. One-seed juniper lateral roots are about 2.5 to 3 times as long as the tree is tall (Gottfried 1989). Roots can occupy most of the interspace areas where they mine soil nutrients and moisture.

Stand Characteristics and Productivity

Most pinyon-juniper stands in the Southwest are uneven-aged (Barger and Ffolliott 1972). Woodland stand productivity is variable. In New Mexico, conifer woodlands contain approximately 2.3 billion trees; 62 percent are pinyon (Van Hooser et al. 1993). About 28 percent of the trees are less than 8 cm in diameter at root collar (d.r.c.) and 86 percent are less than 28 cm at d.r.c. Net volume in New Mexico is 144,000,000 m^3, of which 53 percent is pinyon (Van Hooser et al. 1993). Pinyon and juniper volumes can be calculated using tables and equations developed by Chojnacky (1985). Schuler and Smith (1988) suggested that the higher size/density, leaf area, and growth relations in mixed woodland stands than in pure stands can be related to differences in rooting habits and water relations between pinyon and junipers. High growth rates are found on better sites; for example, an Arizona alligator juniper stand had a net annual growth of 1.4 m^3/ha (Gottfried and Ffolliott 1995), and a pinyon-juniper stand at Zuni, New Mexico, had an annual growth of 1.0 m^3/ha (B. Schwab, personal communication, USDI Bureau of Indian Affairs, 1994).

Damaging Agents

A number of insects attack pinyon (Ronco 1990), including the pinyon sawfly (*Neodiprion edulicolus*), pinyon tip moth (*Dioryctria albovittella*), and the pinyon needle scale (*Matsucoccus acalyptus*). The cone moth (*Eucosma bobana*) is particularly damaging to pinyon. A number of bark beetles attack pinyon; for example, severe mortality associated with the pinyon Ips (*Ips confusus*) recently has been observed in areas of the Apache-Sitgreaves National Forests (Wilson and Tkacz 1992). The mortality may have been associated with an extended drought period that weakened the trees' defenses. Pinyon dwarf mistletoe (*Arceuthobium divaricatum*) is an important parasite that can cause locally severe damage and mortality.

The junipers have their own suite of insect problems and diseases (Gottfried 1989). Some examples are twig beetles (*Phloeosinus* sp.) and twig girdlers (*Stylox* sp.). Rusts (*Gymnosporangium* sp.) attack most junipers, causing witches' brooms, galls, leaf damage, and branch excrescences. True mistletoes (*Phorandendron* sp.), which are spread by birds, also are common parasites but generally do not cause heavy damage.

Additional information about the ecology of pinyon-juniper woodland ecosystems can be found in a number of sources. The autecology of pinyon has been reviewed by Gottfried (1987) and Ronco (1990) and juniper by Gottfried (1989), Johnsen (1962), Johnsen and Alexander (1974), and Noble (1990). Papers included in the proceedings of the 1986 pinyon-juniper conference at Reno (Everett 1987), the 1993 pinyon-juniper symposium at Santa Fe (Aldon and Shaw 1993), and the 1994 pinyon-juniper symposium in Flagstaff (Shaw et al. 1995). Evans (1988) and Gottfried (1992a) also are informative.

Biodiversity

Significant biological variability (biodiversity), as indicated by stand types, relative abundance of species, and species richness, exists in pinyon-juniper woodlands. Many of the pinyon-juniper habitat types that have been described (Moir and Carleton 1987) are present across a diverse range of landscape conditions in the Middle Rio Grande Basin. Biodiversity at any location is a result of many factors, including:

site characteristics like topography, geology, soils, climate; specific site history (Hamburg and Sanford 1986; Ricklefs 1987); successional state; and disturbance processes, which are central in determining the structure and function of ecosystems. Disturbances affect ecosystems at multiple spatial and temporal scales, creating variable conditions (niches), which allow multiple species to co-exist in the same area. Any brief review of pinyon-juniper biodiversity will contain generalizations across the range of existing conditions.

Biodiversity is often considered a function of species richness (number of species) within or between habitats. By this criterion, pinyon-juniper woodlands might be thought of as relatively unimpressive reservoirs of biodiversity, since the visual impression of many woodlands is of uniform conditions, with overstories dominated by only a few species of conifers. These woodlands also harbor relatively few endemic vertebrate species (Brown 1982). However, a broad and detailed examination of the woodlands reveals significant levels of biodiversity in less prominent ecosystem components, particularly herbaceous vegetation and soil organisms.

In pinyon-juniper woodlands, floristic diversity primarily reflects the herbaceous components of the system rather than the several species of pinyon and juniper that dominate many sites. About 450 species of vascular plants, out of a total of 722 species documented at the Bandelier National Monument, New Mexico, occur in pinyon-juniper and juniper woodland zones (Jacobs 1989). Barnes (1983), also working at Bandelier, found 7 shrub taxa, 25 forbs, 21 grasses, and 7 cacti in one survey, while another inventory found 12 shrub species, 47 forbs, 27 graminoids, and 6 cacti (C. D. Allen [unpublished data]). Two other examples are Mesita de los Ladrones, a 405 ha research natural area (RNA) in an open woodland on the Santa Fe National Forest near Pecos, that has at least 100 forb and 36 grass species, and Comanche Canyon, a proposed RNA on 210 ha near El Rito on the Carson National Forest, that has at least 6 tree taxa, 12 shrubs, 31 forbs, and 15 grasses (E. Muldavin, personal communication, The Nature Conservancy, 1994). All of these sites have been little-grazed by domestic livestock in recent decades, although Bandelier was heavily affected by feral burros. Vascular plant richness may generally be lower than these values suggest for most woodlands in the Middle Rio Grande Basin due to historic changes in these ecosystems associated with domestic livestock grazing. Biodiversity has been modified through direct and indirect introduction of alien species and genotypes; at least 20 percent of 722 species at Bandelier are aliens. Some species may have been introduced during aerial seeding, which included non-native genotypes and weed seeds, after the 1977 La Mesa Fire.

Emphasis is placed on soil biota in pinyon-juniper woodlands because of their species richness and critical functional role in ecosystems (Whitford 1991), as well as our relative ignorance of subsurface patterns and processes. A great diversity of microhabitats exists within soils, with a resultant diversity of organisms (Dindal 1990). Whitford (1991) provides an overview of pinyon-juniper soil biota, while the diversity and ecological role of the similar communities of soil-associated invertebrates from arid deserts that adjoin pinyon-juniper woodlands are reviewed by Crawford (1986, 1990). Surface dwelling arthropods recently were sampled in woodlands at Pecos National Historical Park (Parmenter and Lightfoot 1994) and Bandelier National Monument (Lightfoot and Parmenter 1994), with 189 and 115 species reported, respectively.

Soil organisms affect numerous ecosystem processes (Hole 1981; Crawford 1986), including: recycling of plant litter by detritivores (notably springtails in pinyon-juniper woodlands); controlling the rate of nutrient cycling, especially through eating fungi, which are the predominant decomposers (Parker et al. 1984); plant productivity; site hydrology through effects on vegetation and by altered soil porosity through burrowing actions; soil-forming processes through mixing and mounding soil; and consumption of live and dead organisms, especially underground plant parts (like roots). For example, 1990 was a big year for the emergence of cicadas in the Jemez Mountains, with emerging densities of over 25,000 per hectare in much of Bandelier's pinyon-juniper woodlands. This single species has significant effects on local woodland ecosystems, ranging from years of feeding on the roots of perennial plants (including the trees) and the alteration of nutrient cycling and soil physical conditions by their subsurface activities. The effects of harvester ants (*Pogonomyrmex occidentalis*) on the vegetation and soils of local pinyon-juniper woodlands also have been documented (Carlson 1988).

Microbiotic, or cryptogamic, crusts are important features of pinyon-juniper woodlands. These crusts are composed of varying species of cyanobacteria

with lichens, mosses, green algae, fungi, and bacteria (West 1990; Belnap 1990). The cyanobacteria, which have bundles of filaments with sticky, hydrophilic, polysaccharide sheaths (Belnap and Gardner 1993), serve to bind soil, hold nutrients and water, fix atmospheric nitrogen (Loftin and White, in review), and colonize disturbed sites (i.e., initiate primary succession). These crusts are readily damaged by mechanical disturbance, like hoof action or off-road vehicles. More data on the ecological role played by microphytic crusts are needed, given the widespread, but unsubstantiated, belief among many range managers that breaking up such crusts by livestock hoof action can be beneficial (Brown 1994).

Pinyon-Juniper Associated Wildlife

The wide variety of habitats within the pinyon-juniper ecosystem supports at least 70 species of birds and 48 species of mammals (Findley et al. 1975; Balda 1987). Although a few of these species are obligate to pinyon-juniper, most can be found to some degree in adjacent ecosystems. Whether an animal is present or absent or a permanent, summer, or winter resident depends on the species, geographic location, and the type of pinyon-juniper habitat.

Birds that have been found to breed only within pinyon-juniper habitats, in spite of other available habitats, include the screech owl (*Otus asio*), gray flycatcher (*Empidonax wrightii*), scrub jay (*Aphelocoma coerulescens*), and the plain titmouse (*Parus inornatus*). Semi-obligatory species, birds that breed in pinyon-juniper and only one other habitat type, include the pinyon jay, ash-throated flycatcher (*Myriarchus cinerascens cinerascens*), bushtit (*Psaltriparis minimus*), mockingbird (*Mimus polyglottis leucopterus*), black-throated gray warbler (*Dendroica nigrescens*), house finch (*Carpodacus mexicanus frontalis*), rufous-sided towhee (*Pipilo erythrophthlamus*), and lark sparrow (*Chondestes grammacus strigatus*) (Balda and Masters 1980). The gray flycatcher and black-throated gray warbler are inhabitants of mature pinyon-juniper woodlands and are 5th and 15th, respectively, on New Mexico Partners in Flight's prioritized list of neotropical migratory species. Many other species of birds, including raptors, songbirds, corvids, and hummingbirds, breed within pinyon-juniper woodlands but not exclusively.

Although the pinyon jay will forage and nest in ponderosa pine and pinyon and juniper trees (Balda and Bateman 1971), this species, with its physiological and behavioral adaptations for harvesting pinyon nuts, has a life history that is more strongly interwoven with pinyon than is any other avian species. The species' strong, sharply tapered bill and lack of feathers around the nostrils are adaptations for chiseling into unopened cones and reaching between cone scales for nuts without pitch soiling the facial feathers (Balda 1987). For the pinyon jay, the nuts, which are cached singly in many locations, provide a high energy food source for the birds during the winter, and seeds that escape consumption may germinate to produce a new generation of trees.

Water availability, juniper berries, and pinyon nut crops are major factors determining which nonresident species will overwinter in pinyon-juniper woodlands. Good pinyon nut crops in winter may attract red crossbills (*Loxia curvirostra*) and Cassin's finches (*Carpodacus cassinii*), in addition to the resident pinyon jays, scrub jays, Steller's jays and Clark's nutcrackers (Balda 1987). Juniper berry crops may also determine densities of overwintering birds that consume berries and/or seeds, such as Townsend's solitaires, western and mountain bluebirds (*Sialia mexicana bairdi* and *S. currucoides*), and robins (*Turdus migratorius*) (Balda 1987). Merriam's turkeys (*Meleagris gallapavo merriami*) occupy many pinyon-juniper sites where ponderosa pine is available for roost sites (Scott and Boeker 1977). They prefer pinyon seeds but juniper seeds are used during drought periods and when pine and oak seed production is low.

Many species of bats are associated with pinyon-juniper habitats and have been identified by mistnetting over watering tanks, ponds, streams, and other permanent sources of water. Bats that have been commonly captured in pinyon-juniper habitats include eight species of *Myotis*, big brown bats (*Eptesicus fuscus*), spotted bats (*Euderma maculatum*), western pipistrelles (*Pipistrellus hesperus*), and pallid bats (*Antrozous pallidus*) (Findley et al. 1975). Other occasional captures have included big-eared bats (*Plecotus townsendii*) and brazilian free-tailed bats (*Tadarida brasiliensis*). Female hoary bats (*Lasiurus cinereus*) have been seen migrating through New Mexico in the spring and fall, whereas males are commonly found in pinyon-juniper woodlands in the summer. Silver-haired bats (*Lasionycteris noctivagans*) are found in pinyon-juniper woodlands as well as in other habitat types; however, these bats may move to more northern states in midsummer (Findley et al. 1975). Because the mentioned bat species have been captured in pinyon-juniper woodlands by

mistnetting at night, they must use the woodlands for foraging and/or water. Little else is known about the species' use of pinyon-juniper woodlands, i.e., whether they roost in pinyon and juniper trees and, if so, in what structures, whether they feed exclusively in the woodlands, and whether they overwinter/hibernate in the woodland habitats. Although it is likely that rock cliffs, tree branches and bark, and hollows of mature pinyon and juniper trees provide roost sites for many of these species, few studies have been undertaken to prove such hypotheses or investigate other habitat requirements and associations of these bat species.

Many species of small mammals may be found in pinyon-juniper habitats; species composition depends on the mix of vegetation, cover, elevation, soil, and other factors. Many of these species are present in pinyon-juniper stands only at the periphery of their ranges (primarily ponderosa pine or grassland distributions) or have broad distributional ranges that merely include pinyon-juniper woodlands. Species that have distributions centered in pinyon-juniper woodlands include cliff chipmunk (*Tamisas dorsalis*), rock squirrels (*Spermophilus variegatus*), brush mice (*Peromyscus boylii*), pinyon mice (*P. truei*), rock mice (*P. difficilis*), and white-throated (*Neotoma albigula*) and Mexican (*N. mexicana*) woodrats (Findley et al. 1975). Pinyon mice are more or less restricted to pinyon-juniper woodlands and are often the most common small mammal in open stands of this vegetation type (Findley et al. 1975; Short and McCulloch 1977). Abundances of pinyon mice were shown to decline in pinyon-juniper habitats where the overstory was completely removed (Severson 1986a). Populations of pinyon mice, as well as other seed predators, probably explode during mast years. How these fluctuations cascade through the ecosystem remains largely undetermined. Brush mice, essentially oak specialists, can become the predominant species in lower elevational habitats lacking pinyon or in pinyon-juniper habitats with an evergreen oak and shrub understory (Findley et al. 1975). Rock mice and rock squirrels are more common amongst rocks, boulders, and broken terrain within the pinyon-juniper woodlands.

Species that may be found at the periphery of their grassland/desert distributions in the more open pinyon-juniper habitats include white-tailed antelope squirrel (*Ammospermophilus leucurus*), Texas antelope squirrel (*A. interpres*), silky pocket mice (*Perognathus flavus*), Plains pocket mice (*P. flavescens*), Ord's kangaroo rat (*Dipodomys ordii*), and desert cottontail (*Sylvilagus audubonii*). More mesic sites where pinyon-juniper grades into and intermingles with ponderosa pine and mixed conifer species may support eastern cottontails (*Sylvilagus floridanus*), Colorado chipmunks (*Tamias quadrivitattus*), deer mice (*Peromyscus maniculatus*), and Mexican woodrats (*N. mexicana*) (Findley et al. 1975).

The invertebrate and small mammal communities sustain a number of avian and mammalian predators. Mammalian predators may include coyote, gray fox (*Urocyon cinereargenteus*), ringtail (*Bassariscus astutus*), long-tailed weasel (*Mustela frenata*), western spotted skunk (*Spilogale gracilis*), striped skunk (*Mephitis mephitis*), hog-nosed skunk (*Conepatus mesoleucus*), mountain lion (*Felis concolor*), and bobcat (*Felis rufus*) (Findley et al. 1975). Skunks, ringtails, coyotes, and gray foxes may also include fruits and other vegetative matter in their diets, depending on availability and season. Avian predators that hunt small mammals and/or birds within pinyon-juniper woodlands may include golden eagle (*Aquila chrysaetos canadensis*), Swainson's hawk (*Buteo swainsoni*), Cooper's hawk (*Accipiter cooperii*), kestrel (*Falco sparverius sparverius*), red-tailed hawk (*B. jamaicensis*), and great-horned owl (*Bubo virginianus*) (Frischknecht 1975). Although individual predators might forage, breed, and maintain territories in pinyon-juniper habitats alone, all of these predatory species have distributions that extend beyond the pinyon-juniper ecosystem into grasslands, deserts, and/or forests (Findley et al. 1975).

Other year-round residents in pinyon-juniper stands are mule deer (*Odocoileus hemionus*), white-tailed deer (*O. virginianus*), and elk (*Cervus elaphus*), all of which consume leaves and fruits of pinyons and junipers (Martin et al. 1961). Deep snows in the higher elevation forest zones may force additional deer and elk down from these higher elevations into pinyon-juniper habitats during the winter. Although forbs and grasses are utilized by deer during the spring and summer, browse from dwarf trees and shrubs are more important year-round. Elk also utilize forbs in the summer, grass in the summer and winter, and shrubs to some extent throughout the year. Some shrubs and dwarf trees found in the mid- and understory that are important to these cervids are mountain mahogany (*Cercocarpus breviflorus*), desert ceanothus (*Ceanothus greggii*), shrub live oak (*Quercus turbinella*), wavyleafed oak (*Q. undulata*), Gambel oak (*Q. gambelii*), and cliffrose (Short and McCulloch 1977). Bighorn sheep (*Ovis canadensis*

canadensis) may also utilize pinyon-juniper habitat where there is rough terrain suitable to their habitat needs (Short and McCulloch 1977).

Although only a few species of wildlife rely solely on pinyon-juniper habitats for suitable habitat, many species utilize the wide array of habitat types found in this ecosystem. Pinyon and juniper trees provide nesting material and sites for birds and mammals, hiding cover, and valuable year-round food resources in the form of leaves, nuts, and berries. During temperature extremes of the summer and winter, pinyon-juniper habitats also provide thermal protection, the degree of which is determined by canopy closure. On the other hand, birds and mammals are important to pinyon and juniper trees as agents of seed dispersal. Mid- and understory trees, shrubs, and grasses, the quantity of which is inversely related to canopy closure, provide additional food resources, hiding cover, and nesting cover for other wildlife species.

IMPACTS ON PINYON-JUNIPER ECOSYSTEMS

Climatic Variability

Climatic fluctuations throughout time have influenced changes in the distribution and density of pinyon-juniper woodlands. Fluctuations during prehistoric periods have been greater than during historic times. During the Late Pleistocene (40,000–11,000 years ago), for example, pinyon-juniper woodlands occupied lower elevation sites, which currently support deserts, grasslands, or scrublands (Betancourt 1987; Betancourt et al. 1990, 1993). Drier and warmer conditions during the Holocene caused the woodlands to retreat northward and upward in elevation until the woodlands occupied their present locations during the late Holocene (the last 4,000 years). Changes during the late Holocene are presumably due to climate variability at an interdecadal scale, which is currently receiving more attention by climatologists (Karl 1988; Guetter and Georgakakos 1993; Kushnir 1993; Xu 1993; Graham 1994; Miller et al. 1994a).

Interdecadal climate variability, which produces prolonged episodes of wet and dry conditions in the central Rio Grande Basin, has left a strong imprint on the demography of pinyon-juniper woodlands. For example, a less sinuous and more northerly polar jet stream, a weakened subtropical jet stream, and a decrease in the frequency of El Niño events contributed to a subcontinental-scale drought in the middle of the century. The 1950s drought, an event with a 200 to 500-year return period (Stahle and Cleaveland 1988), caused massive die-off of pinyons and junipers throughout the Middle Rio Grande Basin (Betancourt et al. 1993). The drought probably reset demographic clocks, created disequilibria in carbon and nutrient cycling, accelerated shrub invasions into grasslands, and shifted ecotonal boundaries that had been stable for several millennia. Data indicate that the pinyon mortality associated with this outbreak caused the ecotones between pinyon-juniper and juniper woodlands and between the pinyon-juniper and ponderosa pine forests to shift sporadically about 100 m horizontally upslope in the Frijoles watershed of the Jemez Mountains. A similar impact on pinyon populations has been documented at the Sevilleta Long-Term Ecological Research Site, near Socorro, New Mexico (Betancourt et al. 1993). The intense drought may have further reduced the herbaceous ground cover enough to initiate and exacerbate desertification and erosion processes. The return of wet conditions beginning in the late 1970s is likely responsible for the recent recovery of range conditions and may have produced a surge of recruitment of pinyon and junipers throughout the region.

During the last 50 years, the Middle Rio Grande Basin has experienced one of the driest (late 1940s to early 1960s) and one of the wettest (1976 to present) episodes of the last few hundred years, according to the tree-ring record (Fig. 2). The latter period is now gaining a lot of notoriety in the global change arena. There appears to have been a step change in global climate since about 1976, including record warmth that is related, in part, to an almost permanent shift to El Niño-like conditions (Ebbesmeyer et al. 1991; Miller et al. 1994b). This shift suggests that the tropical Pacific may be the pacemaker for the global system at interdecadal scales; climatologists are now entertaining the possibility that an enhanced greenhouse effect accelerates the tropical heat machine (Latif and Barnett 1994; Kumar et al. 1994). Thus pinyon-juniper woodlands in the Middle Rio Grande basin may already be responding to climatic effects from greenhouse warming, in the form of increased cool season precipitation with persistent El Niño conditions in the 1980s and 1990s.

Fire

Fire was the most important natural disturbance in the pinyon-juniper woodlands before the introduction of large herds of livestock in the 19th century.

Although ecologists and managers have long recognized that fire was an important factor in presettlement dynamics of the pinyon-juniper type (Leopold 1924), there is little specific data documenting the range and variability of past fire regimes. There are only a few studies located in the upper border of the pinyon-juniper zone, where it occurs with ponderosa pine, that clearly document the frequency, extent, seasonality, or other presettlement or long-term fire regime patterns (Allen 1989; Despain and Mosley 1990; Swetnam and Baisan 1995).

The evidence of past fire occurrence is visible in many stands as the presence of charcoal in the soil; charred, remnant juniper snags or stumps; and fire scars on living junipers and pinyons. Fire scars on living pinyons are generally rare, especially in comparison with scar abundance in higher elevation ponderosa pine and mixed conifer stands (Swetnam 1990; Swetnam and Baisan 1995). The rarity of pinyon fire scars may be due to high susceptibility of pinyon boles and crowns to damage by surface fires; trees are either killed outright, or do not live long after being scarred. Heart-rotting fungi may enter the fire-scar wounds to hasten mortality. In spite of the poor preservation of the record from fire-scarred pinyon, several specimens from New Mexico have been dendrochronologically dated; one tree located in a stand that contained ponderosa pine had 11 fire scars over a period of 200 years (Swetnam [unpublished data]). In contrast to pinyon, fire-scarred and fire-charred junipers have often been noted but rarely systematically sampled or quantitatively analyzed. Unfortunately, as previously mentioned, junipers in the Southwest cannot be accurately dated because of numerous false and missing rings.

One of the most detailed and informative fire-scar-based studies in southwestern pinyon-juniper woodlands was conducted at Walnut Canyon National Monument near Flagstaff, Arizona (Despain and Mosley 1990). Dead, fire-charred junipers were sampled within a 300 ha stand and a fire chronology was also constructed from fire-scarred ponderosa pine trees in an adjacent stand. Results from the analyses indicated a surface fire interval of approximately 20 to 30 years. Three other fire history studies in New Mexico, based on fire-scarred ponderosa pine trees scattered within the pinyon-juniper woodlands, indicate that stand-wide fires, those burning more than 10 ha, occurred about every 15 to 20 years on the average. These studies were conducted in the Jemez Mountains (Allen 1989), Organ Mountains (Morino and Swetnam [unpublished data]; Swetnam and Baisan 1995), and El Malpais National Monument (Grissino Mayer and Swetham, in press).

Dense pinyon-juniper stands (approximately 1,110 trees per hectare) can burn as crown fires under extreme weather conditions. Wright and Bailey (1982) reported that such stands will burn when relative humidities are lower than 30 percent and winds exceed 55 km per hour. The key conditions for such burns are sufficient canopy closure to promote fire spread between trees, abundance of dead woody fuels on the surface and as standing snags, and extreme weather conditions. Hence, it appears that presettlement pinyon-juniper fire regimes were a mixture of surface and crown fires and of variable intensity and frequency and depended largely on site productivity. Productive sites, such as at Walnut Canyon, probably sustained patchy surface fires at intervals of 10 to 50 years. Some of these stands attained densities sufficient to carry crown fires at intervals of 200 to 300 years or longer.

On less productive sites with discontinuous grass cover, fires were probably very infrequent, and burns were small or patchy when they did occur. In sites with relatively continuous grass cover, frequent widespread fires (10-year intervals or less) probably maintained grasslands or savannas, with pinyons and junipers restricted to rocky outcrops and microsites where grasses were discontinuous. Savannas were maintained because fires tended to kill trees less than about 1 m tall (Johnsen 1962).

The Sevilleta National Wildlife Refuge and adjacent mountains in the central Rio Grande may be an area where the full range of possible fire regimes in pinyon-juniper existed in the past. This area encompasses the National Science Foundation's Sevilleta Long Term Ecological Research site. The Manzano and Los Pinos Mountains border the east and northeast sides of the area. These ranges extend along a north-south axis on the east of the Rio Grande. Today, grasslands and creosote stands occur from the lowest elevations at about 1,450 m at the Rio Grande up to the foothills of the mountains at about 1,800 m. Livestock grazing has been excluded within the Refuge since the late 1970s. As a result, grass cover has expanded and increasingly large grass fires ignited by lightning have burned during the summer months. Most of the larger burns (e.g., greater than 1,000 ha) were extinguished by managers. Some of these fires would have burned substantially greater areas if they had been allowed to, and in some cases,

they would have burned up into the extant pinyon-juniper (or pure juniper) savannas and woodlands. At the lower elevations, grasslands currently extend up to the steep rocky escarpments on the west sides of the Manzano and Los Pinos Mountains, while in other areas, the grasslands ascend bajadas, gentle ridges, and canyon bottoms up onto the mountains where scattered junipers and pinyons form savannas that transition into woodlands. Confirmation of lower densities in the past, however, awaits demographic studies of the kind now being conducted at the Sevilleta Long Term Ecological Research site.

Before the advent of intensive livestock grazing, we suspect that, during certain years, very large areas burned in the grasslands, adjacent savannas, and woodlands. These years of extensive burning were probably dry, and they may have followed wet years when substantial grass and herbaceous growth was enhanced (Rogers and Vint 1987; Swetnam and Baisan 1995). Eyewitness accounts of burning in this area have not been found, but in other parts of the Southwest—such as in southern Arizona and southeastern New Mexico—newspapers reported "millions of acres" burning in grasslands and woodlands during the 1870s and 1880s (Bahre 1985).

Above the grasslands and savannas, many areas are very rocky with thin soils. Grass cover is sparse or nonexistent within most of the woodlands on the slopes and ridges, and it is unclear if these areas ever sustained adequate soil or moisture resources necessary for production of a more-or-less continuous understory cover that is needed to support spreading surface fires. Nevertheless, charred, ancient-looking juniper stumps are commonly seen within these stands. In the Los Pinos Mountains, at an elevation of 1,900 to 2,100 m, a number of fire scars have been discovered that are completely grown over within the stems of living and dead pinyons (Swetnam and Betancourt [unpublished data]). Many of these scars date to the year 1748. Other dendrochronological studies indicate that this was one of the driest years in the Southwest in the past 300 years, and it followed one of the wettest years (1747). Moreover, this was probably the largest regional fire year to occur in the Southwest in several centuries; it is recorded by fire scars in 41 of 63 sites where fire chronologies have been reconstructed (Swetnam and Betancourt 1990, Swetnam and Baisan 1995). Thus, it seems that fire-free intervals in these less productive pinyon-juniper sites were very long (i.e., greater than 100 years), but in unusual climatic conditions, spreading surface or canopy fires did occur.

The other extreme of pinyon-juniper productivity can be observed on the east side of the Manzano Mountains. On this side of the mountains, where summer rainfall is apparently greater than on the west side, dense, closed-canopy pinyon-juniper stands cover large areas. Mixed within these stands are numerous homes that will probably be consumed by future catastrophic crown fires. Such burns have already occurred in some locales on the mountain, prompting the Cibola National Forest, and the Department of Defense, which has jurisdiction over a "military withdrawal area" within the Manzanos, to expend considerable resources in preparing fuel breaks. It is unknown if these more productive pinyon-juniper stands sustained surface fires in the past, but fire-scar studies in adjacent ponderosa pine stands suggest that they did. Despite uncertainty about the frequency or extent of fires, it is very likely that the densities of these productive stands have increased in the late 20th century compared to the 19th and earlier centuries when surface fires, and extensive fuelwood cutting for Albuquerque, probably maintained more open stand conditions.

Succession

Soil development is slow under semi-arid climates with accumulations of many soil properties taking from 1,000 to 100,000 years. Thus, most soils in the pinyon-juniper woodlands are not necessarily in equilibrium with modern climate and vegetation. Many of the soils that may have developed under mixed conifer forests during the last glacial period were later affected by Holocene erosion, approximately 8,500–6,000 years ago; these same areas are now occupied by pinyon-juniper woodlands (S. Reneau, personal communication, Los Alamos National Laboratory, 1994).

There have been numerous studies of succession in old pinyon-juniper burns and tree control areas (Arnold et al.; 1964, Clary et al. 1974; Rippel et al. 1983; Severson 1986b). In woodlands, successive stages usually contain the same species but in different amounts and dominance (Evans 1988). Habitat type also will affect the successional process. Arnold et al. (1964) proposed one successional sere for the Southwest, which started with the establishment of annuals after a fire and progressed to the renewed dominance of the arboreal vegetation; other pathways have also been suggested for the woodlands of southwestern Colorado and western Utah (Evans

1988). Junipers can be the first tree species to invade an area, but they are often followed and replaced by pinyon. In Utah, tree dominance does not occur until 70 to 80 years following fire. Herbage yields decline as tree crown cover increases (Arnold et al. 1964). Succession has been more rapid in some cleared areas because of the presence of tree advance regeneration that survived the initial control treatment.

The invasion of native grasslands by woodland species has been a topic of concern and controversy. As noted above, fire was the dominant factor limiting the spread of trees and maintaining open stands where trees always dominated. However, most of the new tree establishment occurred in juniper savannas, as indicated by the presence of old trees, or in grassland inclusions within the woodlands, and little occurred in true grasslands (Johnsen 1962). The expansion of pinyon in some areas of New Mexico may actually be its reestablishment on previous woodland sites (Samuels and Betancourt 1982; Dick-Peddie 1993). The successful establishment of tree species indicates that they are adapted to existing site conditions, and while competition and fire may have limited their numbers, the invaded areas are climatically woodlands. Trees may also indicate that the initial habitat, especially as it affects species germination and establishment, has been modified and may no longer be optimum for the original mix of species (Dick-Peddie 1993). Some of the tree control operations of the 1950s and 1960s may have failed because woodland soil and microclimatic conditions no longer favored the establishment of seeded grasses.

The distribution and composition of plant communities are dynamic, varying in both time and space (Tausch et al. 1993). Climatic variability, as indicated above, has affected the distribution of pinyon-juniper woodlands within the Middle Rio Grande Basin. Natural changes may be subtle, occurring over a long period, or dramatic, as are the effects of the 1950s drought. Human management can alter site conditions in such a way that certain species can no longer maintain or reestablish themselves once a perturbation has been eliminated. Changes in environmental conditions can change dominance patterns and species compositions; there are several structurally and functionally similar plant communities that could become established on a site (Tausch et al. 1993). More information about ecological thresholds, multiple steady states, and multiple successional pathways is presented by Tausch et al. (1993).

Land Use History and Ecosystem Changes

Land use of pinyon-juniper woodlands by prehistoric and historic human societies has affected these ecosystems. The land use history of the Pajarito Plateau on the eastern flank of the Jemez Mountains, northwest of Santa Fe, is broadly similar to other portions of the Middle Rio Grande Basin. A review of this history is presented as an illustrative case study that probably reflects changes over a larger area. Extensive research has been conducted on the archeology (Head 1992; Mathien et al. 1993; Orcutt [unpublished manuscript]), general land use histories (Allen 1989; Rothman 1989; Scurlock and Johnson 1994), ecology (Barnes 1986; Padien and Lajtha 1992; Breshears 1993; Chong 1993), and hydrology (Wilcox et al., in press) of Pajarito Plateau woodlands. Synthesizing this information with research results from other areas (e.g., Rogers 1982; West and Van Pelt 1987; Cartledge and Propper 1993; Betancourt et al. 1993; Miller and Wigand 1994) leads to the general scenario noted in figure 3 of changes in local pinyon-juniper woodlands.

American Indian effects on local woodlands are thought to have been insignificant or highly localized until the late 12th century, when the Anasazi population began to build markedly (Orcutt [unpublished manuscript]). Recent archeological survey and excavation work at Bandelier shows evidence of extensive Anasazi impacts on woodland resources during the peak occupation period of A.D. 1200–1500. Cutting and burning of pinyon and juniper trees for cooking, heating, building, and agricultural activities likely led to significant deforestation of upland mesas during this time, and local ungulate (primarily mule deer) and rabbit populations may have been reduced by hunting pressure (Kohler 1992).

The overall ecological effect of several centuries of Anasazi occupation of the Pajarito Plateau may have been to favor herbaceous vegetation at the expense of the woodland trees. Intensive soil disturbance certainly occurred in farmed areas and around habitations. But there probably was little net change in landscape-wide erosion rates due to the small size and dispersed location of farm "fields" and habitations and the effectiveness of herbaceous vegetation at protecting soils from erosion. Perhaps, the lack of well-developed, old-growth pinyon-juniper woodland in the Bandelier area can be partly traced to the slow recovery of woodlands from the effects of Anasazi deforestation.

Conceptual Model of Piñon-Juniper (PJ) Ecosystem Changes on the Pajarito Plateau

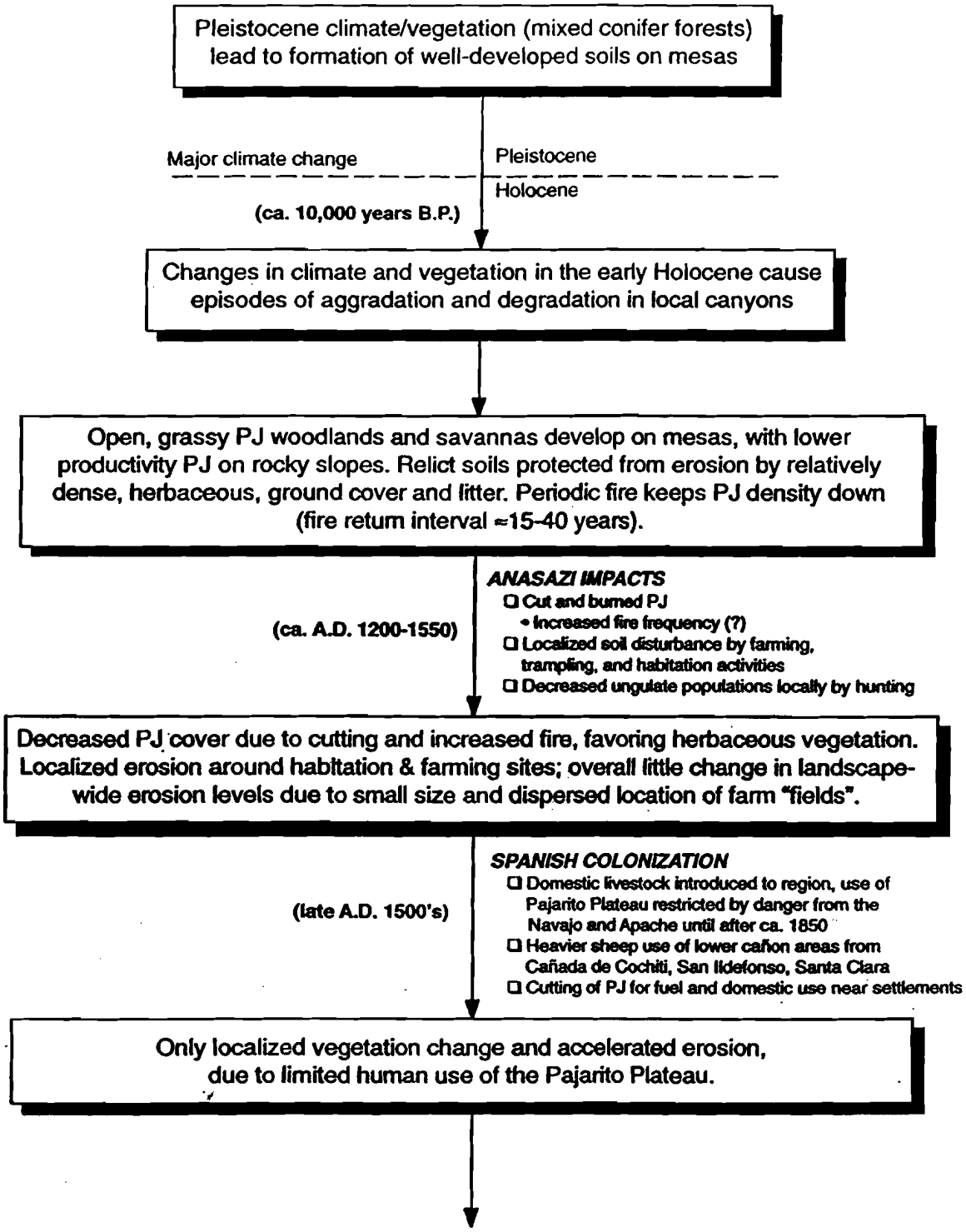

Figure 3.

continued on next page

(post A.D. 1880)

ANGLO SETTLEMENT

- ❑ Continued increase in livestock numbers, especially in 1880's after railroads access area, large numbers of sheep and cattle are brought to the Pajarito Plateau; widespread use of the plateau for intensive grazing through 1930's
- ❑ Extreme droughts in 1890's and 1950's
- ❑ Selective cutting of PJ becomes more extensive through 1930's
- ❑ Widespread fires eliminated by intensive grazing; active fire suppression initiated and institutionalized ca. 1910
- ❑ Large numbers of feral burros in Bandelier National Monument from 1940's until 1983
- ❑ High deer numbers in Bandelier since 1940's
- ❑ Elk reintroduced, local population grows to thousands

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PJ ecosystems, especially understory/soil subsystems, forced into disequilibrium:

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- ❑ Decreased ground cover by grazing and hoof action from ungulates, rodents, rabbits, and ants
  - • Decreased vegetation and litter cover, particularly in the interspaces between trees
    - – Soil surface microclimate becomes more extreme
- ❑ Increased bare soil --> thresholds reached where precipitation becomes surface runoff instead of infiltration
  - • Less water available for shallow-rooted herbs
- ❑ Increased soil erosion --> leading to further increase in bare soil and surface runoff in a positive feedback cycle (note: the same magnitude rainfall event causes more erosion than before)
- ❑ Increased precipitation intensity (?) from ca. 1880-1930 may have further increased surface runoff
- ❑ Decreased fire frequency (as inadequate ground fuels remain to carry fire) allows unimpeded tree establishment
- ❑ Increased PJ tree density and PJ invasion of former grasslands downslope and ponderosa forests upslope
  - • Increased tree competition with herbs for water, nutrients, light, space (and allelopathic effects?)
    - – Further decreases in herbaceous vigor and cover reducing herbaceous controls on tree establishment
- ❑ Droughts may have further reduced herbaceous vegetation, perhaps triggering erosion on additional sites by causing thresholds of exposed soil to be exceeded
- ❑ Herbaceous plant re-establishment becomes very difficult, even after livestock are removed, due to dominance of physical processes in the desertified interspaces:
  - • Loss of organic matter litter/mulch at soil surface (therefore, drier surface soil and difficult surface microclimate), increased freeze/thaw soil displacement (disrupting seedling roots), nutrient/texture problems for seedling establishment in exposed clay-rich B-horizons, nutrients translocated and concentrated beneath trees, decreased seed sources, depleted soil seed pools, animal use of herbs and their limited seed sources (by ungulates, rodents, and ants)
- ❑ Soil erodes from uplands --> canyons --> Rio Grande/Cochiti Reservoir
- ❑ Bare soil --> bare rock, with even further increases in surface runoff and attendant erosion down-drainage

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Today: most local PJ woodland ecosystems are unstable from a soils perspective, with many moving towards "PJ rocklands"

Figure 3 (continued).

European settlement of the adjoining Rio Grande Valley, and the introduction of domestic livestock grazing, began in 1598 at today's San Juan Pueblo. During most of the historic period, livestock use of the Pajarito Plateau was apparently restricted by danger from Athabascan raiders (Navajo, Apache, Ute), although lower elevation areas near the valley's Spanish and Puebloan communities may have been heavily utilized by livestock (M. L. Smith [unpublished manuscript]). This constraint on use of the Pajarito Plateau eased by ca. 1850 but was not eliminated until the final suppression of Navajo raids in the 1860s. Still, at the time the Ramon Vigil Grant in the heart of the Pajarito Plateau was surveyed by the U.S. General Land Office in 1877, herbaceous vegetation was noted to be abundant, and livestock use was apparent. Cutting of pinyon and juniper for fuelwood and building materials was important in many areas on the Pajarito Plateau in the late 19th and early 20th centuries. The stubs of axe-cut junipers remain obvious in most pinyon-juniper woodlands north of Frijoles Canyon on the Plateau.

The development of railroad links to external commercial markets during the 1880s led to marked increases in the numbers of domestic livestock grazed on the Pajarito Plateau, beginning in 1880 on the Ramon Vigil Grant (Rothman 1989). Similar increases were reported from throughout New Mexico; there were about 1.3 million cattle and 3 million sheep in the state by 1888 (Wooton 1908). The resultant high intensity grazing apparently triggered ecological changes in local woodlands. Overgrazing caused sharp reductions in the herbaceous ground cover and associated organic litter, effectively suppressing the previously widespread fires. Cool season grasses, which green up early in the growing season, are thought to have been most affected by this year-long grazing pressure, so that today they are largely found only beneath the crowns of older woodland trees or on steep slopes. Reduced cover of herbaceous plants and litter led to decreased water infiltration and increased surface runoff from the typically intense local rainfall events, and thresholds were reached that initiated accelerated erosion. An increase in the intensity of extreme storm events in the late 1800s may have further exacerbated erosional processes. In any event, by 1913 grass cover was considered "scant" and a surveyor in 1938 identified inadequate water infiltration as a cause. Current woodland conditions in Bandelier display tree canopy coverage ranges of 12 to 45 percent, herbaceous plant coverage (basal intercept) of only 0.4 to 9 percent, and exposed soils covering between 38 and 75 percent of ground surfaces.

Overall, the most significant ecological changes in local pinyon-juniper woodlands in historic times involve diminished and altered herbaceous ground vegetation, fire suppression, increased tree densities, and accelerated soil erosion. However, the consequences and interrelationships among the variables are open to several interpretations.

Over the past century many young pinyon and juniper trees became established in the absence of thinning fires and competing herbaceous vegetation, with increases in tree density continuing to the present on mesic sites. Thus, tree densities increased within pinyon-juniper woodlands, while pinyon and juniper expanded their ranges upslope into ponderosa pine forests and juniper moved downslope into former grasslands. As these trees grew they became increasingly effective competitors for water and nutrients in the shrinking tree interspaces, directly limiting herbaceous plant establishment and growth and keeping much bare soil exposed; allelopaths in juniper needle litter may augment this process. While there are several reasons for the accelerated erosion within some watersheds, these changes apparently interacted as a positive feedback cycle in which decreased herbaceous ground cover promoted tree invasion and continued erosion, which in turn fostered further decreases in ground cover. Increased grazing pressure on residual herbaceous species and increased soil compaction in interspace areas by livestock and wildlife also contributed to the problem. As a result, large portions of the Pajarito Plateau are becoming pinyon-juniper "rocklands" as the soil mantle erodes away—this is most evident on the southerly, low elevation mesas of the park where shallower soils were already present before this modern erosion began.

The reestablishment of herbaceous ground cover under today's desertified mesa-top conditions is difficult. Heavy utilization of the current herbaceous vegetation by animals ranging from harvester ants and mice to increasing numbers of elk may be limiting the availability of seed sources in many woodland areas (Carlson 1988). Seedling establishment has been inhibited by changes in soil surface conditions. Losses of organic matter litter, which acts as a mulch, and porous nutrient-rich surface soils have resulted in reduced water infiltration and soil nutrient availability, and have caused the soil surface microenvironment to become more xeric and experience more

extreme temperatures. The relatively impermeable clay-enriched horizons that consequently are exposed present a more difficult nutritional and water-balance environment for prospective seedlings. Winter freeze-thaw activity churns the top soil layer and creates polygonal cracking patterns in bare soils that damage or kill the roots of seedlings that managed to establish successfully the previous summer. Interestingly, in Bandelier, herbaceous vegetation today is generally far more vigorous and dense on canyon walls than on the adjoining, eroding uplands, even on dry southerly aspects, as rock cobbles on the canyon slopes create a relatively stable, mulching substrate where adequate moisture and nutrients are available to the interspersed plants.

Once initiated, this pattern of desertification is apparently difficult to break (Evans 1988). Physical rather than biological processes now dominate these sites. Biological capital that once moderated the elemental forces has been dissipated, leaving harsh sites for plant establishment. Soils that likely formed under more mesic climate and vegetation conditions during the Pleistocene are eroding at unsustainable levels in many pinyon-juniper woodlands on the Pajarito Plateau today. For example, in 1993, erosion bridge measurements at 360 points on a 1 ha watershed in Bandelier revealed a mean degradation in the soil surface level of 0.34 cm between July and November (Wilcox et al. 1993). Also, herbaceous vegetation cover continues to decline in some areas (Potter 1985).

Simply eliminating the livestock grazing that apparently triggered the development of the current situation is not sufficient to halt the erosion. Livestock grazing ceased in 1932 over most of the Bandelier area in which erosion is currently occurring. However, the subsequent increase in the burro populations south of Frijoles Canyon may have contributed to soil degradation. North of Frijoles Canyon, where burros were never a significant factor, removal or reduction of domestic livestock was accomplished by 1943, with absolute exclusion of trespass livestock since the early 1960s; yet, serious erosion also occurs there and may even be more severe in areas like the detached Tsankawi Unit.

As a result of the pervasiveness of human activities in pinyon-juniper woodlands for most of the past millennia, current efforts by land management agencies to move toward ecosystem management, where desired conditions are based on the historical range of natural variability, will be challenging and possibly controversial. A better understanding of historic and prehistoric human interactions with the pinyon-juniper woodlands is needed to help define sustainable goals for managing these important ecosystems.

PINYON-JUNIPER WOODLAND MANAGEMENT

Woodland Control Programs

Livestock interests maintain that forage for livestock has declined as the pinyon-juniper type has increased in area and density since European settlement. While the invasion question is still being debated (Miller and Wigand 1994), research has demonstrated that total forage production declines as tree crown closure increases (Arnold et al. 1964).

In the period following World War II, efforts were started on western ranges to eliminate pinyon and junipers in favor of forage species. By 1961, 486,000 ha of Arizona pinyon-juniper lands had been treated using a variety of techniques such as cabling, bulldozing, individual tree burning, grubbing, and chopping (Cotner 1963). The treated sites were usually seeded with grasses once trees were removed. However, the number of hectares treated annually in Arizona began to decline by the late 1950s as the availability of productive and easily treated areas declined (Cotner 1963).

The value of pinyon-juniper control efforts has been controversial. Arnold and Schroeder (1955) indicated that herbage yields could be increased by removing juniper trees. Clary (1971) also reported increases in understory vegetation following removal of Utah juniper in Arizona; however, yields of seeded exotic grasses declined after 4 to 6 years, while native species tended to increase slowly over time. Seeding was generally unsuccessful in large conversion areas. Successful herbage production following tree removal depends on annual precipitation, pretreatment tree cover, and on pretreatment soil nitrate-nitrogen content (Clary and Jameson 1981). Production was also lower on soils derived from limestone. An increase of between 0.5 and 0.8 AUM per hectare was indicated for the most successful projects (Clary et al. 1974). The benefits of many treatments have declined over time. In a New Mexico study, Rippel et al. (1983) evaluated a treated area after 20 years, and found that the cover of grasses and forbs was greater in an undisturbed pinyon-juniper stand than in the cabled area.

Control programs were also justified by the assumption that they increased water yields. The hypothesis held that replacing comparatively deep-rooted trees with shallower-rooted grasses would result in decreased evapotranspiration and increased runoff, which would eventually reach downstream reservoirs. However, while this mechanism works in vegetation types found on moister sites, the basic moisture requirements on dry sites are similar regardless of vegetation, and one vegetation type is about as efficient at using available moisture as another. Little opportunity exists for streamflow augmentation on warm, dry sites where annual precipitation is less than 460 mm and is exceeded by potential evapotranspiration (Hibbert 1979). Most pinyon-juniper woodlands fall into this category. Watershed research in Arizona at Beaver Creek (Clary et al. 1974) and at Corduroy Creek (Collings and Myrick 1966) failed to show significant water yield increases following control treatments. The only experiment to demonstrate an increased water yield (about 5 mm in an area where average annual precipitation was 463 mm) utilized aerial spraying of herbicides and did not allow the immediate harvesting of the dead, standing trees (Baker 1984). However, results from Beaver Creek (Clary et al. 1974) indicated that soils deeper than 30 cm within treated areas retained more soil moisture than did similar soils in untreated areas. This additional moisture would benefit vegetation on the site even if it did not contribute to streamflow.

There is a common belief that the active erosion and gullying observed in the woodlands and the related decline in long-term site productivity are the result of the tree cover (Gifford 1987). However, Gifford (1987) stated that there is no evidence to support this hypothesis and, in fact, existing limited research indicates otherwise. Erosion is a natural process but has accelerated because of reduced vegetation cover and overuse of channels and wet areas by livestock. The role of trees in soil stabilization is often ignored in the pinyon-juniper woodlands, although trees are planted throughout the world for this purpose. Reduced infiltration is one cause of overland flow and accelerated erosion. Infiltration rates are similar in wooded and chained areas (Evans 1988). Plots with pinyon or juniper litter had significantly lower total sediment concentrations and yields than plots with herbaceous cover or bare plots (Bolton and Ward 1992).

Sediment movement is greatest in the interspace areas. Interspace areas contributed the most runoff and erosion on the litter plots studied by Bolton and Ward (1992). It is generally agreed that the characteristics of interspace areas have changed, especially since European settlement. While erosion has been attributed to the decline of herbaceous vegetation, there are other reasons for both factors. Wildlife and livestock concentrate on the herbaceous vegetation in interspace areas and use some areas as trails, contributing to reduced cover and increased soil compaction. Changes in the steepness of watershed channel gradients and slopes, because of erosion, have accelerated surface runoff rates in many areas, contributing to continued erosion and making reclamation more difficult. It is more difficult for litter layers to develop or seeds of herbaceous species to become established. Wilcox (1994) reported that interspace runoff and erosion vary spatially and temporally. They also vary with watershed size (large watersheds react differently than runoff plots), with fluctuations in soil moisture and soil infiltration capacity, and with degree of soil surface compaction throughout the year. Erosion from interspace areas with little bare soil was minimal but increased as the extent of bare ground increased. However, when the impacts of tree cover and interspace were integrated, as Wilcox (1994) also noted, sediment delivery was less from a small wooded watershed than from a non-wooded watershed (Heede 1987). Clary et al. (1974) concluded that there were no significant differences in sediment production on Beaver Creek between untreated watersheds and watersheds where trees had been removed.

It was anticipated that woodland control treatments would benefit wildlife because of increased forage. Numerous studies have analyzed the effectiveness and efficiency of such conversions, the relative success, and the responses of game and nongame wildlife. Expensive tools for improving range, chaining, and cabling only proved cost-effective in areas where posttreatment forage production potential was high (Short and McCulloch 1977). Goals of improving deer foraging habitat, if reached at all, were achieved only if more abundant and succulent spring forage resulted from the conversion and if converted tracts were small and interspersed within the woodland (Terrel and Spillett 1975). A study at Fort Bayard, New Mexico (Short et al. 1977), found that large clearings limited deer and elk use, because the animals would venture only a short distance away from cover. Because animal use declined as tree density increased, Short et al. (1977) recommended small clear-

ings interspersed within the stands for improved big game habitat. Simultaneous improvement of big game habitat and range appeared difficult because the small, interspersed nature of conversions that would benefit deer was contrary to the large, open tracts that would provide the greatest benefit to range (Short and McCulloch 1977).

Although additional spring forage can be beneficial to big game, the most valuable component of pinyon-juniper woodlands may be winter browse provided by shrubs and other woody vegetation typically removed by chaining and cabling (Terrel and Spillett 1975). The removal of the mid- and overstory by chaining or cabling also results in the loss of hiding and escape cover and important thermal cover for deer and other wildlife during the winter (Howard et al. 1987). Important questions to be answered are whether pinyon-juniper conversions truly improve foraging habitat for big game, and if so, is increased spring forage more important than the winter browse and thermal protection provided by partial overstory or intact pinyon-juniper habitats? Thus, the value of pinyon-juniper conversion as a tool in big game habitat management still remains a debated topic.

Knowledge of the effects of management practices in pinyon-juniper woodlands on wildlife and general ecosystem integrity is more important now with the greater emphasis on ecosystem management and multiple use. Tausch and Tueller (1995) determined that native plant species were best retained/augmented and mule deer winter use was higher in sites with a high species diversity and cover before chaining. Sites with little initial understory had a higher cover of introduced (seeded) species and cheatgrass (*Bromus tectorum*) and less deer use after chaining. Bird species diversity and use by the foliage/timber searching guild, aerial foraging guild, and hole nesting guild were lower on chained plots than unchained. However, species of the ground nesting and foraging guilds were not affected (Sedgewick and Ryder 1987). Causes for the declines in bird use on chained plots may have been related to changes in predominant vegetation type, amount and distribution of foliage, vegetation height, and canopy cover. Through these proximate factors, changes occur in factors that ultimately determine the presence or absence of the species such as food availability, microclimate, quantity and quality of nest sites, perch site availability, and protection from predators (Sedgewick and Ryder 1987).

The woodlands are considered a nutrient-poor ecosystem. Nutrients can be lost from the ecosystem by chaining and broadcast burning of slash. These activities in singleleaf pinyon (*P. monophylla*)-Utah juniper stands could result in a loss of approximately 13 percent of the total ecosystem nitrogen because of nitrogen volatilization (Tiedemann 1987). Assuming that 60 percent of the aboveground total nitrogen (about 855 kg per hectare) would be volatilized and a natural replenishment rate of between 1 and 2 kg per hectare per year, Tiedemann (1987) estimated that this lost nitrogen would be restored in 425 to 855 years. A study in Arizona (Perry 1993) found that 91 percent of the vegetative nitrogen was lost following prescribed burning of pinyon-juniper slash. Such large losses would result in lowered long-term productivity. Other nutrients such as phosphorus are also affected, especially if large amounts of litter are consumed in the burning (DeBano and Klopatek 1987, 1988; Perry 1993). In addition, burning also has a detrimental effect on soil microorganisms (Klopatek et al. 1990). However, soil nutrient levels may increase or remain constant in the top 5 cm following prescribed burning of slash that was not piled, because of oxidation of organic materials from the vegetative material (Perry 1993). More information concerning the effects of fire on pinyon-juniper soils is presented in Covington and DeBano (1990).

A benefit-cost analysis of tree control projects using data from throughout the Southwest demonstrated that the most successful projects only broke even (Clary et al. 1974). Fuelwood sales and potential losses in long-term site productivity were not included in these analyses.

Multiresource Management

Reevaluation of pinyon-juniper management strategies began in the 1970s, partially because of the increase in fuelwood demands resulting from the oil embargoes (Gottfried and Severson 1993). Ffolliott et al. (1979) found a 400 percent increase in wood usage in five Arizona markets between 1973 and 1978. The possibilities of sustained production of fuelwood and integrated resource management began to be considered, especially in mature woodland stands. The woodlands provide a full array of products including fuelwood, pinyon nuts, fence posts, Christmas trees, landscape trees, forage for livestock, habitat for common and rare and endangered wildlife species, and watershed protection. Demands for

these products and values continue to be strong or are increasing. The demand for fuelwood has fluctuated but continues to be high. Although current values are unavailable, approximately 227,000 m^3 of pinyon and juniper fuelwood were harvested in New Mexico in 1986 (McLain 1989). Some of the national forests that are located near population centers are concerned that demand will exceed supply within the next 50 years. The livestock industry is one important current and traditional use of the pinyon-juniper woodlands in the Middle Rio Grande Basin. Watershed protection, restoration, and site productivity are major concerns, especially in areas where past land uses have caused degraded conditions. There are issues concerning improving wildlife habitat for game species, nongame species, and threatened, endangered, and sensitive species. Most of the prehistoric archeological sites in the Southwest are concentrated in the pinyon-juniper zone because of the availability of resources and the moderate climate. Fifty-seven percent of 2,000 surveyed archeological sites at Bandelier National Monument are in juniper and pinyon-juniper woodlands. There also has been an increase in the use of woodlands for recreation and for second and primary home sites.

The public has also begun to recognize that the pinyon-juniper woodlands are connected to the culture and history of many rural and indigenous populations, and that their concerns must be integrated into land management plans. The woodlands have been used by American Indians since prehistoric times for construction timber, fuelwood, pinyon nuts, medicines, ceremonial items, and a place to hunt and gather food. The early European colonists and their descendants have used the woodlands for many of the same products. Most tribal and rural communities in the Southwest depend on fuelwood as the primary source of heating and cooking fuel, and also upon the commercial sale of fuelwood as a source of income. Birds, game, small mammals, and woodland predators are important for hunting, viewing, and traditional and religious purposes in American Indian cultures. Many aspects of the Zuni religion revolve around their relationship with the deer, which is hunted for traditional and recreational purposes. Zuni Indians also hunt game birds for meat and woodland songbirds for their feathers (Miller and Albert 1993).

Ecosystem management mandates the sustained productivity of the land while maintaining a diverse and healthy ecosystem. Recognizing the potential value of managing the woodlands for multiple products and benefits was the first step in the changing approach toward pinyon-juniper woodlands. Managers now have to develop prescriptions that meet their production goals and still produce or maintain productive and healthy stands. Naturally, sound prescriptions must account for the variability of habitat types and existing stand conditions. Management procedures also must be developed that will not damage archeological and historical sites within the woodlands. True ecosystem management should be based on an integrated planning effort that includes the inputs of managers representing the natural resource and related disciplines. Contributions of sociologists, community representatives, or marketing specialists could be useful, especially on private lands.

Not all sites have the potential to produce the full range of resource benefits, and this factor too must be evaluated. A classification for pinyon-juniper woodlands based on site productivity can aid management planning. The pinyon-juniper woodlands can be divided into high-site and low-site categories based on site productivity and stand volume (Conner et al. 1990; Van Hooser et al. 1993). Productivity is a measure of how well a site is able to sustain itself, and is determined by soil depth and texture, rockiness, slope, and presence of regeneration (Van Hooser et al. 1993). High-site lands produce wood products on a sustainable basis, while low-site lands include areas where volumes are too low to be included in calculations of allowable harvest levels. Some trees can be harvested from low-site lands, but they would only support a single harvest and the total volume harvested would be insignificant. Almost 86 percent of the pinyon-juniper and 80 percent of the juniper woodlands in New Mexico are in the high-site category (Van Hooser et al. 1993). High-site lands have the best potential for integrated resource management. The Bureau of Indian Affairs in Albuquerque tentatively defines commercial woodlands as those producing 0.4 m^3/ha of wood products annually.

The new ecosystem approach to pinyon-juniper management must be based on sound scientific information. However, our knowledge of the ecology of woodlands and impacts of management options is incomplete. The Rocky Mountain Forest and Range Experiment Station of the USDA Forest Service, personnel from land management agencies, and universities are attempting to fill gaps in our knowledge, but the emphasis on woodlands is relatively recent and many questions remain to be answered. The Rocky Mountain Station is currently conducting re-

search on tree regeneration ecology and silviculture; watershed management including soil erosion and site productivity; effects of stand treatments on wildlife habitat relationships; and tree mensuration (Gottfried 1992b).

Management of High-Site Woodlands

Silvicultural approaches

Silviculture provides the tools for manipulating the woodland tree cover to sustain production of wood products and maintain woodland health. One chief goal of silviculture is to obtain satisfactory tree regeneration for the future. Silviculture can also be used to improve forage production and wildlife habitat and to create an aesthetically pleasant landscape.

Managers from the USDA Forest Service, USDI Bureau of Indian Affairs, and other federal and state agencies are also attempting to develop prescriptions that would provide integrated resource management. Management prescriptions and objectives vary throughout the Southwest. Bassett (1987) reviewed the common silvicultural prescriptions and concluded that single-tree selection and two-step shelterwood methods are best for sustained stand health and productivity of woodlands. These methods are compatible with the dispersal patterns of heavy tree seed, provide protected micro-sites for regeneration, and are aesthetically acceptable. There are, however, some disadvantages, especially related to the costs associated with intensive management and potential damage to residual trees during initial and subsequent harvests.

Bassett (1987) presented a discussion of the tradeoffs that must be evaluated in preparing a prescription. A single-tree selection treatment designed to reduce stand density but still retain uneven-aged structure and horizontal and vertical diversity is being studied by the Rocky Mountain Station in cooperation with the Heber Ranger District of the Apache-Sitgreaves National Forests at Heber, Arizona (Gottfried 1992b). Single-tree selection prescriptions also are being evaluated by the USDI Bureau of Indian Affairs in western New Mexico and southern Colorado (Schwab 1993). The effects of different stand densities on pinyon nut production are being studied jointly by the Rocky Mountain Station and the Albuquerque Area Office of the Bureau of Indian Affairs. Group selection, which creates small openings within the stand, is less common and needs further study. Success from the forestry perspective would depend on achieving satisfactory regeneration from residual seedlings and seed, and from movement of seed into openings from the surrounding stand. Two-step and three-step shelterwood methods are being evaluated in New Mexico. A one-step shelterwood method can be used when advance regeneration is satisfactory.

The clearcut method and the seed-tree method generally result in unsatisfactory regeneration success because of poor seed dispersal. Small clearcuts can be appropriate when dwarf mistletoe control is necessary. Silvicultural prescriptions should be compatible with habitat type characteristics. Proper management for sustained production of the tree resources also requires additional growth and yield information related to site characteristics (Gottfried 1992a).

Wildlife-range approaches

Factors related to the proximity of suitable vegetation conditions and the availability of food, hiding and thermal cover, and nesting sites actually determine the distribution of most wildlife species, whether bird or mammal. Because of different species' habitat requirements, the same changes that degrade habitats for some species are likely to improve habitats for others. Although total numbers of small mammals increased in plots cleared by chaining or bulldozing in a New Mexico study, responses of individual species varied depending on habitat requirements (Severson 1986a). Woodrats and brush mice increased when slash was left, regardless of overstory. Pinyon mice and rock mice increased when slash was left and overstory was relatively intact but decreased if overstory was completely removed (Severson 1986a; Sedgewick and Ryder 1987). Kruse et al. (1979) also found that species preferring woodland habitats decreased in treated woodlands. Grassland species increased when the overstory and slash were completely removed but decreased when slash was left. Initial data indicate that fuelwood harvesting may negatively affect pinyon mice and positively affect deer mice populations and species diversity (Kruse 1995). Although grassland and other open woodland species may benefit from partial or entire removal of overstory, the amount of remaining mature woodland (and thus habitat for species that require a more dense canopy) should be considered.

Pinyon-juniper woodlands are used and manipulated for a variety of purposes (grazing, range im-

provement, fuelwood harvesting, recreation, wildlife habitat, etc.). A greater knowledge of species' habitat requirements and the effects of nonwildlife-related activities on wildlife resources may enable managers to select tools for achieving their goals that minimize detrimental effects or maximize beneficial effects on wildlife. For example, species that require midstory and understory plants may decline in mature pinyon-juniper habitats with dense canopies and thus may benefit from fuelwood harvests (Short and McCulloch 1977). In addition, the different hiding cover and overstory requirements of individual species suggest that slash disposal and degree of overstory removal are factors that may be manipulated to determine which species benefit from the treatment (Severson 1986a).

Current management is integrating livestock and wildlife with tree product objectives. A careful assessment of wildlife and other needs must be made to ensure tradeoffs in resource allocation are acceptable. Clearing small dispersed areas of trees benefits elk, mule deer, and livestock (Short et al. 1977). Openings create a more diverse landscape that favor many wildlife species. For example, small mammal populations may increase within cleared areas (Severson 1986a) and thus may attract more predatory birds and mammals. Birds that feed on insects associated with openings should also benefit from this landscape. However, openings should not be too large (Severson and Medina 1983) and the woodlands should not become fragmented. In many cases, the actual size of the openings may not be critical if continuous corridors of adequate width are maintained. Stands surrounding openings can remain untreated or may be partially harvested. Forage production is also stimulated in areas harvested using an overstory removal cut and in group selection openings.

Managers must decide if cleared wildlife-livestock areas should be maintained or if trees should be allowed to reoccupy the sites. If trees are allowed to reoccupy the openings, a management scheme could be created that involves a variety of seral stands. There is a need to define spatial and temporal patterns by habitat type that maximize plant and animal diversity. Springfield (1976), Severson and Medina (1983), and Short and McCulloch (1977) present reviews of range management and wildlife management within the woodlands.

Treatments that reduce tree densities, such as the single-tree selection and shelterwood method, should benefit livestock and native ungulates by providing additional forage while maintaining some degree of thermal and hiding cover. Increased herbaceous cover will also help stabilize the soils on some sites. However, the impacts of residual trees on understory dynamics is unclear. Large reductions in tree canopy cover are necessary to improve total herbage yields (Arnold et al. 1964; Pieper 1990). However, while total herbage biomass and blue grama biomass decline with increased canopy cover, the biomass of cool-season grasses such as pinyon-ricegrass (*Piptochaetium fimbriatum*) and New Mexico muhly (*Muhlenbergia pauciflora*) actually increase with increased tree cover (Pieper 1990). Further research relating herbage production to stand density is being planned.

Many high-site areas treated during pinyon-juniper control programs have been reoccupied by healthy stands of trees over the past 30 to 40 years, often the result of advance regeneration that survived the initial treatment. If regeneration is vigorous and dense enough to result in a healthy tree stand, the area should not be treated again because successful regeneration of large openings is difficult. The young stands would be part of the diverse landscape required by many wildlife species. Increased herbage production would occur until the tree canopy closes.

Slash disposal

Slash disposal after harvesting or vegetation type conversion is another important issue in woodland management. Slash disposal may vary according to management objectives (Severson and Medina 1983). On any one management area, several slash treatments may be warranted and practical. It is generally accepted, based on work in the Great Basin (Tiedemann 1987), that burning slash in large piles is unacceptable because of the adverse effects on soils and overall site productivity. However, slash in small piles may be burned with the intent of creating areas containing earlier seral stages that increase floristic species richness on the treatment area. Other piles could be left unburned to provide habitat for small mammals (Severson 1986a). Slash piles can break up sight distances and provide security cover for wild ungulates. Slash can be scattered in some areas to provide protection for herbaceous growth and to provide nursery sites for young trees. On other sites, scattered slash could be burned in a cool fire to promote temporary increases in nutrient contents of the herbaceous forage components.

Slash also provides some erosion protection by retarding surface water movement and serving as a

place where sediment can accumulate and not be lost from the site. A study of several slash disposal treatments showed that a slash-scattered treatment resulted in the least surface runoff and sediment loss and in relatively higher soil moistures (Wood and Javed 1992). This treatment also had the best vegetation response, which helps protect the soil, slows runoff and erosion, and increases infiltration. Slash burning and complete slash removal had the reverse effect. Slash can also be placed into small gullies to reduce erosion.

Slash is often collected by older members of rural communities or American Indian tribes for cooking and heating purposes. It is an inexpensive and easily obtained resource. It also provides a useful activity for younger members of the community. There have been conflicts between these communities and land management organizations when slash is burned.

Single Resource Emphasis

While the wisdom of woodland control for production of forage for livestock on high-site public lands is questionable from economic and ecological perspectives, some private owners may still prefer this option. Some benefits of multiresource management can still be achieved. Fuelwood and other wood products should be harvested rather than leaving downed trees on site. This harvest provides a cash return and makes subsequent activities easier and more economical. One approach, even when livestock production is the main objective, is to create mosaics of tree-covered areas interspersed with grass-forb-dominated areas. Such a pattern should favor a mixture of cool-season and warm-season grasses (Pieper 1990). A mosaic landscape is beneficial for wildlife and livestock and is aesthetically pleasing.

Another approach is to create savannas by retaining some of the larger pinyon and juniper trees from the original stand. Such savannas can be more aesthetically pleasing than large openings and still provide some limited wildlife habitat benefits and shade for livestock. Although large savannas have drawbacks, this treatment should be integrated into a landscape that includes untreated and lightly treated stands and small openings.

Management of Low-Site Woodlands

Tree control to enhance forage production is easier to justify on low-site lands where management for tree products is not economically or biologically feasible. Forage production should be stimulated by normal range management activities, with the actual techniques depending on equipment demands and site characteristics. Stands of low stature and density could be the result of arid conditions that would affect the quantity, quality, and rate of replacement of grasses and forbs. Site factors, such as soil physical and chemical characteristics and annual precipitation, have to govern the appropriateness of treatments and the selection of forage species.

Even when tree control is desired, managers should consider the size and placement of openings and consider prescriptions recommended for high-site lands. Large openings are detrimental to deer and elk and to many nongame species. Mosaics of trees interspersed with cleared areas create a more acceptable landscape. The covered areas provide hiding and thermal cover for both wildlife and livestock. It is common to find cattle concentrating in pockets of residual trees within chained or cabled areas.

Regardless of the treatment and objectives on high- and low-site lands, proper grazing management is an important key to successful range improvement activities. Although data are scarce, there is a general belief that the poor response of native and introduced forage species to tree control activities can be related to poor livestock management. Animals were often allowed onto areas before the plants had become established. Some grazing deferral for at least two grazing seasons probably is necessary although the amount must be governed by site, climate, and forage species.

RESEARCH NEEDS

Numerous gaps exist in our knowledge of the ecology and management of the pinyon-juniper woodlands of New Mexico and the Southwest. Based on experiences, responsibilities, and needs, land managers and scientists have identified high priority topics that should be studied. Obviously, the research topics and their priorities vary. A list of recommended research and management activities was included as part of the proceedings from the 1993 Santa Fe pinyon-juniper symposium (Aldon and Shaw 1993). A subsequent meeting was held at the New Mexico State Land Department Office in Santa Fe to develop an action plan for managing New Mexico's woodlands for sustainability and social needs (Aldon and Shaw 1993). Achieving most of the goals in the plan would require additional basic and applied research.

Landscape and plant ecological studies provide the basis for high priority research opportunities (table 1). Ecological research in these areas is applicable to all aspects of ecosystem management. For example, stand dynamics information and predictive models would be needed for effective tree product and wildlife management, and for efforts to maintain and enhance biological diversity. Currently, many assumptions are being made about historic tree densities and encroachment of grasslands without the benefit of accurate historical information. Demographic studies would provide important information about changes in the woodlands related to climatic fluctuations, natural and human perturbations, and general land-use histories.

Enhanced biodiversity is central to ecosystem management. There is a need to develop agreements on desired conditions for the different pinyon-juniper ecosystems, particularly an understanding of the structural, functional, and spatial arrangements over time and space that will allow management for sustainability. How do management and natural processes affect plant and animal composition and density? How do they affect ecosystem stability? Priority studies would investigate the relationships between faunal and floral structure and community composition of different types of woodland stands. There is also a need to understand the major influences, processes, and relationships between trophic levels, for example, how mammals, birds, and insects influence cone production, seed dispersal, and regeneration, nutrient cycling, and food webs. An understanding of soil ecology is basic to the health of the entire ecosystem. The effects of land use on soil resources has often been ignored but has implications for the sustainability of all ecosystem components. Landscape fragmentation is a concern. How does fragmentation limit the efficiency of ecosystem processes and effectiveness of ecosystem functioning and, consequently, impact sustainability?

Woodland managers need further research on pinyon and juniper diseases and insects and their impacts on regeneration, growth, yields, and mortality. The effects of pinyon dwarf mistletoe and of flower and cone insects are of particular concern. There is a need to develop and evaluate various silvicultural prescriptions for promoting various management options, including enhanced understory composition and density. Growth and yield information, related to habitat type, is vital to sound management. Concerns about pinyon-juniper watershed conditions are especially important in New Mexico; effective and efficient methods are needed to restore watershed stability without compromising the integrity of woodland ecosystems.

Fire history, ecological impacts, and management require additional attention. Fire or the lack of natural fire, as has been noted, was an important factor in shaping the composition and structure of presettlement and present stand conditions. Misuse of fire, for example in some slash disposal activities, has often damaged site productivity. Managers are beginning to consider methods of reintroducing fire to the ecosystem but more information is needed. There are very few fire histories from the pinyon-juniper woodlands and only a few studies concerned with fire effects on soil resources.

Many concerns about management and the preservation of cultural resources exist. While some research has been conducted on the effects of fire on cultural resources in the Southwest (Knight 1994; Lent et al., in press; Lissoway and Propper 1990; Traylor et al. 1990), much of this work has been conducted in ponderosa pine forests, the work of Switzer (1974) and Eininger (1990) being exceptions. Given the inclination of many resource managers to increase the role of prescribed fire in these woodlands, it would be desirable to have better information on fire effects on cultural resources in woodland areas. The increase in the number of unauthorized roads and off-road vehicle use have implications for cultural, hydrological, and biological resources in the Southwest. More sociological and educational research is needed on ways to deter the public, as well as commercial pot-hunters, from degrading our heritage resources. Little is known about the effects of other ecological processes in pinyon-juniper woodlands on the integrity of cultural resources. The recent Bandelier Archeological Survey recorded extensive impacts to archeological sites from such phenomena as trees and large cholla cacti growing and tipping over in sites; animal burrows; trampling impacts from large mammals like elk; feral burro wallows; and erosion. About 76 percent of approximately 1,500 surveyed archeological sites were affected by one or more types of erosion, particularly displacement of surface artifact assemblages by sheet erosion (Head 1992). Increasing elk numbers in the Southwest has raised concerns that elk may impact archeological sites through trampling (much like cattle) and by facilitating erosion problems through grazing pressure on the limited herbaceous ground cover. More re-

Table 1.—Landscape and plant ecological research needs for the Middle Rio Grande Basin.

	Research topic	Justification	Examples
1	Spatial analysis of site and stand characteristics at the regional scale using remote sensing and GIS.	Provide a mapped data and information system for modeling efforts, regionalization of local research products, and proposed management actions.	Milne (in progress) , University of New Mexico
2	Regional-scale biogeographic and biogeochemical models, integrated with models of stand dynamics.	These models provide useful tools for organizing ecosystem research, for investigating factors that influence ecosystems at various scales, and for forecasting ecosystem response to climate and land use at the regional scale.	MAPPS: Neilson and Marks 1994; TEM: McGuire et al. 1992; Rastetter et al. 1991; Forest-BGC: Running and Coughlan 1988; CENTURY: Parton et al. 1988
3	Stand dynamics models.	Simulation of forest successional and ecosystem process at the local scale; gap models can simulate tree establishment, growth, and mortality. These models can be used to assess sustainability of fuel harvesting of both dead or live wood, or to evaluate the impact of tree removal on stand structure. Models can be used to project changes in local and regional stand structure due to land use and climate. Simulations can be tested against historical reconstructions (see 4).	FORSKA-2: Prentice et al. 1993; HYBRID: Friend et al. 1993; SILVA: Kercher and Axelrod 1984; CLIMACS: Dale and Hemstrom 1984; ZELIG, Urban et al. 1993; FORMAN I: Samuels and Betancourt 1982
4	Historical studies of pinyon-juniper woodlands, particularly the last 2,000 years; this includes the use of archival information on land use, historical ground and aerial photography, long-term demographic disturbances, climatic reconstructions from tree rings, and packrat midden studies to determine ecotonal shifts.	Provide long-term data about baseline (background) conditions and natural variability in the more sensitive parts of the ecosystem (p-j/grassland or p-j/ponderosa ecotones); description of the range of possible trajectories. Better understanding of how climate, disturbance (e.g., fire) and land use maintain or shift ecotonal boundaries. Gage the regional and long-term impact of catastrophic droughts as well as wet episodes on ecotonal boundaries, age structure, and species composition.	Betancourt et al. 1993; work in progress through collaboration by Forest Service, USGS-Desert Laboratory, and University of Arizona-Tree Ring Laboratory (in progress)
5	Reconstruction and long-term monitoring of pinyon seed crops (masting) at the regional scale using cone-scar assays, tree-ring data, and annual field surveys.	Masting influences pinyon age structures, as well as population cycles of seed predators. Pinyon nuts are an important cash crop in New Mexico. Climatic influences on geographic and temporal patterns in bumper crops cannot be determined without reconstructing and monitoring seed crops at the regional scale.	Forcella (1981); Floyd (1987); Little (1940)
6	Regional seedling surveys carried out every two years for trend detection; first-year effort will be inclusive, subsequent years will be less intensive.	Detect trends in regeneration of pinyon-juniper woodlands relative to climate, land use, and site characteristics. Resolve controversies about trends in regeneration relative to fire suppression and grazing. Address if the wet episode of the 1980s and 1990s has produced a surge in pinyon and juniper recruitment.	None
7	Regional inventories of woody debris and fuel harvesting; quantify woody debris in undisturbed vs. disturbed stands; quantify regional availability and exploitation of wood debris as fuel.	Woody debris inhibits soil erosion, provides microsites for germination/establishment, offers shelter for fauna, and serve as storage and recycling points for nutrients. Fuel gathering is an important economic activity in some rural communities; it tends to concentrate near population centers. There is currently no regional assessment of fuel harvesting or of its ecological consequences.	Ernest et al. (1993); DeBano et al. (1987); Barth (1980)
8	Regional study of successional processes at woodland conversion sites along environmental gradients.	The long history of conversions in the 1950s and 1960s provides a useful framework to study changes in succession of both herbaceous and perennial plant cover and in nutrient distribution 30 to 40 years after wholesale removal of trees. Trees in p-j accumulate nutrients beneath their canopies; what were the long-term effects of different tree removal techniques on distribution and loss/gain of nutrients?	Rippel et al. (1983)
9	Water and sediment budgets at watershed and basin scales; develop models of water balance and sediment yield, storage, and transport.	Linkage with biogeographic and biogeochemical models; estimate erosion losses during 20th century.	Lane and Barnes (1987); Dortignac (1960); Hawkins (1960); Renard 1987; Milne (in progress), University of New Mexico

search on these topics would be useful to help resource managers assess how to best maintain the cultural resources in their care.

There is consensus among southwestern archeologists that a key need is for better information on interactions between prehistoric and historic human societies and woodland environments. Such information would also help define sustainability goals for management of these ecosystems. A focal issue is to ascertain the magnitude and causes of changes in woodland ecosystem structure and function in the Middle Rio Grande Basin during periods of human occupation, and to determine how those changes in turn affected the societies that were contributing to them (Samuels and Betancourt 1982). Toward this end, better paleoenvironmental information is needed, especially data on vegetation change, trends in soil erosion and development, and climatic reconstructions. Integrated geomorphological and archeological investigations relating patterns of soil erosion to human land use practices would be valuable, given the widespread perception that current high erosion rates are unsustainable and largely triggered by modern human agency.

There is a need for socioeconomic studies within the pinyon-juniper woodlands of the Middle Rio Grande Basin. Managers must be attuned to the needs and views of society. Sociological studies concerned with conflict and policy issue resolution techniques would be desirable. Analyses of policies that relate to the sustainability of natural resources have been the focus of similar studies throughout the world, but unfortunately, not in relation to the pinyon-juniper woodlands. A study is needed to assess current policies in relation to technical issues. Technical problems often are related to the lack of effective policies or the lack of proper implementation and compliance with these policies. Frequently, technical problems can be solved once suitable policies are available. There is a need for better, and perhaps more formal, communication and coordination among the land management and the research communities concerned with pinyon-juniper woodlands. Public education about the aesthetic and commercial values of the woodlands would enhance public support and understanding for ecosystem management. Economic considerations may not be important in setting management goals on sensitive or fragile sites but they should be evaluated in the preparation of most multiresource, ecosystem management decisions. Economic analyses of common prescriptions would increase the information needed for management decisions.

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