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Changes in the landscape of the Jemez Mountains, New Mexico

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University of California, Berkeley, 1989

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Changes in the Landscape of the Jemez Mountains, New Mexico

By

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Finally, I dedicate this work to the source of my inspiration and my ongoing teacher, the landscape of the Jemez Mountains. May the Jemez Mountains ever be a landscape of hope and wonder for those who inhabit this fascinating portion of our world.

CHAPTER I. INTRODUCTION

> The mountain ranges skirting the Rio Grande del Norte on the west, nearly opposite the town of Santa Fe, in the territory of New Mexico, are today but little known. The interior of the chain, the Sierra de los Valles, is as yet imperfectly explored. Still, these bald-crested mountains, dark and forbidding as they appear from a distance, conceal and shelter in their deep gorges and clefts many a spot of great natural beauty, surprisingly picturesque, but difficult of access (Bandelier 1890).

The Jemez Mountains (Sierra de los Valles) rise as a large volcanic landmass at the southern edge of the Rocky Mountains in north-central New Mexico (Figure 1-1). While better explored and more accessible than in Adolph Bandelier's day, the Jemez Mountains still hold many ecological mysteries.

Past ecological research and natural resource management approaches here have focused on ecosystem components or isolated fragments of the overall Jemez Mountains landscape. This study presents a landscape-level framework for considering the ecology and management of the Jemez Mountains, based upon the following propositions:

1) the Jemez Mountains can logically and usefully be treated as a single, ecological landscape;
2) the landscape of the Jemez Mountains has undergone extensive, but little-recognized, structural and functional changes in historic times due to the interactions of human and natural processes; and
3) consideration of a landscape management perspective and recognition of widespread landscape change have significant implications for local land management agencies.

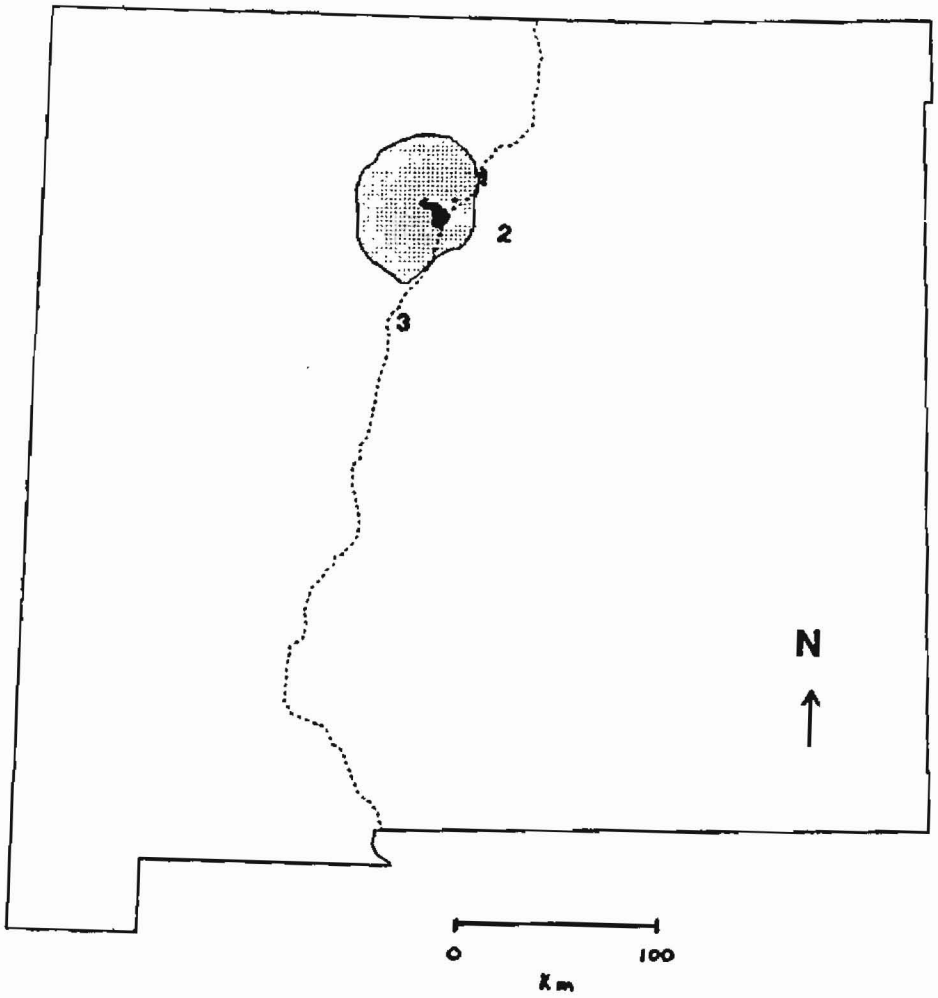


Figure 1-1. The location of the Jemez Mountains (patterned patch), Bandelier National Monument (solid black patches), and the cities of Española (1), Santa Fe (2), and Albuquerque (3) in northern New Mexico. The dashed line represents the Rio Grande.

These propositions are briefly justified below.

Relatively homogenous local ecosystems (or communities, sensu Lidicker 1988) can be distinguished within the Jemez Mountains based on site-specific vegetation, landforms, and soils (Rowe and Sheard 1981, Barnes et al 1982, Driscoll et al 1984). When spatially aggregated, the local ecosystems of the Jemez Mountains form a repeated mosaic pattern that may be considered to compose a single landscape (Miller 1978, Forman and Godron 1981, Bailey 1985). Physical isolation of the island-like Jemez Mountains by surrounding lowlands and a common geologic origin provide logical boundaries for this landscape. Bandelier National Monument (BNM) comprises a complete altitudinal transect on the southeast flank of the Jemez Mountains, including examples of most of the ecosystems present in this landscape.

Anthropogenic landscape change is obviously occurring today as the Jemez Mountains are subject to an increasing variety and intensity of human activities, with uncertain cumulative impacts on individual species, ecosystems, and the landscape as a whole (Loucks 1985). Currently prominent are impacts from logging, grazing, recreational activities, alien species introductions, air and water pollution, fire suppression, highways, reservoirs, power-line corridors, and building construction (BNM 1989). Less obvious natural changes are also occurring. Still, many local observers and resource managers implicitly or explicitly consider most modern ecological patterns to be largely natural in origin and rather static. Various lines of evidence indicate that this landscape has undergone much unrecognized historic change; this study will illustrate some of that evidence for a dynamic landscape.

Recent works in conservation biology (Noss and Harris 1986, Morse et al 1986) and landscape ecology (Risser et al 1984, Forman and Godron 1986, Franklin and Forman 1987) illustrate that effective management of local ecosystems requires consideration of the landscape context in which they are imbedded. Land management agencies that are mandated to maintain natural features (e.g., biodiversity) and processes (e.g., fire regimes), such as the U.S. Forest Service and the National Park Service, are coming to realize that their management practices need to reflect a landscape perspective that crosses agency boundaries (Agee and Johnson 1988, NPCA Commission on Research and Resource Management Policy 1989, USDI National Park Service 1988). Recognition that landscape change calls for management action is not new (e.g. Leopold et al 1963, Stone 1965), but an increased awareness of and dialogue on the management implications of widespread landscape change has occurred recently (White and Bratton 1980, Gruell 1984, Bonnicksen and Stone 1985, Parsons et al 1986, Christensen et al 1986, Chase 1987, Bonnicksen 1988).

The unifying theme of this study is a consideration of the landscape ecology of the Jemez Mountains in and around Bandelier National Monument. The specific objectives of this study are to:

1) document current landscape and vegetation patterns;

2) explain current landscape and vegetation patterns;

3) identify historic landscape and vegetation changes;

4) explain historic landscape and vegetation changes in terms of human land use practices and natural processes; and

5) discuss the implications of the above for land management in the area, particularly for Bandelier National Monument.

CHAPTER II. BACKGROUND AND RELATED RESEARCH

STUDY AREA

GEOLOGY, LANDFORMS, AND SOILS

The Jemez Mountains are located in north-central New Mexico (Figure 1-1). They range in elevation from 1590 m at the Rio Grande to 3,526 m at the summit of Tschicoma Peak, with a geologic boundary enclosing about 543,522 ha (Smith et al 1976). The Jemez Mountains are the remnants of a large, collapsed volcano that underwent massive eruptions 1.4 and 1.1 million years ago. Prominent landforms include two central calderas, secondary domes within the calderas, the mountainous remnants of the pre-collapse volcanic pile which rim the calderas, an encircling skirt of canyon-dissected tuff plateaus, White Rock Canyon, and the basaltic cones of the Cerros del Rio (Burton 1982). The central Toledo and Valles calderas are 12 and 24 km in diameter. The caldera rim peaks are known as the Sierra de los Valles. The Pajarito Plateau, on the east flank of the Jemez Mountains, is composed of up to 300 m of consolidated ash tuff that was deposited during the big eruptions. The townsite of Los Alamos is situated on the upper end of the Pajarito Plateau, at the base of the Sierra de los Valles. The Rio Grande flows through White Rock Canyon, a 300 m deep gorge that separates the Pajarito Plateau from the Cerros del Rio to the southeast. An immense amount of geological research has been conducted in the Jemez Mountains (c.f. Mills 1987). The locations of some landforms and other place names are mapped in Figure 2-1.

Common soil parent materials in the Jemez Mountains range from rhyolites and andesites, with some dacites and latites, at high elevations, to tuff and pumice on the plateaus and basalts near the Rio

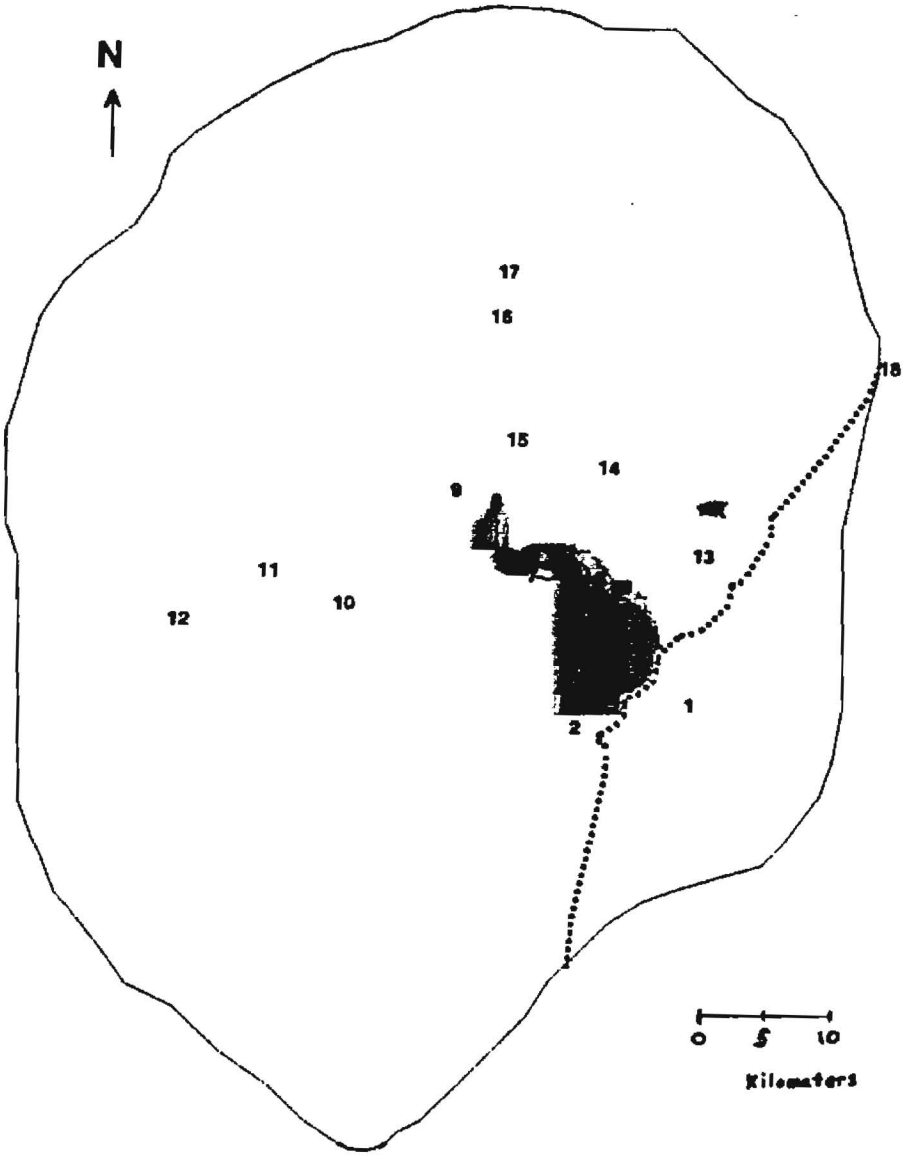


Figure 2-1. Place name locations in the Jemez Mountains. Cerros del Rio (1), Canada de Cochiti Grant (2), Frijoles Mesa (3), Burnt Mesa (4), Escobas Mesa (5), Apache Mesa (6), Mesa del Rito (7), Cerro Grande (8), Valle Grande (9), Cerro Pelado (10), Monument Canyon Research Natural Area (11), Jemez Springs (12), White Rock (13), Los Alamos (14), Canada Bonito (15), Chicoma Ridge (16), Polvadera Peak (17), and Espanola (18). Shaded patches are Bandelier National Monument, and the dotted line is the Rio Grande.

Grande (Nyhan et al 1978). Patches of pumiceous soils are particularly prominent in a band roughly centered on the Frijoles watershed, corresponding to the axis of deposition from the El Cajete eruptions (J. Hawley, New Mexico Bureau of Mines and Mineral Resources geologist - personal communication). Soil orders found here include Entisols, Inceptisols, Alfisols, Mollisols, and Aridisols (Nyhan et al 1978). Soil survey work in the Jemez Mountains includes Nyan et al 1978, USDA Soil Conservation Service (in review), USDA Forest Service (in progress), and Earth Environmental Consultants Inc. (1974, 1978).

CLIMATE

Overall the Jemez Mountains experience a semi-arid continental mountain climate (U.S. DOE 1979), but this designation masks a great deal of variability associated with elevational gradients and topography. For example, annual precipitation ranges from 30 cm at the lowest elevations to about 90 cm at the caldera rim; mean annual precipitation at Bandelier's weather station (1990 m elevation) is 40.7 cm (BNM - records on file). There is usually a dry period from late April through the end of June, terminated by the onset of the summer "monsoon". Sixty percent of the annual precipitation falls between June and September, with thunderstorms reported for 58% of the days in July and August (U.S. DOE 1979). These convectional thundershowers, frequently accompanied by hail, bring 40% of the total annual precipitation in July and August. Cyclonic storms in winter bring snow to all elevations. Los Alamos has a 5-month growing season (May 6-October 16), with perhaps a 100-day growing season found at the highest elevation or in canyon bottoms with cold air drainage. July is the

warmest month at Los Alamos (mean temperature = 28° C), and January the coldest month (mean temperature = -1.6° C).

Local climate is also temporally variable, with wide fluctuations in annual precipitation common. Cyclic El Niño climate events bring increased spring and summer precipitation to this area about every four years (Andrade and Sellers 1988). Dendroclimatological work documents irregular occurrences of dry and wet periods extending back to 598 A.D. in the Jemez Mountains (Dean and Robinson 1977). Weather records at Los Alamos and Bandelier begin in 1911 and 1925, respectively, with data missing from some early years (Los Alamos National Laboratory and BNM - records on file).

VEGETATION

The Jemez Mountains are in the southernmost extension of Bailey's (1980) Rocky Mountain Forest Province, with vegetation communities similar to those found throughout the southern Rocky Mountains. The general conceptualization of the vegetation pattern in this area is a zonation of communities based on elevation and slope exposure. Upward along the 1900 m elevational gradient from the Rio Grande to the Jemez peaks one passes through juniper grasslands (Juniperus monosperma, Bouteluoa sp.) from about 1600-1900 m; piñon-juniper woodlands (Pinus edulis) at 1900-2100 m; ponderosa pine forests (Pinus ponderosa) at 2100-2300 m; mixed conifer forests of ponderosa pine, Douglas-fir (Pseudotsuga menziesii), white fir (Abies concolor), aspen (Populus tremuloides), and limber pine (Pinus flexilis) at 2300-2900 m; and finally into spruce-fir forests of Engelmann spruce (Picea engelmanni) and corkbark fir (Abies lasiocarpa var. arizonica) on the north slopes

of the highest peaks above 2900 m. High elevation grasslands (Festuca thurberi, Danthonia parryi) occur as large breaks in the mixed conifer forests on upper south-facing slopes (Allen 1984-a), and large moist meadows occupy the caldera "Valle" basins. A similar pattern of vegetation zones is used by several authors (Osborne 1966, Nyhan et al 1978, U.S. DOE 1979, U.S. DOE 1980). While other authors explicitly recognize more categories (e.g. Moir and Ludwig 1979, Potter and Foxx 1981, USDA Forest Service 1987-a), their specific vegetation types are generally consistent with this broad zonal framework. This study recognizes that the current vegetation of the Jemez Mountains reflects the underlying diversity of landforms, soils, climate, and site histories present in this landscape, and treats local vegetation in more detail below.

Several previous vegetation studies merit mention here. Koehler (1974) and Potter and Foxx (1981) mapped vegetation types in Bandelier; Cully (1986) integrated these two maps. Barnes (1983) recognized the existence of three habitat types within the piñon-juniper portions of the Pajarito Plateau. Gosz and Mark (1974), Tierney (1977), Oppenheimer (1979), Foxx (1983), Foxx and Potter (1978), Potter (1981), Potter and Foxx (1979-a, 1979-b, 1979-c, 1986), Potter and Tierney (1985), Potter et al (1982), Williams (1984), Allen (1984-a), Hink and Ohmart (1984), Dick-Peddie et al (1984), and Tierney and Potter (1985) have conducted quantitative vegetation studies in limited areas of the eastern Jemez Mountains. The flora of portions of the Jemez Mountains has been treated by Osborn (1966), Robertson (1968), Jones (1979), Foxx and Tierney (1985), Jacobs and Jacobs (1988), and Jacobs (1989).

The vascular plant flora of Bandelier National Monument includes

collections of 720 species in 347 genera representing 86 families (Jacobs 1989). Rare and endangered plant species found locally include the yellow ladyslipper (Cypripedium calceolus), cut-leaved grape fern (Botrychium multifidum), and grama grass cactus (Pediocactus papyracanthus) (Jacobs and Jacobs 1988).

FAUNA

The Jemez Mountains harbor a diversity of animal species and communities, reflecting the wide variety of available habitats in this landscape. Bandelier has inventory information for most faunal species. Recent park surveys indicate the presence of approximately 1000 arthropod species (including 10 likely new to science) (Pippin and Pippin 1984), 5 amphibians and 14 reptiles (Degenhardt 1975, Fleisher 1978), and 44 terrestrial mammals and 12 bats (Guthrie and Large 1980). About 115 breeding birds (Travis, in preparation) and 90 species of ants (Mackay et al 1988) have been recorded in adjacent Los Alamos County. The endemic Jemez Mountains salamander (Plethodon neomexicana) merits specific mention, as this state-listed endangered species is currently a Class I federal notice-of-review species (C. Painter, New Mexico Department of Game and Fish - personal communication). Other threatened or endangered species found in the Jemez Mountains include the bald eagle (Haliaeetus leucocephalus), peregrine falcon (Falco peregrinus), spotted owl (Strix occidentalis), and meadow jumping mouse (Zapus hudsonius).

Noteworthy local faunal ecology research includes the work of: T. Johnson (1986, 1988) on peregrine falcons and bald eagles; J. Johnson (1986, 1989) and Johnson and Johnson (1988-a, 1988-b) on spotted owls;

Kennedy (1985) on several other raptors; Arganbright (1988) on bats; Flavill and Whitford (1979) on the biota of Redondo Creek; Eberhardt and White (1979), Conley et al (1979), White (1981), and Rowland et al (1983) on elk (Cervus elaphus) and mule deer (Odocoileus hemionus); Koehler (1974), Morghart and Ohmart (1976), and Morgart (1978) on feral burros; and Ramotnik (1985, 1986) on Jemez Mountains salamanders.

PALEOECOLOGY

While much paleoecological research has been conducted in the American Southwest, the Jemez Mountains have received relatively little attention. During Late Wisconsin glacial times (24,000 - 10,000 years BP) the Southwest was characterized by increased effective moisture, peak alpine glacier development, the filling of large playa lakes, depressed treelines, and the widespread presence of woodlands and forests across intermountain lowlands (Van Devender and Spaulding 1979, Spaulding et al 1983, Spaulding 1984, Betancourt 1987). Vegetation associations were often anomalous relative to modern communities (Spaulding et al 1983), due to the individualistic nature of plant species response to different environmental conditions (Gleason 1926, Delcourt et al 1983). The absence of glacial landforms in the Jemez Mountains (personal observation) indicates that alpine glaciers did not develop here, although the adjacent Sangre de Cristo Mountains, only 600 m higher, display much evidence of Wisconsin glaciation. The observed abundance of felsenmeers in the Jemez Mountains (Tierney and Potter 1985) may reflect near-glacial conditions in Wisconsin times.

With the transition to the Holocene (ca. 10,000 years BP) modern vegetation associations developed in montane areas of the Southwest

(Spaulding et al 1983), although lowland woodlands persisted until the present climate and vegetation regimes established after about 8000 years BP (Van Devender and Spaulding 1979). Relatively stable vegetation associations apparently developed rapidly after the major early Holocene climatic changes in the Southwest (Van Devender and Spaulding 1979), in contrast to the continuing Holocene migrations of eastern North American tree taxa (Davis 1976, Webb 1987). A pollen core taken from a bog in the center of the Jemez Mountains reveals modern taxa present throughout the past 4600 years that this core documents, indicating that climate has been "fairly stable" over this time period (Stearns 1981). Climatic reconstructions in the Southwest, including sites in and around the Jemez Mountains, indicate that recurring episodes of dry and wet conditions characterize the Southwest over the past 2000 years (Dean and Robinson 1977, Euler et al 1979, Rose et al 1981, Petersen 1988). Ongoing research by Swetnam (1989), and proposed research by Betancourt and Turner (1988) and Kohler et al (1988) may shed more light on Jemez Mountains paleoecology.

LAND OWNERSHIP

Most land in the Jemez Mountains is currently administered by the Santa Fe National Forest (SFNF). Other important lands include: Bandelier National Monument; Los Alamos National Laboratory (LANL); the townsites of Los Alamos, White Rock, Cochiti Lake (on Cochiti Pueblo land), Jemez Springs, Coyote, Abiquiu, and Española; the reservations of San Juan, Santa Clara, San Ildefonso, Cochiti, San Felipe, Santo Domingo, Zia, and Jemez pueblos; U.S. Bureau of Land Management (BLM) tracts; and the private Baca Location #1, originally

TABLE 2-1. Land ownership across 187,858 ha of the Jemez Mountains.

Land Ownership Type	Area (ha)	% of map area
Santa Fe National Forest (SFNF)	78,848.8	41.97
Private (PRIV)	48,241.6	25.68
Pueblo (PUEB)	27,982.8	14.90
Bandelier National Monument (BNM)	13,306.6	7.09
Los Alamos National Laboratory (LANL)	11,281.5	6.00
Bureau of Land Management (BLM)	6,914.2	3.68
State of New Mexico	766.4	0.41
General Services Administration	329.2	0.18
Los Alamos County	26.0	0.01

a Mexican land grant, centered on and containing the core region of the Valles Caldera.

Current land ownership over a 187,858 ha area around Bandelier is displayed in Figure 2-2, with associated land areas in Table 2-1. Mapped pueblo lands include a slice of Jemez Pueblo and the triangular tip of Santo Domingo Pueblo in the southwestern quarter of Figure 2-2, along with portions of Cochiti, San Ildefonso, and Santa Clara pueblos in the south-central, eastern, and northern reaches of the map. The private land category is dominated by the large square of the Baca Location in the northwest quarter of Figure 2-2, although it also contains the Los Alamos and White Rock townsites adjacent to LANL on the north and east, as well as numerous inholdings within the Santa Fe National Forest. Some Los Alamos County lands are also included in the private category in the Los Alamos and White Rock townsites. The

Figure 2-2. Map of land ownership across 187,858 ha of the Jemez Mountains. Bandelier National Monument (BNM) = red, Santa Fe National Forest (SFNF) = green, private (PRIV) = yellow, Los Alamos National Laboratory (LANL) = light blue, pueblo (PUEB) lands = dark blue, Bureau of Land Management (BLM) = magneta, state of New Mexico = gray pattern, General Services Administration = black, and Los Alamos County = orange.

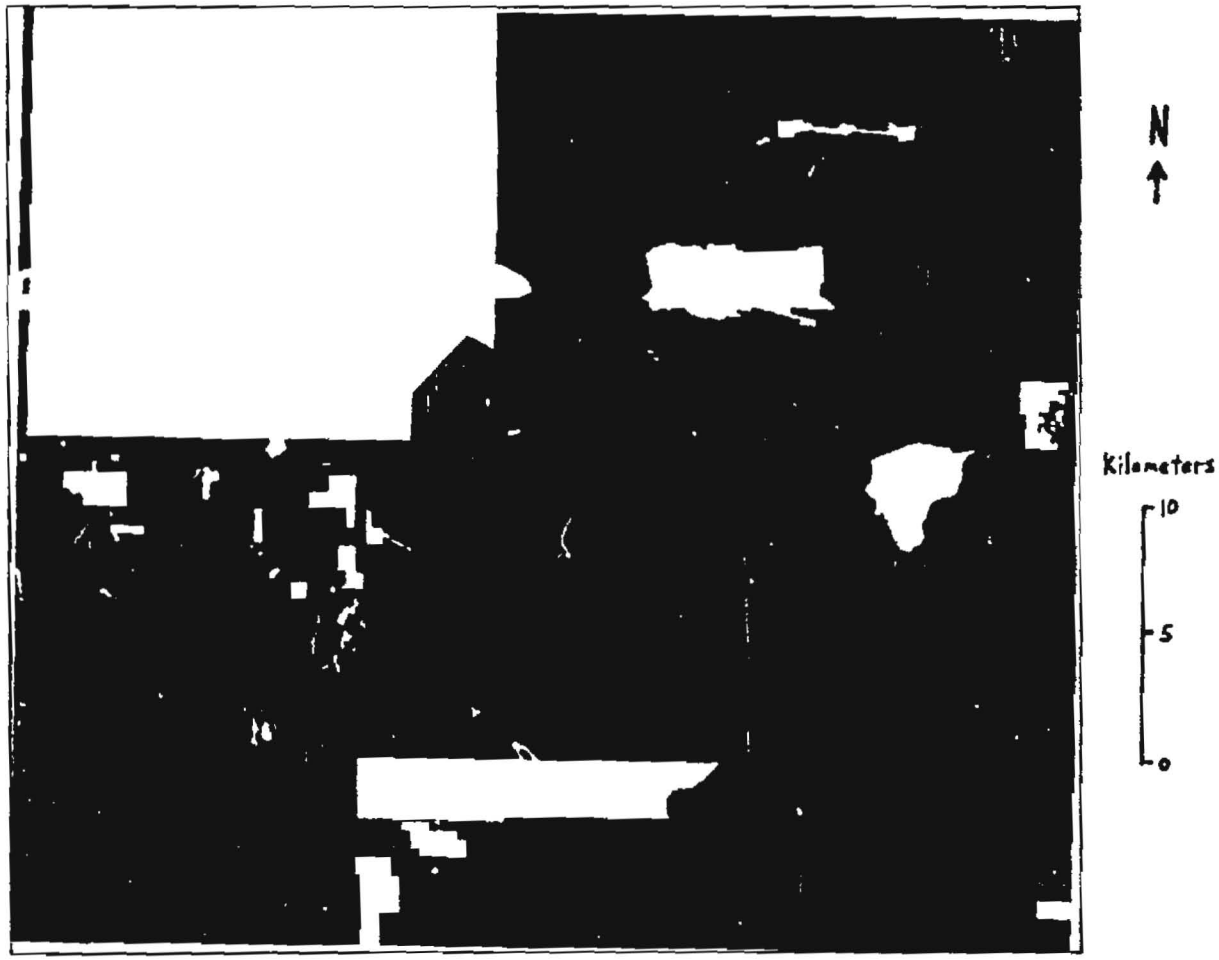


Figure 2-2.

General Services Administration (GSA) currently owns land adjacent to the city of Los Alamos. The portion of the SFNF in the southeast quarter of the map contains the Cerros del Rio, and is separated from the Pajarito Plateua lands of LANL and BNM to the northwest by White Rock Canyon and the Rio Grande. The linear slice of private land adjacent to BNM's southern boundary is the Cañada de Cochiti grant, which is currently owned by the University of New Mexico. The small outlying portion of BNM is known as the Tsankawi Unit.

OVERVIEW OF HUMAN OCCUPATION AND USE OF THE JEMEZ MOUNTAINS LANDSCAPE

Human presence in New Mexico dates back to at least 11,000 years BP with documented Folsom and Clovis Paleo-Indian sites scattered across the state, including areas adjacent to the Jemez Mountains (Stuart 1986). Small numbers of Paleo-Indian artifacts have been found on the Pajarito Plateau (R. Gauthier, local archeologist - personal communication). Agricultural practices began to supplement hunting and gathering after 7500 years BP in New Mexico during the Archaic Period, with Archaic sites found in the Jemez Mountains.

The ancestors of the modern Puebloan peoples developed sedentary agricultural lifeways with communities of permanent dwellings beginning about 1100 A.D. in the Jemez Mountains (R. Gauthier - personal communication). Population in the Jemez Mountains increased dramatically in the late 12th century with the influx of settlers from abandoned sites associated with the collapse of the Chaco Canyon culture. Population increase led to extensive Anasazi settlement of the mesas of the Pajarito Plateau occurred the 13th century, with large pueblos of up to 1000+ rooms being constructed by the 14th century.

During the 14th and 15th centuries much of the population shifted to adjacent river valleys with permanent water (Stuart and Gauthier 1986), although several large mesa-top sites were occupied in the 1500's. The Anasazi left immense numbers of ruins as evidence of their former presence; Bandelier alone contains an estimated 3000 to 4000 archeological sites.

Don Juan de Oñate established the first Spanish settlement in the Upper Rio Grande Valley in 1598 at the foot of the Jemez Mountains. By this time the mesas and mountains of the Jemez range were abandoned. The Spanish conquered and colonized the river valley pueblos, bringing domestic livestock that became the economic focus of this frontier area (Carlson 1969). Still, Navaho and Apache raiding parties kept the Jemez Mountains a dangerous and wild place until the early 1860's (Scurlock 1981, Rothman 1989), and the actual amount of earlier livestock grazing or other historic land use in the Jemez Mountains remains undocumented. It appears that at least the core areas of the Jemez Mountains were little-used by people throughout the early historic period, until the Navaho and Apache were "pacified" in the 1860's. Certainly by the 1880's large numbers of sheep and cattle were being grazed in the Jemez Mountains (Denevan 1969, Rothman 1989), and commercial logging has occurred continuously since the late 1890's (Foxx and Tierney 1984).

Anglo penetration and economic domination of north-central New Mexico has steadily increased since the opening of the Santa Fe Trail in 1821. This area was wrested from Mexico by the United States in the Mexican Cession of 1848. Since then the Jemez Mountains region has developed largely as a natural resource supplier for the U.S. economy,

providing timber, livestock, minerals, and fruits for processing and consumption elsewhere. Although the massive influx of Anglos and their capital into this Sunbelt region is rapidly changing the situation, the lands in and around the Jemez Mountains retain a substantial ethnic Spanish and Native American majority with a rural way of life still prominent. However, a major exception to this scenario is the "Atomic City" of Los Alamos, and its adjacent bedroom community of White Rock, on the Pajarito Plateau. Here workers in high-technology government research laboratories have developed nuclear weapons and other defense and energy related projects since 1943, when Los Alamos was established to build the first atomic bombs. "The Lab" has drawn in a largely Anglo, highly-educated population which has created an enclave of white-collar, middle-class "America" in this Hispanic and Indian world. Los Alamos has also acted as a catalyst for change in northern New Mexico by employing thousands of people from surrounding communities. In recent decades the Jemez Mountains have also become an important recreational resource due to the increase in outdoor recreational activities, the importance of Santa Fe as a world-wide tourist destination, and rapid population growth and urbanization in Los Alamos, Santa Fe, the Española Valley, and the nearby Albuquerque metropolitan area.

Adolph Bandelier conducted some of the first archeological work in the Southwest on the Pajarito Plateau in the late 1800's. Bandelier National Monument was established in 1916 to protect a number of prominent Anasazi ruins that Adolph Bandelier had made famous. The administrative history of the park is treated in detail by Rothman (in press). The U.S. Forest Service (USFS) operated Bandelier until it was

transferred to National Park Service (NPS) control in 1932. Bandelier's boundaries were enlarged in 1959, 1963, and 1977 to encompass all of the Rito de los Frijoles (Frijoles Creek) watershed. In 1976 Congress designated 9416 ha of the park as the Bandelier Wilderness, adjoining the smaller USFS Dome Wilderness. Bandelier's current boundaries encompass 13,248 ha, with another 1713 ha of the adjacent Cañada de Cochiti grant authorized by Congress for addition in 1976 still remaining outside NPS control. Visitation to Bandelier has been increasing at nearly 10% annually over the past decade; about 300,000 people visited Bandelier in 1988 (BNM - records on file).

THE DEVELOPMENT OF A LANDSCAPE ECOLOGY PARADIGM IN NORTH AMERICA

CONCEPTUAL BACKGROUND AND RECENT RESEARCH

The term **landscape ecology** was apparently coined by Carl Troll, a German geographer, in 1939 (Naveh and Lieberman 1984). Landscape ecology has existed for several decades in Europe and the Mediterranean as an applied science at the intersections of ecology, geography, and land use planning. Landscape ecology in North America has emerged more recently and along somewhat different lines (Romme 1987), which will serve as the focus of the rest of this review.

Advances in hierarchy theory (Allen and Starr 1982) and ecosystem theory (O'Neill **et al** 1986) have contributed to the development of the conceptual basis of North American landscape ecology, particularly in answering the question, "What is a landscape?" The ecosystem concept is flexible enough to include any system of living and inorganic components as an ecosystem, from the scale of a drop of pond water up to the ecosphere of the entire planet Earth (Rowe 1961, Schultz 1967,

Lidicker 1988). However, it is useful to consider as distinct (yet interacting) levels certain hierarchical divisions within this spectrum of various-sized ecosystems (Miller 1978, Allen et al 1984, Urban et al 1987). Start by restricting the definition of ecosystems to relatively homogeneous spatial "patches" on the surface of the earth, like an aspen stand or a clearcut. From the air, the earth is covered with a contiguous mosaic of such patches. A landscape can be defined as an area at least several kilometers wide where a recurring pattern of ecosystems (patches) is identifiable (Forman and Godron 1981). Landscapes aggregate into regions, regions form continents, and finally the continents and oceans aggregate at the global scale to form the ecosphere. Landscape ecology is concerned with the spatial distribution and interactions between ecosystem patches at the landscape level, as well as changes through time of landscape form and function (Forman and Godron 1986). While the definitions outlined above will be used in this work, note that dissent exists over preferred definitions for "ecosystem" and "landscape" (Lidicker 1988, T. Allen 1988), and over the scope of the field of landscape ecology (Toth 1988, Golley 1988).

Geographic thought also forms an important conceptual basis for landscape ecology (Forman and Godron 1986), particularly in Europe (Troll 1971, Naveh and Lieberman 1984). Landscape and regional studies are central to geography (James 1972). Physical geographers, from geomorphologists (Huggett 1985) to biogeographers (Troll 1971, Vale 1982) have devoted much effort to understanding the physical and biological phenomena of landscapes. Human geographers have looked at landscapes as cultural features, emphasizing that people are part of

landscapes (Sauer 1925, Tuan 1971). Geographers have long studied the strong human interaction with the earth at the landscape and regional levels. People live and work in landscapes, and our sense of belonging to a place is associated with landscapes (Tuan 1974). Geographers continue to explore the natural and human dimensions of landscape change (Turner et al - in press). In Europe geographers have been quite active in land use planning and natural resource management and form a core group of landscape ecologists, while in the U.S. geographers have been less involved with natural resource management. Meanwhile, American landscape ecology is developing in response to the increasingly clear perception that an understanding of and solutions to many current ecological questions and resource management problems must be sought at a landscape level (Risser 1985, Golley 1987). In the U.S. this has attracted ecologists who have become interested in working at this higher scale and landscape architects who see landscape ecology as an integrative paradigm for planning purposes as in Europe. However, relatively few geographers have become involved to date (personal observation).

Other ideas have also been important in the recent development of landscape ecology. Note that many of the ideas outlined below have also catalyzed the concurrent development of the field of conservation biology, which is explicitly aimed at maintaining as much of the Earth's biological diversity as possible (Wilson 1985, Soule 1985).

In the 1960's researchers theorized that species richness (S) is a function of island area (A) and distance from mainland for oceanic islands, producing species/area curves of the following form:

$$S = cA^z \quad \text{or} \quad \log S = \log c + z(\log A)$$

These curves presumably represent an equilibrium between local extinction and colonization processes, where:

-increasing island area leads to decreased extinction rates, due to higher population sizes and greater habitat diversity; and

-increasing distance from mainland species sources leads to lower colonization rates (MacArthur and Wilson 1967).

Some controversy continues to exist over the relative importance of extinction/colonization processes versus other factors (e.g. habitat diversity or *in situ* species evolution) in the development and maintenance of species diversity (Gilbert 1980), and over what role such island biogeography theory should play in conservation practice (Simberloff and Abele 1982, Boecklen and Gotelli 1984, Quinn and van Ripper 1986, Newmark 1987, Quinn *et al* 1988). Still, island biogeography theory has contributed to the recent development of several fields, including landscape ecology (as described below).

It was quickly realized that island biogeography concepts could be applied to continental "islands", such as forested montane islands in the arid sea of the Great Basin. For example, the conifer diversity of Great Basin mountains is a function of mountain area and distance from Rocky Mountain source areas (Wells, 1983), Great Basin montane mammal diversity is related to mountain area (Brown 1978), and small mammal distributions on Southwestern montane islands are correlated with area and degree of isolation (Lomolino *et al* 1989). In addition, paleoecological work with pollen, packrat middens, and vertebrate fossils over the last 20 years has given us a better understanding of how dynamic species distributions can be over the time frame of

millenia (Spaulding et al 1983, Delcourt et al 1983, Webb 1987). Where the flow of species colonization has been interrupted by barriers to dispersal, species richness has declined.

It has also become apparent that these concepts apply to mainland habitat "islands" that are created by human fragmentation of formerly continuous habitats (Wilcox and Murphy 1985, van Dorp and Opdam 1987). The effects of such habitat fragmentation is particularly worrisome in tropical and other Third World forests, where deforestation is rapidly occurring to create cattle pastures or farm fields, and to extract lumber and fuel wood. Habitat fragmentation increases local extinctions by dropping species populations below minimum viable sizes (Gilpin and Soulé 1986, Soulé 1987) and by inhibiting recolonization of isolated habitat patches, potentially serious problems in tropical forests where many species exist at low population densities. Tropical moist forests, "originally" (in historic times) covering only 7% of the Earth's land surface, contain at least half of the world's terrestrial species and are currently being cleared at a rate of 1-2% per year (Myers 1988). Ten percent of the Earth's plant and animal species may become extinct by the year 2000 (Raven 1988), largely due to the deforestation and fragmentation of tropical forests.

A large, long-term, research effort, the Minimum Critical Size of Ecosystems Project (Lovejoy et al 1984), is studying the effects of habitat fragmentation on extinction and colonization processes in the Amazonian rainforests of Brazil. Initial results confirm that small islands rapidly lose many species, and that edge effects may extend for several hundred meters into the interior of small stands. Concerns over the impacts of habitat fragmentation led to the successful raising

of over $2,000,000 in 1985 to keep one of the world's most productive tropical research sites, La Selva Station in Costa Rica, from becoming an island in a sea of cattle pasture. The money is being used to buy out small farmers in a strip of land that will maintain La Selva's connection to the higher elevation Braullio Carrillo National Park.

This burst of tropical habitat destruction and fragmentation has regional, continental, and global impacts that affect our temperate zone landscapes. Small islands of fir forest in south-central Mexico, shelter "our" overwintering populations of monarch butterflies in a few, threatened colonies (Colin 1986). About one half of "our" U.S. birds are Neotropical migrants that spend most of each year in tropical habitats that are undergoing rapid alteration - evidence of widespread population declines in some migrants are becoming apparent (Serrao 1985, Cunningham 1988, Robbins et al - manuscript). And the rapid and widespread destruction of tropical forests may lead to changes in the global climate affecting all of the Earth's landscapes (Myers 1988).

Habitat fragmentation is also worrisome in North America. For example, the impacts of the fragmentation of the eastern deciduous forests upon avifauna have been widely studied (Sharpe et al 1981, Wilcove 1985, McLellan et al 1986). The problems associated with the isolation of small old-growth islands in a sea of patch clearcuts in the Douglas-fir forests of the Pacific Northwest have led to calls for management at a landscape level (Harris 1984, Franklin and Forman 1987).

In addition, a great deal of recent research has documented the near-ubiquitousness of disturbance regimes in terrestrial ecosystems, from fire and flood to insect outbreaks and windthrow (White 1979,

Sousa 1984, Pickett and White 1985). Disturbances vary in frequency, intensity, and areal extent across a landscape (Turner 1987, Rykiel et al 1988), helping to create a landscape mosaic of "patches" that shifts through time as patches change through succession and disturbance (Bormann and Likens 1979). Local extinctions occur as each patch is periodically subject to disturbance, but recolonization from undisturbed nearby patches allows most species to persist (Pickett and Thompson 1978). However, when patches become isolated and/or decreased in size through habitat fragmentation, extinction rates increase, colonization rates decrease, and species diversity declines (Wilcox and Murphy 1985).

Recognition that landscapes are dynamic is another important cornerstone of landscape ecology - much recent work is focused on the topic of landscape change (Decamps et al 1988, Turner and Ruscher 1988, Delcourt and Delcourt 1988). While study of landscape change is certainly not new (Thomas 1956), the recent conceptual developments outlined above, contemporary approaches for environmental impact analysis (Dickert and Tuttle 1985, Walker et al 1987), advances in paleoecological techniques (COHMAP members 1988, Hunter et al 1988), and the availability of new technologies (e.g. remote sensing [Liliesand and Kieffer 1979] and geographic information systems [Burroughs 1987]) have provided fresh insights into the magnitude and extent of landscape change.

The recognition of regional and global changes is also leading to a surge of interest in the ecology of these levels (Bioscience 1984, Earth System Science Committee 1988, Committee on Global Change 1988). While initiatives like the International Geosphere-Biosphere Program

will be focused at the global level, some landscape level work will likely be promoted by these programs as well. For example, the effects of global atmospheric warming are predicted to have major impacts upon local landscapes that will require study (Peters and Darling 1985, Roberts 1988).

Conservation biologists and landscape ecologists have begun to use the concepts outlined above in the design and management of biotic preserves, with an emphasis on landscape-level concerns (Pickett and Thompson 1978, Kushlan 1979, Noss 1983, Harris 1984, Miller and White 1986, Soulé and Simberloff 1986, Noss and Harris 1986, Patterson 1987, Morse et al 1987, Franklin and Forman 1987, Office of Technology Assessment 1987, Simberloff and Cox 1987, Noss 1987, Schonewald-Cox 1988, Hunter et al 1988, Agee and Johnson 1988). Although biological diversity was stressed in the foregoing presentation, it is only one of many resources and ecological functions that are regulated, and thus need to be managed, at a landscape level (Swanson et al 1988). Other examples include surface water hydrology and groundwater flow (Dunne and Leopold 1978), soil development and movement (Hole and Campbell 1985, Steiner and Osterman 1988), energy flow (Ryszkowski and Kedziora 1987) nutrient cycling and associated site productivities (Bormann and Likens 1979), disturbance regimes (Turner 1987), and aesthetics (Zube 1987, Hendler 1988).

NATIONAL PARK SERVICE INTEREST IN LANDSCAPE ECOLOGY

In light of these developments the National Park Service has begun to view its park units as connected to larger surrounding landscapes, and to recognize that what happens to those larger landscapes affects

the parks (Forman 1987). Even the largest national parks have been forced to consider what is happening outside their boundaries. For example:

-Mount Ranier National Park, 7.5 times the size of Bandelier, has lost 15 species of mammals (31% of its total) between 1935 and 1976 (Weisbrod 1976). Newmark (1987) extended a similar analysis to 14 western North American parks and found that local mammalian extinctions were commonplace, indicating that almost all parks were too small to maintain their original complement of species due to insularization. While these data and interpretations are currently in dispute (Quinn et al 1988), they point out a major potential problem for all biological reserves.

-Yellowstone National Park. In order to handle issues like the management of grizzly bears, elk herds, and fire, 25 different state and federal entities are now attempting to work together to manage what has come to be known as the Greater Yellowstone Ecosytem. The agencies involved include Yellowstone and Grand Teton national parks, seven different national forests in three different regions, the Bureau of Reclamation, and the game and fish departments of Wyoming, Montana, and Idaho (Barbee and Varley 1985).

-Glacier National Park, beset by numerous development forces on both sides of the international border, may be the most threatened U.S. national park. Efforts are being made to coordinate resource management plans among the various agencies involved in this Front Range landscape (Haraden 1985).

-The Everglades. Since 1962, when a large water diversion project

went into effect "upstream", the Everglades have been subject to extreme water fluctuations that have damaged a variety of resources (Kushlan 1987). Water problems exist throughout southern Florida, and the state has decided to take the lead in trying to solve those problems. The state, irrigation districts, the U.S. Army Corps of Engineers, the NPS, and a variety of citizens groups are currently working together to better manage water flows in this landscape. These efforts include restoring the heavily channelized Kissimmee River to its natural, meandering condition by returning water to the still-existing but abandoned channel, at a cost of $65 million (Flowers 1985).

All of these efforts and trends reflect the emerging perception that the ecological boundaries of parks and reserves extend beyond their legal boundaries (Newmark 1985, Schonewald-Cox and Bayless 1986, Agee and Johnson 1988). Thus, a more holistic, landscape level approach to resource management is called for (Risser 1985, Noss and Harris 1986).

From this perspective there is a management need to view the mosaic of local ecosystems in Bandelier National Monument as part of the larger landscape of the Jemez Mountains. Over 100 ecological studies have been undertaken in Bandelier and the encompassing Jemez Mountains over the last 15 years, but this work has largely been directed at ecosystem components in isolation from the landscape in which they are imbedded over short time scales of one to several years. In contrast, this project takes a landscape-level look at Bandelier and its environs over longer time scales of decades to centuries.

CHAPTER III. METHODS

A. STUDY AREA AND TIME FRAME OF INTEREST

The study area for this research project was the entire landscape of the Jemez Mountains, although time and resource limitations dictated decreasing study attention with increasing distance from Bandelier National Monument. The landscape of the Jemez Mountains was delineated by:

- a common geologic origin, from the Jemez Volcanic Field;
- a common topographic pattern (the central caldera, surrounding rim peaks, and canyon-dissected skirting plateaus);
- an associated pattern of ecosystem patches on this topographic island; and
- a somewhat arbitrary extension of boundaries to the stream corridors of the Rio Grande, Rio Chama, Rio Cebolla, and Jemez River in order to encompass whole watersheds.

My research focused on landscape changes that have occurred over approximately the last four hundred years - the time period resolvable by the methods outlined below. Note that the time period accessible by each method varied, e.g., interpretation of landcape change from aerial photos only went back to the date of the earliest available imagery (1935), while analysis of tree rings for fire scars extended back to the 15th century.

B. DATA COLLECTION

1. FIELDWORK

Point Sampling of the Frijoles Watershed

The Rito de los Frijoles watershed in BNM provides a representative elevational transect of this landscape, with a rise of 1480 m from the Rio Grande to the caldera rim in the southeast flank of the Jemez Mountains (Figure 2-1). A stratified-random array of 969 sample points was established on a 1:24,000 USGS topographic map, providing complete coverage of the Frijoles watershed and small portions of adjacent drainages (Figures 3-1 and 3-2). Sample point density averaged one point per 6.7 ha across this 6500 ha sample area. Each pre-determined sample point was located in the field using topographic cues and triangulation from landmarks where necessary. I sampled all points between April 30, 1987, and December 3, 1987, usually with the extraordinary assistance of Ms. Lee Sullivan.

At each point data were collected on numerous landscape features typical of the "patch" which contained the sample point. Patches were qualitatively determined based upon relatively uniform combinations of overstory vegetation species association, overstory canopy cover, and landform, with a minimum patch size of 1.5 ha. Landforms were classified following the criteria used by the U.S. Forest Service in regional soil surveys (USDA Forest Service 1986). Slope position was split into five categories: bottom (in a canyon, valley, or swale landform); lower (lower 20% of slope length); mid (middle 60% of slope length); upper (uppper 20% of slope length); and crest (on top of a ridge landform). Elevation was determined from the topographic map. Percent slope was measured with a clinometer. Site aspect was

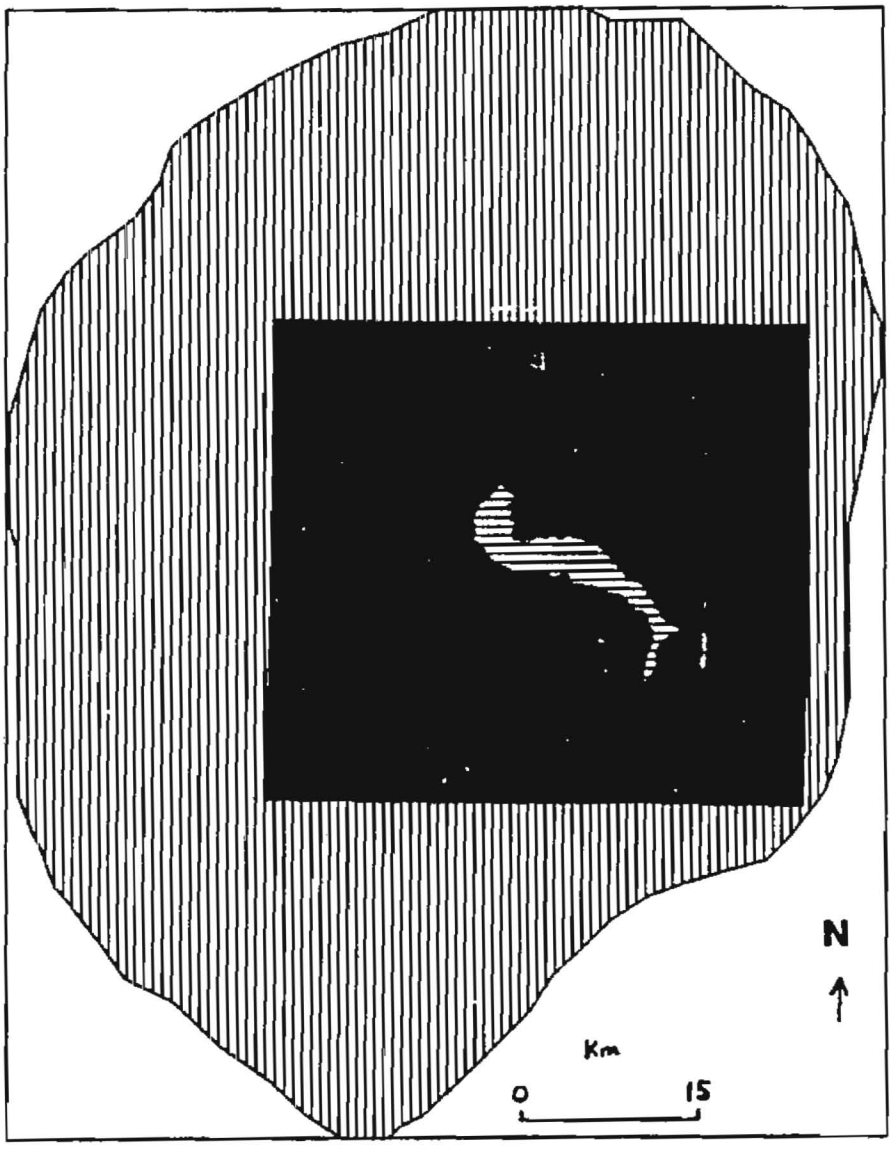


Figure 3-1. Location of mapped areas in the Jemez Mountains (vertically striped area). The black square defines the 187,858 ha map area for 1935 and 1981 road networks and 1980 land ownership, the gray patch was mapped for 1935 and 1981 landscape cover-types, the irregular black area was mapped for 1981 ecosystem patches, and the horizontally striped area is the Frijoles watershed which was sampled with 969 points in the field.

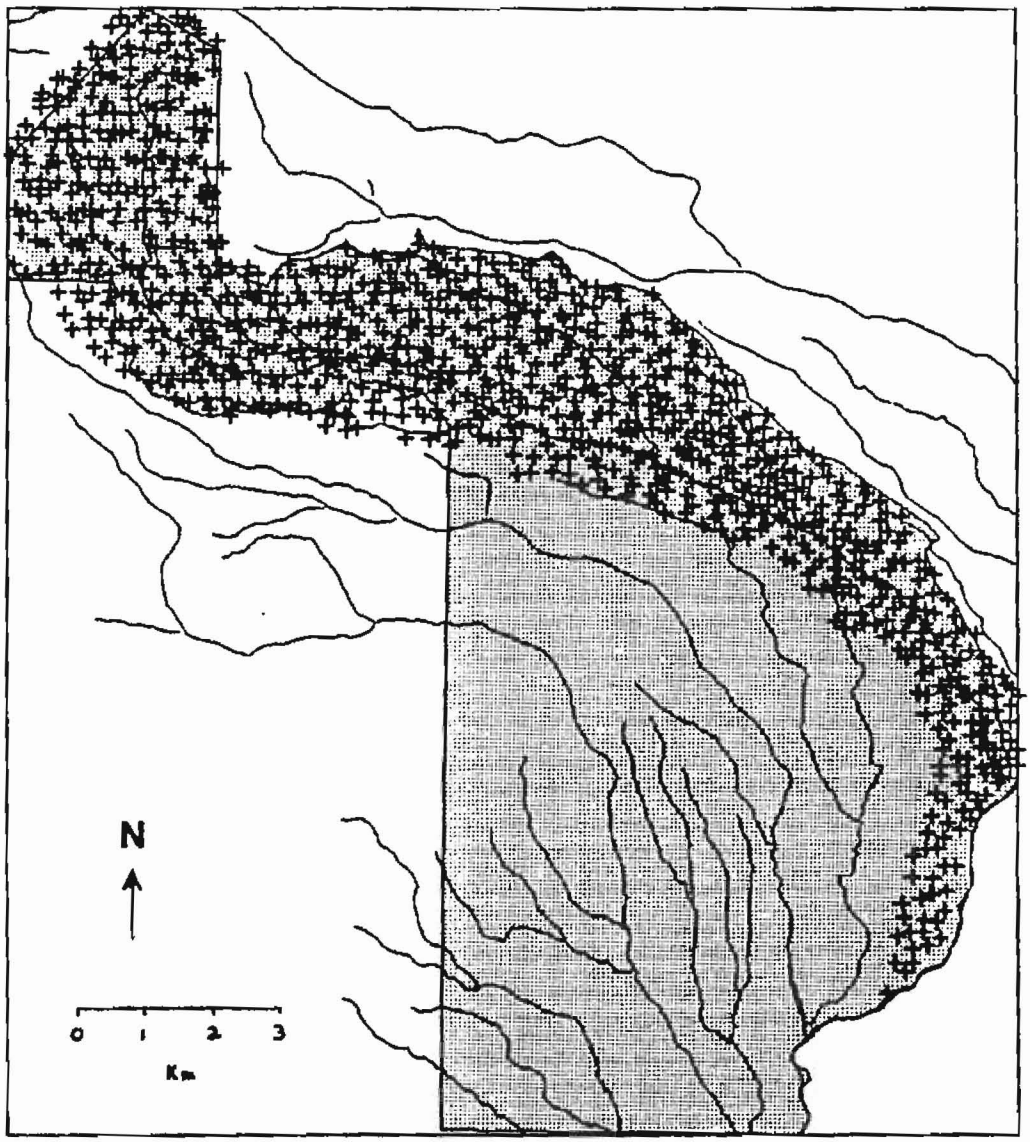


Figure 3-2. Frijoles watershed sample point location, with the main portion of Bandlelier National Monument shown in gray and stream drainages as lines.

determined with a compass to the nearest 5 degrees, corrected for magnetic declination. Percent bare rock was visually assessed. Apparent pumice-derived soils were recorded if covering 20% to 50%, or > 50%, of the site. Soil erosion was qualitatively classified as severe, moderate, or past severe/moderate based upon such evidence as amount of bare surface soil, pedestalled plants, gullies, absence of lichen on (freshly) exposed bedrock surfaces, and desert pavement-like surfaces of coarse gravel and cobbles which persist after the finer soil particles have been lost from the site.

Each tree species present within the sampled patch (typically within 50 m of the sample point) was assigned an abundance value. Scarce trees constituted < 1% absolute canopy coverage, abundant species were codominants with > 15% absolute cover (such as piñon in a piñon-juniper woodland) or subdominants with > 25% canopy coverage (e.g. white fir in the understory of an aspen stand), and common species filled the range between scarce and abundant. If snags, logs, or stumps indicated recent declines in species abundance a pre-decline abundance value was also estimated. Presence data were collected for all prominent shrub species. The common name, scientific name, and an acronym for each woody species discussed in this dissertation is displayed in Table 3-1.

The patch sampled by each field point was assigned a current cover-type based upon overstory vegetation (usually) or dominant non-vegetative site characteristics. A dichotomous key was developed to classify the cover-type of each site. Sites with $\geq$ 10% absolute canopy cover of trees were classified into one of the forest or woodland cover-types displayed in Table 3-2, based upon the relative cover of

TABLE 3-1. Common names, scientific names, and acronyms of tree and shrub species used in Figures, Tables, and text.

Common Name	Scientific Name	Acronym
Corkbark fir	Abies lasiocarpa var. arizonica	ABLA
Engelmann spruce	Picea engelmanni	PIEN
Rocky Mt. maple	Acer glabrum var. neomexicana	ACGL
Quaking aspen	Populus tremuloides	POTR
Blue spruce	Picea pungens	PIPU
Limber pine	Pinus flexilis	PIFL
White fir	Abies concolor	ABCO
Douglas-fir	Pseudotsuga menziessi	PSME
Scouler willow	Salix scouleriana	SASC
Ponderosa pine	Pinus ponderosa	PIPO
Hawthorn	Crataegus erythropoda	CRER
Serviceberry	Amelanchier bakeri	AMBA
Cherry species	Prunus spp.	PRsp
Alligator juniper	Junierus depppeana	JUDE
Water birch	Betula occidentalis	BEOC
Rocky Mt. junier	Juniperus scopulorum	JUSC
Mountain alder	Alnus tenuifolia	ALTE
Piñon	Pinus edulis	PIED
One-seed juniper	Juniperus monosperma	JUMO
Box elder	Acer negundo	ACNE
Narrowleaf cottonwood	Populus angustifolia	POAN
Gambel oak	Quercus gambelli	QUGA
Wavyleaf oak	Quercus undulata	QUUN
New Mexico locust	Robinia neomexicana	RONE
Narrowleaf hoptree	Ptelea trifoliata	PTTR
Mountain mahogany	Cercocarpus montanus	CEMO
Fendler ceanothus	Ceanothus fendleriana	CEFE
Common juniper	Juniperus communis	JUCO
Apache plume	Fallugia paradoxa	FAPA

the dominant species. These forest cover-types are similar to those described for this area by Potter and Foxx (1981) and (USDA Forest Service 1987-a). Dominant species were quite variable in riparian associations, so these cover-types were lumped into three groups based upon the growth habit of the dominant trees (deciduous, evergreen, or mixed). Non-forest cover-types were classified into physiognomic categories without regard for species dominance (Table 3-3). The percent absolute tree cover was estimated for each patch, which placed the site into one of five cover classes (Table 3-4). If dead woody

Table 3-2. Forest and woodland cover-type abbreviations used with field sample points, landscape cover-type maps, and the ecosystem patch map.

RI-E	Riparian - Evergreen
REPP	RI-E (Ponderosa Pine)
REMC	RI-E (Mixed Conifer)
RI-M	Riparian - Mixed Evergreen/Deciduous
RMPD	RI-M (Ponderosa Pine - Deciduous)
RMCD	RI-M (Mixed Conifer -Deciduous)
RI-D	Riparian - Deciduous
RDNC	RI-D (Narrowleaf Cottonwood)
RDAN	RI-D (Box Elder)
WRSW	White Rock Canyon Spring/Seep Woodland
J	Juniper (JUNI) Woodland
J-S	Juniper - Shrub Woodland
PJ	Piñon-Juniper Woodland
PJ-S	Piñon-Juniper - Shrub Woodland
PPPJ	Ponderosa Pine - Piñon-Juniper
PPJS	Ponderosa Pine - Piñon-Juniper - Shrub
QG	Gambel Oak (trees)
PP	Ponderosa Pine
PPQG	Ponderosa Pine - Gambel Oak (trees)
PP-S	Ponderosa Pine - Shrub
PPDF	Ponderosa Pine - Douglas-fir
PDFS	Ponderosa Pine - Douglas-fir - Shrub
PPMC	Ponderosa Pine - Mixed Conifer
PMCS	Ponderosa Pine - Mixed Conifer - Shrub
DFPP	Douglas-fir - Ponderosa Pine
DFPS	Douglas-fir - Ponderosa Pine - Shrub
DFES	Douglas-fir - Engelmann Spruce
BS	Blue Spruce
MC	Mixed Conifer
MC-W	Mixed Conifer - White Fir
MC-D	Mixed Conifer - Douglas-fir
MC-A	Mixed Conifer - Aspen
MC-S	Mixed Conifer - Engelmann Spruce
MC-B	Mixed Conifer - Blue Spruce
MC-S	Mixed Conifer - Shrub
TA	Aspen
TAPP	Aspen - Ponderosa Pine
TAMC	Aspen - Mixed Conifer
TAES	Aspen - Engelmann Spruce
ES	Engelmann Spruce
ES-F	Engelmann Spruce - Corkbark Fir
EDSF	Engelmann Spruce - Douglas-fir - Corkbark Fir
CWCX	Canyon Wall Complex (variable forest cover on cliffs)

Table 3-3. Non-forest cover-type abbreviations used with field sample points, landscape cover-type maps, and the ecosystem patch map.

CWCX	Canyon Wall Complex (variable non-forest cover-types on cliffs)
ROCK	Bare Rock (cliffs)
FELS	Felsenmeer
TALU	Talus
JSAV	Juniper Savanna (5-9% juniper cover)
SHRU	Shrubland
SH-O	Oak Shrubland
MG	Montane Grassland (upper slopes of caldera rim peaks only)
MEAD	Meadow (depressions and other moist sites)
GRAS	Grassland (all other grasslands)
GRSH	Grass-Shrub (a mixture of grassland and shrubland)
STRE	Stream (Rio Grande)
POND	Pond
LAKE	Cochiti Lake
SAND	Sandbar/Mudflats (along Rio Grande)
DROW	Recently Drowned by Cochiti Reservoir
FARM	Agricultural Fields
RES	Residential Areas
COMM	Commercial Areas
IND	Industrial Areas (LANL Technical Areas)
ARCH	Archeological Site (large sites)
SKI	Ski Area
GOLF	Golf Course

Table 3-4. Cover classes used with field point samples, landscape cover-type maps, and ecosystem patch maps.

Cover Class	% Absolute Canopy Cover
0	0
1	1 - 9
2	10 - 39
3	40 - 69
4	70 - 100

material indicated a recent alteration of the cover-type on the site (e.g., the snags and logs left by a crown fire) a pre-alteration cover-type and cover-class were estimated.

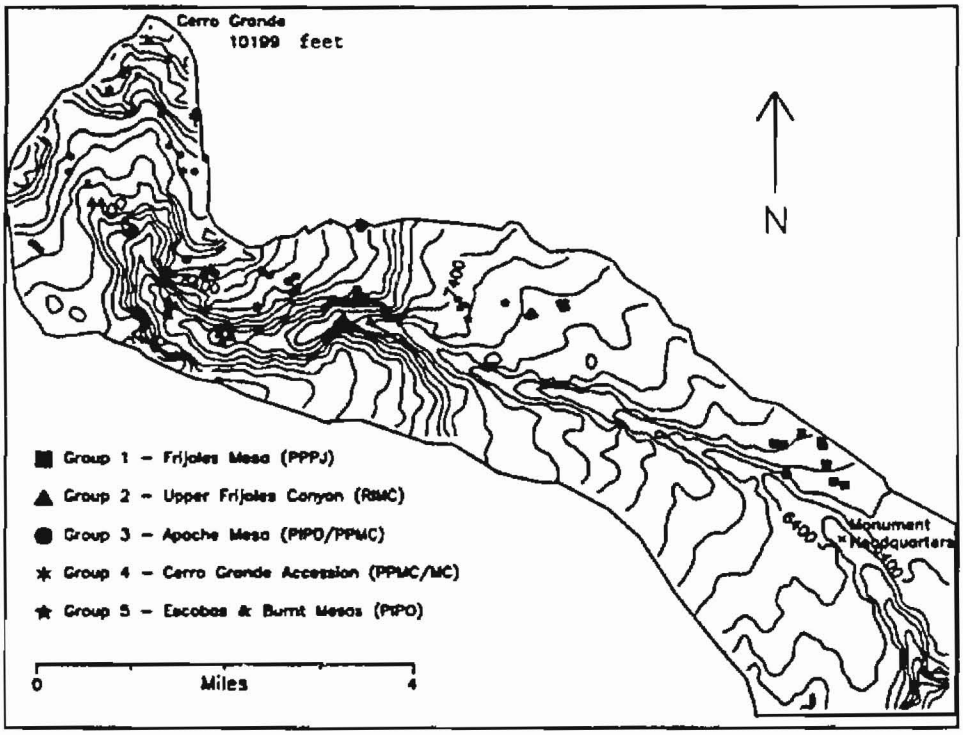
At each sampled point obvious indications of past disturbances were recorded. In 1977 the intense La Mesa Fire burned through the

core of the Frijoles watershed - the burn pattern (crown fire, patchy crown fire, surface fire) was recorded at each sample site. Old fire-scarred trees and snags that pre-dated the La Mesa Fire, as well as recent scars from the 1977 fire, were documented. Lightning-scarred trees were noted, if present. The occurrence of various tree diseases was recorded, including: dwarf mistletoes (Arceuthobium spp.) on ponderosa pine, Douglas-fir, and piñon; true mistletoes (Phoradendron spp.) on one-seed and Rocky Mountain junipers; western gall rust (Endocronartium harknessii) on ponderosa pine; and juniper rust (Gymnosporangium spp.) on junipers. Obvious signs of recent damage from western spruce budworm (Choristoneura occidentalis), western tent caterpillar (Malacosoma californicum), or fall webworm (Hyphantria cunea) were noted. Where large numbers of long-dead ponderosa pine or piñon trees were present along their respective lower range limits a suspected bark beetle disturbance was recorded. Mammal disturbance to vegetation was documented, including buckrubs, severe/moderate browsing of shrubs (based upon degree of shrub hedging), and bark removal from aspen by the gnawing of elk (Cervus elaphus) and from conifers by porcupines (Erithizon dorsatum). Relict stumps and snags were used to document various types of tree cutting, including old selective logging of timber species, old woodcutting of piñon and juniper, salvage logging after the La Mesa Fire, old felling of snags, tree cutting associated with firefighting, other management-related tree cutting within BNM, and recent timber sales on the SFNF. Evidence of flooding, hail and snow damage, landslides, and windthrow were also recorded.

Fire Scar Sampling and Processing

Fire scars were collected from five areas in the Frijoles watershed (Figure 3-3), namely: Group 1, ponderosa pine/piñon juniper transitional forests on Frijoles Mesa; Group 2, riparian mixed conifer forests in upper Frijoles Canyon; Group 3, ponderosa pine/ponderosa pine - mixed conifer forests on Apache Mesa; Group 4, ponderosa pine-mixed conifer/mixed conifer forests from the upland headwaters area of the Frijoles watershed in the Cerro Grande Accession; and Group 5, ponderosa pine forests on Escobas and Burnt Mesas. Within each group I sought trees with the maximum number of visible fire-scars, and where possible, samples were collected from clusters of 2-4 nearby trees. An April 1988 visit by Dr. Tom Swetnam and Chris Baisan of the Tree-Ring Lab (University of Arizona, Tucson) provided valuable insight into the desirability of collecting scarred snags and downed trees to extend the fire history as far back in time as possible. With the assistance of Bandelier's seasonal firefighting crew, one to three cross-sections were removed from each of 73 fire-scarred trees in the five groups. These samples received initial sanding to a 200 grit surface by Bandelier's firefighters before delivery to the Tree-Ring Lab.

At the Tree-Ring Lab each sample was sanded to a 320 or 400 grit surface. Sixty of the samples were crossdated by the Tree-Ring Lab, and I crossdated four samples, by comparison of skeleton plots with the Rio Grande master chronology (Schulman 1956). Nine samples are undatable at this time due to complacent ring series that fail to match the master chronology patterns from some mesic Group 2 and Group 4 sites or because of a combination of missing rings and extensive bark beetle galleries in some Group 1 samples. All samples are archived at



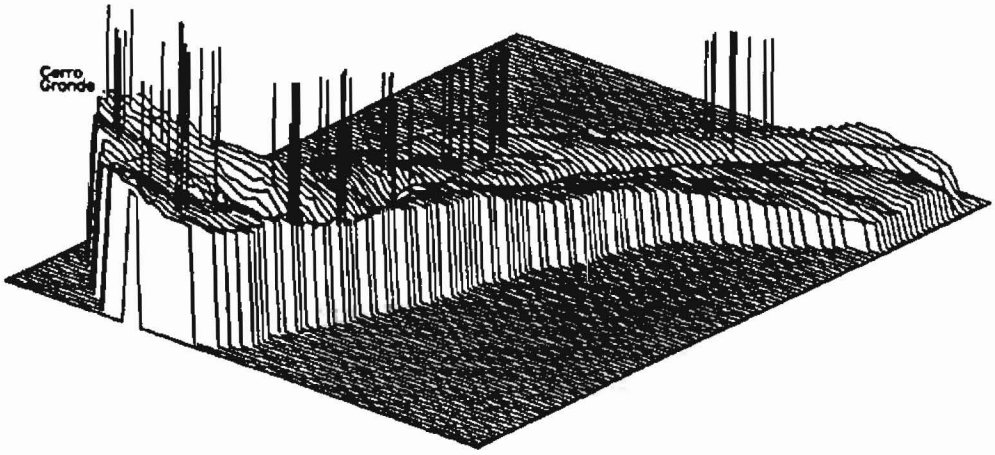


Figure 3-3. Fire scar sample locations in the Frijoles watershed.

the Tree-Ring Lab. More detailed descriptions of the methods employed in dating these samples is provided by Caprio et al (1988).

Fire scar dates were accurately determined to year of occurrence by location of scars within dated annual rings (cf. Madany et al 1983). Scars of possible, but not certain, fire origin were noted. The relative position of each fire scar within its annual ring of occurrence was classified into one of the following seven categories: EE = early earlywood, scar occurs within the first third of the earlywood portion of the ring; ME = mid earlywood, scar occurs within the middle third of the earlywood; LE = late earlywood, scar occurs within the last third of the earlywood adjacent to the first latewood cells; E = earlywood, scar occurs anywhere in the earlywood - used when further subdivision was not possible or when scar occurs within a false ring; L = latewood, scar occurs within the latewood cells of the ring; D = dormant, scar occurs on the boundary between the latewood of one year and the earlywood of the next year - by convention dates for dormant scars are recorded as the year of the adjacent earlywood in the Southwest (Dieterich and Swetnam 1984); and U = unknown, relative scar position could not be determined although the fire year was datable.

I collected only one sample from Group 5 (sample # 26, Table 4-2). The other eleven samples used to compile a fire history for this group were previously collected and dated by other researchers. Three cross-sections (sample #'s 89, 90, and 99) were collected in 1976 by J. Dieterich and crossdated at the Tree-Ring Lab (unpublished dating sheet on file at the Tree-Ring Lab), while tree #'s 91-98 were collected by T. Foxx in 1977 and also crossdated at the Tree-Ring Lab (Robinson 1978). Foxx and Potter (1978, 1984) used these earlier data, in

association with scar dates that they independently determined from another 15 samples, to develop a fire history for this area. For consistency my fire history chronology used only earlier data from samples crossdated at the Tree-Ring Lab, which were analyzed in the same fashion as the data from my 1988 collections. Field locations for the earlier samples are from T. Foxx (unpublished map). Data on relative scar position within annual rings are absent for most scars in this group.

Tree sample number, group number, innermost ring date, outermost ring date, fire scar dates, and relative scar position data were entered into dBase III Plus databases and delivered in hardcopy and database format to BNM as part of the Tree-Ring Lab's contracted final report on fire scar dates (Caprio et al 1988). Utilizing this database I compiled composite fire interval graphs (Dieterich 1980-a) and reviewed this voluminous data set carefully to insure accurate and consistent data. Scars of uncertain causal origin were deleted from the data base unless confirmed by a certain fire scar date from the same year and from the same sample group. This procedure deleted 48 questionable scar dates from the database, leaving a highly accurate, conservative data set of 1160 dated fire scars.

Plots for Forest Structure

Quantitative measurements were taken of tree diameters and species composition in 600 m^2 plots at eight forest sites. These 20 m by 30 m plots were located in forest stands that I believed illustrated representative changes in forest structure and species composition. Most plots were associated with field sample points on sites that were

never logged. The diameter of each living stem was measured at breast height (1.4 m) and placed into a 5 cm size-class (0 - 4.9 cm, 5 - 9.9 cm, etc.) by species; stems < 1.4 m tall were classified as seedlings. In each plot several trees in different size-classes were aged with an increment corer or by basal cutting.

Montane Grassland Studies

This dissertation updates the extensive fieldwork I conducted in Jemez montane grasslands between 1982 and 1984 (Allen 1984-a). Field data were collected from eleven montane grasslands found on the caldera rim peaks in an arc extending from Cerro Pelado to Polvadera Peak. The most detailed information was collected from Cerro Grande, Cañada Bonito, Chicoma Ridge, and Polvadera Peak. Methods relevant to the discussion presented here are described below.

On Cerro Grande a belt transect (VA), composed of 54 5 X 5m adjacent quadrats in a line, was run 270m upslope out of an aspen clone, through a closed-canopy stand of young ponderosa pine, and out into the open grassland on the ridge-crest. In each of the 54 quadrats the 2 largest trees of each species were cored to determine their age. Every 30m a soil pit was trenched to reveal the nature and extent of the surface horizon; moist and dry color, depth, and hand-texture of the exposed horizons were recorded. Four additional soil pits were dug downslope. In the open grassland just above the end of Transect VA, 46 young, invasive trees were randomly selected, cut off at the base, and aged (VX). Another belt transect (VB), 140 X 5m, was placed through the remnant grassland on the south-facing slope over the ridge-crest ecotone into an Engelmann spruce forest on the north-facing slope. All

trees in the grassland were dated, while in the spruce forest the 2 largest trees/species/ quadrat were cored. Four soil pits were dug along this transect.

At Cañada Bonito a belt transect (IA), 35 X 5m, was placed across the east-side ecotone between grassland and aspen forest. All trees were cored. A soil pit was dug in each of the seven 5 X 5m quadrats. A 1.08 hectare block (IB) of 12 30 X 30m quadrats was established in a grid pattern in the most heavily-invaded central portion of this grassland. All 171 trees within these quadrats were aged. Data from a concurrent U.S. Forest Service soil survey transect that crossed this site were utilized for soil information.

At Chicoma Ridge a belt transect (IIA), 120 X 5m, was run downslope out of a large grassland into an aspen stand on its lower slopes. The largest 2 two trees in each 5 X 5m quadrat were cored. Two soil pits were dug.

On Polvadera Peak a belt transect (IIIC), 45 X 3m, was run across a patch of tree invasion near the summit. All trees in each 5 X 3m quadrat were cored. Two soil pits were dug.

At all sites, each tree in every quadrat was mapped, with species and dbh recorded. A 0.406m increment borer was used to obtain cores at the lowest possible height on the tree trunk; core height was recorded. The cores of trees with clearly visible growth rings were generally read immediately after extraction, using a 10X hand lens and counting each core at least twice. Cores which were difficult to count for any reason, and all aspen cores, were counted in the laboratory under a 10-45X zoom microscope after sanding a smooth surface. Seedling and sapling trees were cut off at the base and the rings read

directly from the knife-smoothed stump face with a hand lens. The number of rings to the pith of an off-center increment core was estimated as per Campbell (1981). Total tree age was calculated as the sum of the increment core ring count, the estimated miss from the center (usually 0-3 years), and the estimated number of years to reach the core height (typically 1-4 years, depending on core height and tree species).

Soil information was obtained from soil surveys of the area (Nyhan et al 1978; J.Gass, P.Price, and W.Lucas 1979-1983, unpublished pedon descriptions on file at the Santa Fe National Forest), consultations with these soil scientists, and my own field observations. Mollic epipedons, the diagnostic surface horizon of a Mollisol, were indicated in the field by soil colors of 10YR 4/2 (dry) and 10YR 2/2 (moist) or darker.

The sheep-grazing history of the Santa Fe National Forest since 1910 was compiled by reviewing the grazing permits on file at the Forest Supervisor's Office in Santa Fe. One third of the total permits were sampled, including known large permits. These permits do not allow the grazing history of individual grasslands to be resolved. These data reflect sheep numbers on the whole SFNF, not just those in the Jemez Mountains. The totalled data were averaged to give the mean number of sheep/year by five-year period.

2. GEOGRAPHIC INFORMATION SYSTEM DEVELOPMENT

My research has utilized a public domain geographic information system (GIS) known as MOSS (Map Overlay and Statistical System). The versions of MOSS I have used run on Data General mainframe computers.

The methods used to develop a GIS database for this project are reviewed below.

Mapping

Two sets of aerial photographs were interpreted with a mirror stereoscope to create most maps: 1935 black and white photos at a scale of 1:31,680 (the earliest available imagery for the study area); and 1981 color photos at a scale of 1:24,000 (the latest comparable-scale, complete coverage). While not perfectly compatible, this was the best-matched imagery available.

All mapping was done on mylar overlays of 1975 vintage, 1:24,000 scale, black and white, orthophoto transparencies on a light table. Each orthophoto corresponded precisely to a 7.5 minute, USGS topographic quadrangle. This technique allowed direct and spatially accurate mapping. Some prior knowledge of the landscape is necessary to make correct map interpretations. Field point sampling and other field excursions were used to ground truth all maps.

Landscape Cover-Types - 1935/1981

Polygon maps of 1935 and 1981 landscape cover-types were created for a 85,726 ha portion of the Jemez Mountains in and around BNM (Figure 3-1). Map units were based upon overstory cover-type (Tables 3-2, 3-3), cover class (Table 3-4), and landform. Forest and woodland cover-types were mapped at a resolution of 15 ha, while non-forest cover-types had a minimum map unit size of 1.5 ha. At this coarse level of resolution all forest cover-types were lumped together as "forest", while all woodland cover-types were recorded as "woodland".

Following my specifications, John Selkirk created these 1935 and 1981 landscape cover-type maps, which I edited and slightly revised.

Road Networks - 1935/1981

All roads visible on the 1935 and 1981 aerial photographs were mapped across 187,858 ha of the Jemez Mountains (Figure 3-1). Roads were classified into five categories: railroads; paved roads; improved roads, which typically displayed a gravel surface; dirt roads, exhibiting a dirt surface where a road bed had clearly been cut into the terrain; and primitive roads, a catch-all category of logging skid trails, four-wheel drive roads. and various informal road traces. Most road types mapped easily and directly onto the orthophoto overlays, but interpretation of primitive roads involved some subjectivity as they were often at the limits of aerial photo resolution and difficult to see where present beneath forest overstories. In particular primitive road densities were likely underestimated for portions of the 1935 landscape due to limited resolution of the available photographs. John Selkirk mapped most of the 1981 roads, while I mapped the 1935 landscape and edited and revised portions of the 1981 road map.

Ecosystem Patches

Ecosystem patches were mapped at a 1.5 ha level of resolution across 28,684 ha of the 1981 landscape in and around BNM (Figure 3-1). Map units were based upon combinations of cover-type (Tables 3-2, 3-3), cover class (Table 3-4), and landform. Mary Clare Thomas mapped about 1/2 of the map area, which I subsequently revised and completed.

Creation of this map was extremely labor-intensive.

Miscellaneous Maps

Field sample point and fire scar sample locations in the Frijoles watershed (Figures 3-2 and 3-3) were mapped directly onto 1:24,000 scale, USGS topographic maps. The land ownership map of the 187,858 ha around BNM (Figure 2-2) was aquired as a GIS map from the USDI Bureau of Land Management, New Mexico state office, in Santa Fe. I revised the land ownership map to reflect more accurate boundaries for BNM, LANL, GSA land, and the townsites of Los Alamos and White Rock.

Digitizing

All of the primary maps used in this research were digitized using the Analytical Mapping System (AMS), a software package that produces precise results. The maps of field sample points, fire scar samples, 1935 and 1981 landscape cover-type maps, 1935 roads, and ecosystem patches were digitized by various personnel at the Southwest Regional Office of the National Park Service in Santa Fe. Hardware and software problems with the Regional Office system, combined with the large and complex nature of the maps being entered, greatly slowed the digitizing process. The intricate 1981 road network maps were digitized under contract by TGS Technology Inc. of Fort Collins, Colorado. Once digitized each map was loaded into MOSS at the Regional Office in Santa Fe. I have made revisions to several maps, and created several small maps, using the "GENERATE" routine in MOSS.

Database Development

MOSS allows multiple attribute databases to be attached to maps, allowing identification of any individual map unit or analysis of the entire map to be conducted on the basis of any of the attribute data fields. Multiple attribute databases have been contructed and attached to the Frijoles watershed field sample point map, the 1935 and 1981 landscape cover-type maps, and the 1981 ecosystem patch map. For example, I entered the field sample point data into a dBase III Plus database on a personal computer, generating a 79-field database for each sample point. Bruce Panowski, then with the Regional Office staff of the NPS, wrote a number of programs that converted the data (with some difficulty) from dBase into MOSS format and attached them properly to the map. This turned out to be a surprisingly difficult procedure. Once databases were attached to the MOSS maps both maps and databases were reviewed for errors and edited before analytical use of the GIS began.

3. REVIEW OF HISTORICAL INFORMATION AND PAST LOCAL RESEARCH

Historical sources of data sought out include old maps used to check the accuracy of 1935 airphoto interpretations. Caution must be used when interpreting old maps, as mapped locations are sometimes approximate or in error. Old photographs were reviewed from the USGS library in Denver, BNM files, SFNF files at the Supervisor's Office in Santa Fe, and the Los Alamos County Historical Society. Written records of many types were examined. I found old files at BNM to be particularly useful, as they document and pinpoint the dates of past natural events such as windthrow and bark beetle outbreaks that can

still be detected in the field but are hard to reconstruct precisely, as well as past management actions such as pesticide spraying that are now lost from the collective memory of BNM staff. The historical literature on local land use which is vital to assess the timing and magnitude of past human landscape impacts like livestock grazing and logging. All known local ecological research, over 150 post-1975 references, has been reviewed for landscape implications.

4. CONSULTATION WITH LOCAL RESEARCHERS AND RESOURCE MANAGERS

Finally, I consulted with local researchers, resource managers, and "old-timers" to tap their experience and unique perspectives on this landscape. To some people this may seem like a trivial way to acquire information, but I believe it is important. It often takes patience and judgement to sort through the idiosyncractic communication styles and experiences of different individuals, but the unique and often long-term contact of these people with a local landscape is an information source that should be explored.

C. DATA ANALYSIS

The data analysis for this study emphasizes simple descriptive statistics, direct gradient analyses, and GIS analysis of spatial patterns. I used three computer software tools to assist in the analysis of my data: dBase III Plus, a database management program; CHART, a data analysis and graphics program; and MOSS, a GIS. Databases were constructed in dBase III Plus to store and analyze data from field sample points, fire scars, landscape cover-type maps, and the ecosystem patch map. Data were transferred from these databases,

or directly entered, into CHART to calculate descriptive statistics and produce scatter, line, and bar graphs. All GIS data analysis and map production has been conducted with the new 32-bit version of MOSS, currently undergoing beta-testing at LANL, Facilities and Planning Division (USDI Bureau of Land Management and USDI Western Energy and Land Use Team 1988). Once analyzed the eclectic array of multi-level, spatiotemporal information collected for this research was organized and integrated mentally to produce scenarios of landscape change. Specific methods of data analysis are described below.

FRIJOLES WATERSHED POINT SAMPLES

A topographic gradient value (TGV) was calculated for each sample point, based upon site aspect or landform as shown in Table 3-5. These TGV's simply assign each site a numeric position along a gradient of moisture status ranging from one for the most mesic sites to fifteen for the most xeric sites. This allows the field data to be analyzed and displayed as a function of site moisture status, a form of direct gradient analysis (Whittaker and Niering 1965, Gauch 1982).

A topographic potential moisture index (TPMI) was also calculated for each sample point, in a modification from Parker (1980). The TPMI provides a more sensitive measure of site moisture status than the TGV because it adds % slope and slope position information to the site aspect and landform factors, as presented in Table 3-5. TPMI values range from 50 on the most mesic sites to 5 on the most xeric sites. TPMI values provide a broader spread of site distributions than TGV's that was advantageous for graphically displaying landscape patterns.

Data on landscape phenomena of interest, for example, the points

TABLE 3-5. Calculation of TGV and TPMI values.

TGV	COMPASS BEARINGS	ASPECT (DEGREES)	TPMI ASPECT VALUE
4	NNE	10-27	20
5	NE, N	28-45, 352-9	18
6	NE, NNW	46-63, 334-351	16
7	ENE, NW	64-81, 316-333	14
8	E, NW	82-99, 298-315	12
9	ESE, WNW	100-117, 280-297	10*
10	SE, W	118-135, 262-279	8
12	SE, WSW	136-153, 244-261	6
13	SSE, SW	154-171, 226-243	4
14	S, SW	172-189, 208-225	2
15	SSW	190-207	0

MODIFICATIONS FOR SPECIAL LANDFORMS:

TGV	LANDFORM	TPMI ASPECT VALUE	TPMI TOTAL
1	Canyon		50
2	Valley		47
3	Swale		44
11	Ridge Crest, and ≤ 10% Slope	7	

% SLOPE	TPMI SLOPE VALUE
0 - 5	10
6 - 10	9
11 - 15	8
16 - 21	7
22 - 26	6
27 - 32	5
33 - 38	4
39 - 44	3
45 - 51	2
52 - 57	1
≥ 58	0

SLOPE POSITION	TPMI POSITION VALUE
Bottom	20
Lower	15
Middle	10
Upper	5
Crest	0

$$\text{TOTAL TPMI VALUE} = (\text{TPMI}_{aspect} + \text{TPMI}_{slope} + \text{TPMI}_{position})$$

*Also use TPMI_{aspect} = 10 if site is not a ridge crest and slope ≤ 5%.

that exhibited dwarf mistletoe infestations of Douglas-fir, were extracted from the dBase III Plus database and loaded into CHART. CHART calculated basic descriptive statistics on any loaded data, providing mean, median, standard deviation, and range information where desired. Sample distributions of phenomena of interest were graphed along gradients of elevation and moisture status in CHART, allowing direct inspection for patterns.

The multiple attribute database in MOSS allowed sampled phenomena to be visually displayed as point maps and examined for spatial patterns. Point maps were overlayed with other maps to check for patterns of overlap. Displayed maps were printed on a variety of media as needed.

FIRE SCARS

A scar index, indicating the relative importance of fire events in a specific year (Swetnam - in press), was calculated for all fire years recorded in the fire scar database. The scar index =

$$\frac{\text{(\# of trees scarred in year Z)}}{\text{(\# of fire scar susceptible trees alive in year Z)}} \times 100$$

where fire scar susceptible trees are defined as trees previously scarred by fire (Romme 1980). **Sample depth** is used as a synonym for (# of fire scar susceptible trees alive in year Z) in this dissertation. Scar indices will be high in years with widespread fires, and may exceed 100% during fire years which scar many trees for the first time. Tony Caprio of the Tree-Ring Lab wrote dBase III Plus programs to calculate scar indices and sample depths through time for each of the

five sample groups and for all groups combined; these data and associated graphs were included in the Tree-Ring Lab's final report to BNM (Caprio et al 1988).

I analyzed the fire scar database provided by the Tree-Ring Lab to develop a fire history for the Frijoles watershed. Descriptive statistics were calculated for a variety of scar-related values of interest using dBase III Plus and CHART; some of these results are presented in Tables 4-1 and 4-4 through 4-11. Tables 4-2 and 4-3 are adapted from Caprio et al (1988). I calculated numerous mean fire interval (MFI) values, where each MFI was defined as the "arithmetic average of all fire intervals (years between successive fires) determined in a designated area during a designated time period" (Romme 1980). I calculated MFI's for each sample group and for all groups combined for a number of different time periods. In addition I defined a variety of minimum scar index values which were used as thresholds for calculating the MFI's - i.e., if a threshold scar index $\geq$ 25 was specified only years with an index $\geq$ 25 were used to calculate fire intervals. MFI values generally increase as threshold scar indices are increased, as high scar index values restrict the fire years under consideration to those which record widespread fires. The procedures and tables used to calculate the Wilcoxon rank sum and chi-square tests came from Chiang et al (1984).

LANDSCAPE COVER-TYPE AND ECOSYSTEM PATCH MAPS

The MOSS GIS provided map unit area and perimeter values for individual cover-types in the polygon maps of landscape cover-type and ecosystem patches. A patch dissection index, PDI (Sharpe et al 1981),

was calculated where

$$PDI = \frac{\text{perimeter}}{(2 \times [\text{area} \times 3.14159]^{0.5})}$$

A landscape diversity index, H (Romme 1982), was computed with

$$H = \text{the summation as } k \text{ ranges from 1 to } m \text{ of: } (P_k) \times \ln(P_k)$$

where P_k = the proportion of the landscape in cover-type k, and m is the total number of cover-types present in the landscape. A landscape dominance index, D (O'Neill et al 1988), was calculated as $D = H_{max} - H$ where $H_{max} = \ln(m)$. Relative landscape evenness, E (Romme 1982), was computed as $E = (H/H_{max})$. Cover-type categories were combined to examine the affect of changes in m upon these three indices.

ROAD NETWORKS

The MOSS GIS provided total lengths for each type of road. Each road type map was overlayed with the land ownership map of the same 187,858 ha portion of the Jemez Mountains to determine road length by road type and land ownership category. I used data on road surface width provided by the SFNF (B. Simms, SFNF hydrologist). Paved roads were estimated to have a mean width of 7.93 m, resulting in 0.793 ha/km of paved road surface. Improved road width was 5.49 m, with 0.549 ha/km of improved road surface. Dirt road width was 4.27 m, with 0.427 ha/km of dirt road surface. Primitive road width was 3.05 m, resulting in 0.305 ha/km of primitive road surface. I used the primitive road

width values for the 1935 railroad bed surface. These road surface area estimates are quite conservative as they include travelled road surfaces only, excluding shoulders, cut and fill slopes, ditches, and cleared right-of-way.

CHAPTER IV. RESULTS AND DISCUSSION

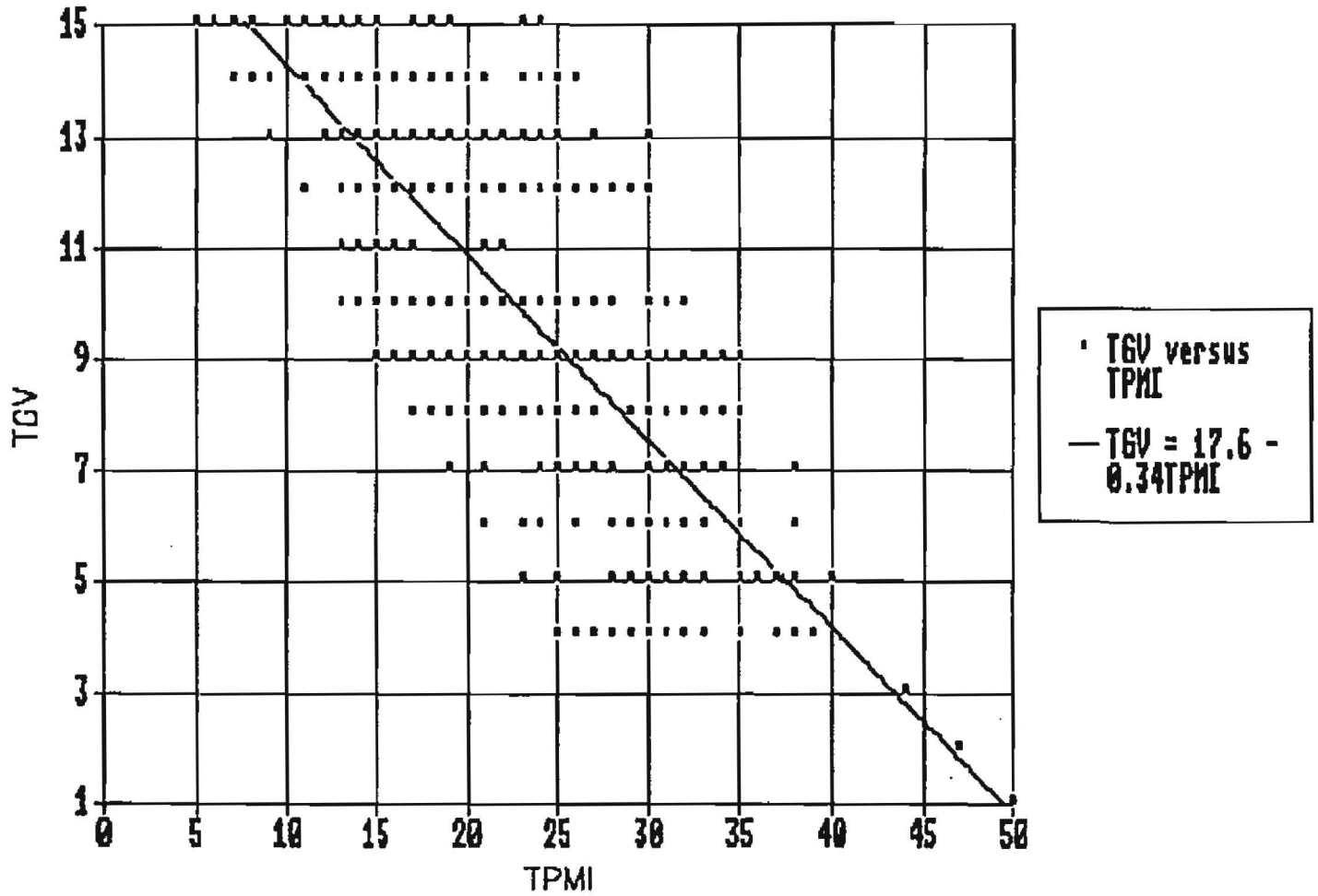
A. FRIJOLES WATERSHED SITE CHARACTERISTICS

The Rito de los Frijoles watershed extends southeast from 3109 m at the summit of Cerro Grande on the caldera rim to 1630 m at its junction with the Rio Grande. Its perrennial stream runs in a 150 to 250 m deep canyon, cut into the tuff of the Sierra de los Valles and Pajarito Plateau, for most of its 24 km length. This watershed comprises about half of the 13,248 hectare Bandelier National Monument, with the rest of the Monument primarily composed of the lower, Pajarito Plateau portions of several other watersheds adjacent to the south Figure 3-2). The site characteristics of the 969 field points sampled in and adjacent to the Frijoles watershed are described here to provide a framework for subsequent discussions of vegetation and landscape patterns.

Figure 4-1 displays the strong correlation (r^2 = 0.796) between topographic gradient value (TGV) and topographic potential moisture index (TPMI) for all sample points. The TGV of sample points has a range of 1-15, median of 9, mean of 8.5, and a standard deviation of 4.3, while TPMI values range from 5-50, with a median of 25, mean of 27.0, and a standard deviation of 11.4. By definition 5 is the lowest possible TPMI value. Only TPMI data will be discussed further here, because: 1) TPMI provides a more sensitive indication of site moisture status than TGV due to its addition of percent slope and slope position information to the aspect factor considered by both indices (cf. Parker 1980); and 2) TPMI provides a broader spread of site distributions that is advantageous for graphically displaying landscape patterns.

The elevation of each sample site is graphed as a function of TPMI

FIGURE 4-1. TGV versus TPMI for all sample points.



in Figure 4-2. Sample points range in elevation from 1650 - 3076 m, with a median of 2236 m, mean of 2308.5 m, and a standard deviation of 326.9 m. Steep slopes (Figure 4-3) preclude high TPMI values at the highest elevations, while steep slopes and southerly aspects (Figure 4-4) prevent high TPMI values at the lowest elevations. A variety of other landscape features will be similarly graphed, and these distributions may be compared to this basic distribution of all sampled points.

Elevation as a function of percent slope is displayed in Figure 4-3 for all sample points with slopes $\leq$ 120%. The slopes of sample sites range from 0 - 250%, with a median of 16%, mean of 27.37%, and a standard deviation of 28.9%. Most points have moderate slopes of less than 20%, with extremely steep slopes found on near-vertical canyon and valley walls. The gap in the distribution of points with slopes < 20% observed around 2360 m reflects the escarpment of the Pajarito Fault-this escarpment separates the relatively level surface of the Pajarito Plateau below from the gentle slopes of Apache Mesa and Mesa del Rito above. The high elevation portions of the watershed along the caldera rim lack both canyon walls and flat sites, and thus display steep but not extreme slopes. The lowest elevations of the watershed are dominated by the extreme slopes of the walls of White Rock Canyon.

Figure 4-4 displays the elevation of all sample sites as a function of aspect. Sample point aspects range from 0° - 360°, with a median of 150°, mean of 149.8°, and a standard deviation of 83.1°. The mean aspect reflects the southeast orientation of the Frijoles watershed as it drains from the caldera rim to the Rio Grande.

The percent bare rock of all sample sites is graphed as a function

FIGURE 4-2. Elevation versus TPMI for all sample points.

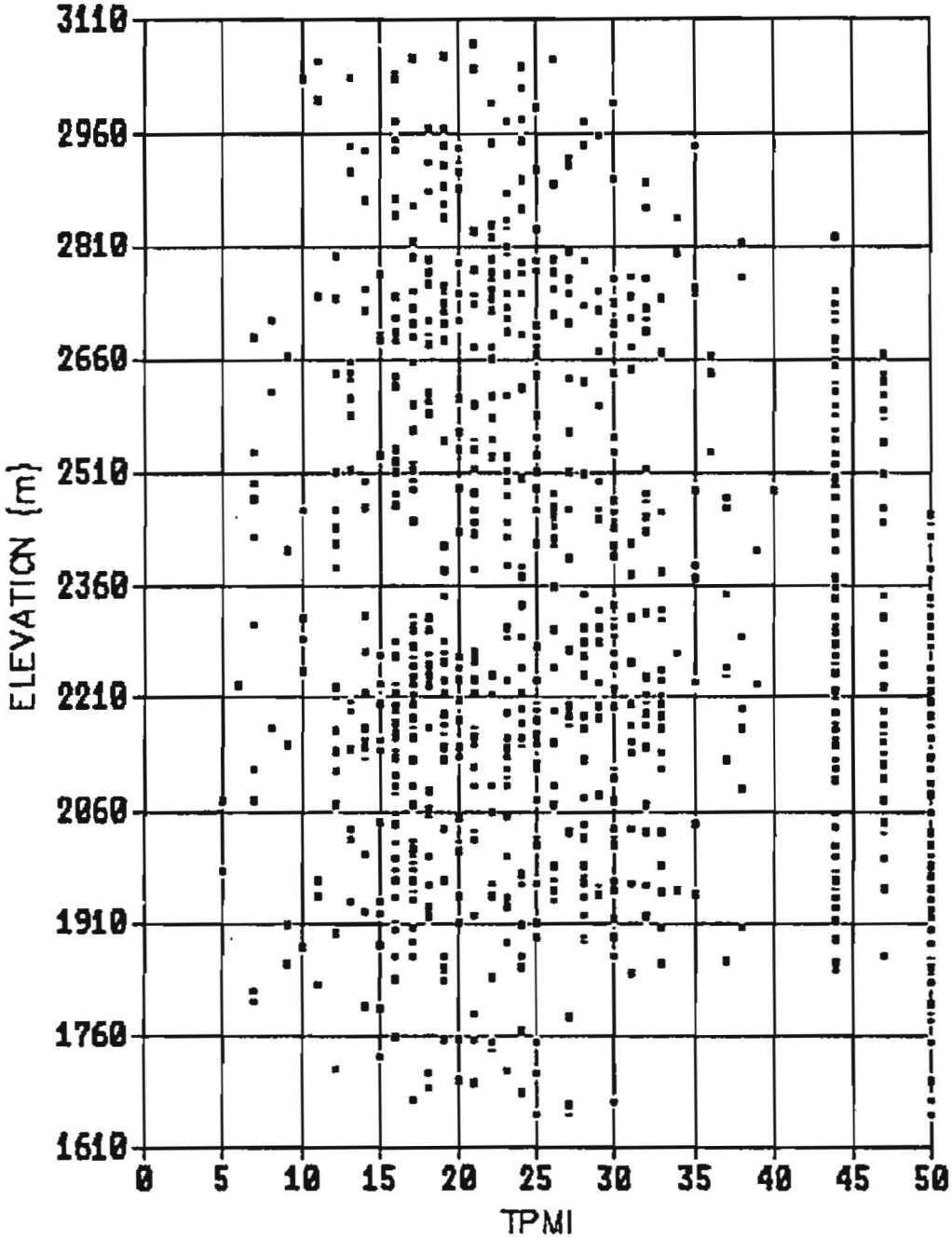


FIGURE 4-3. Elevation versus percent slope for all sample points.

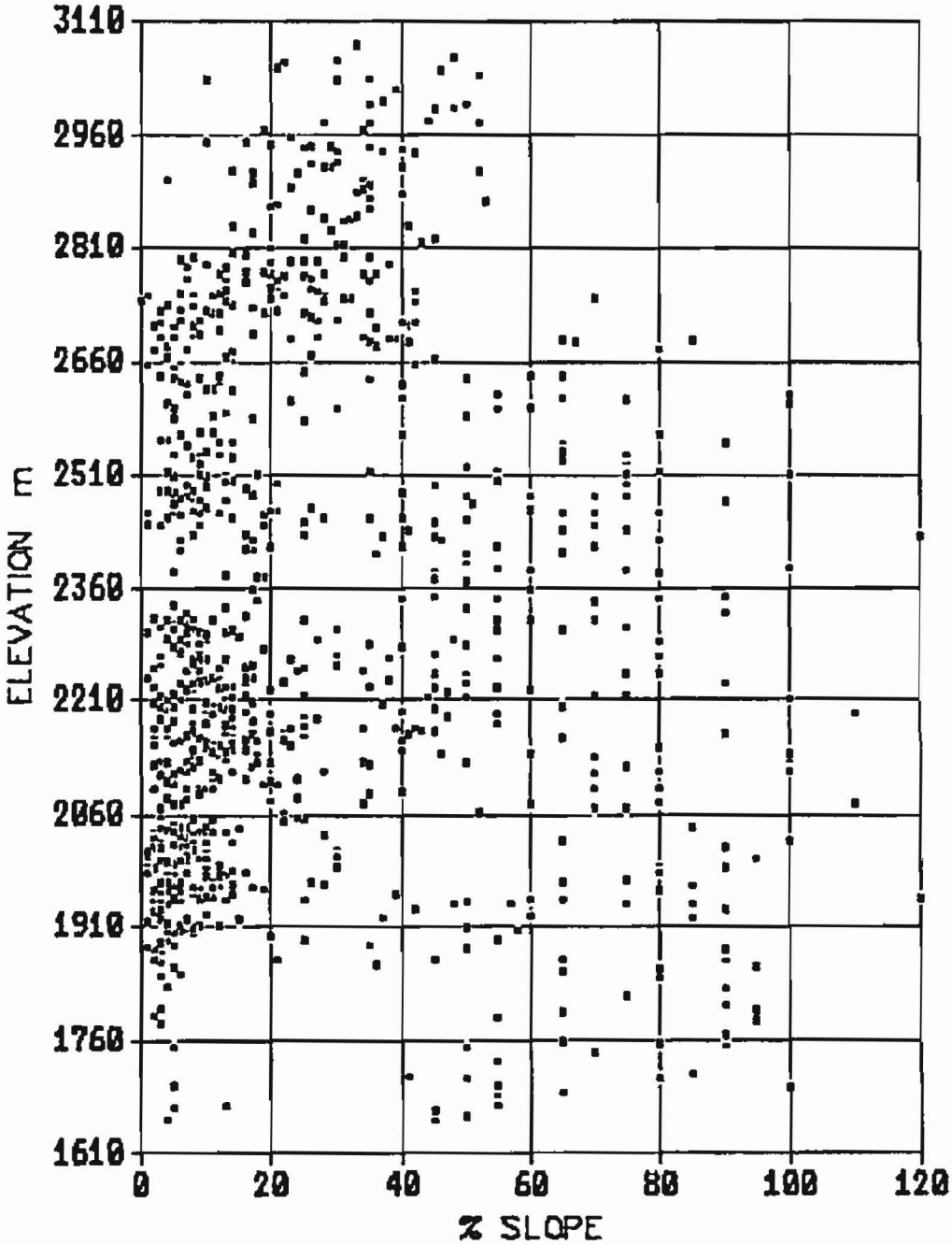


FIGURE 4-4. Elevation versus aspect for all sample points.

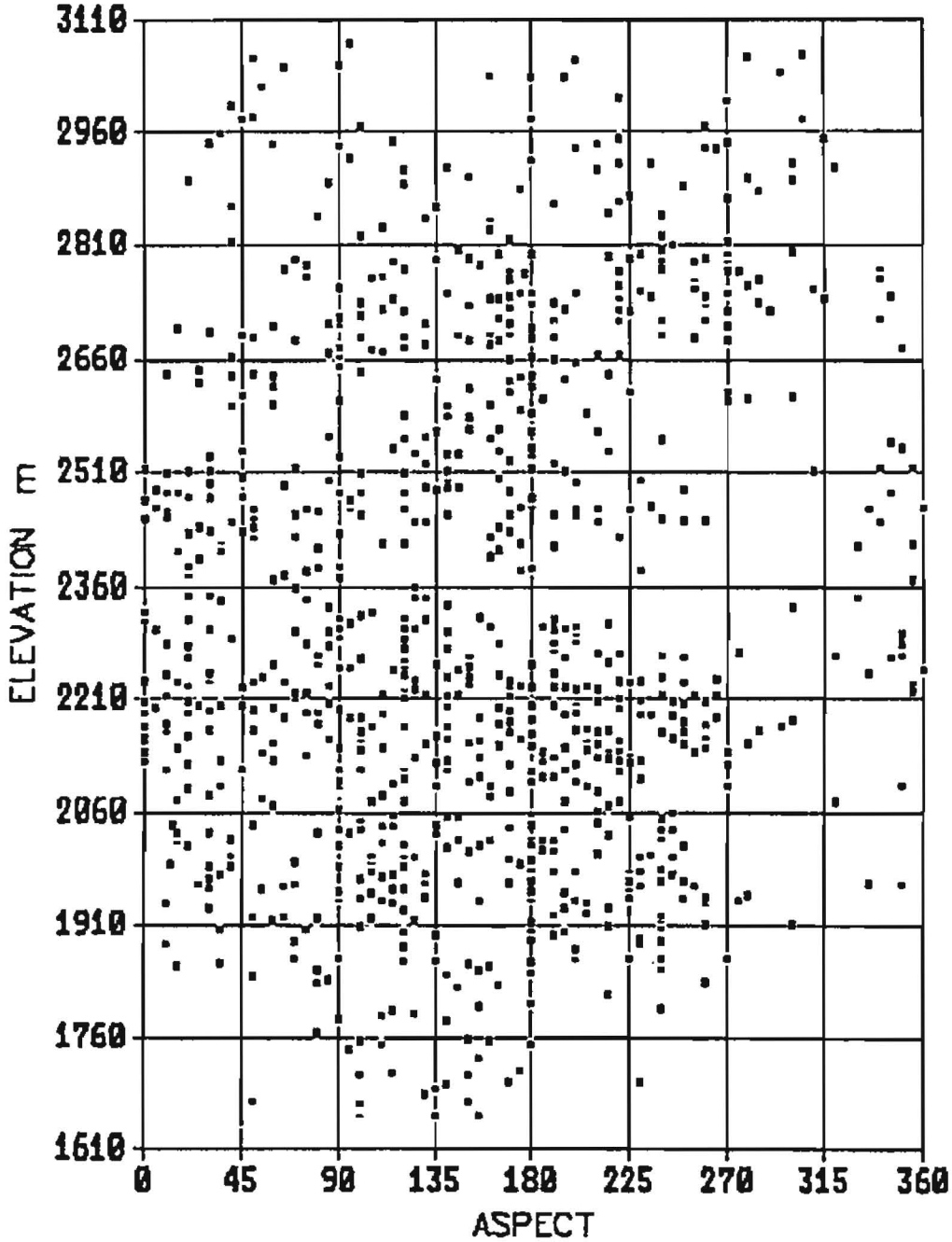
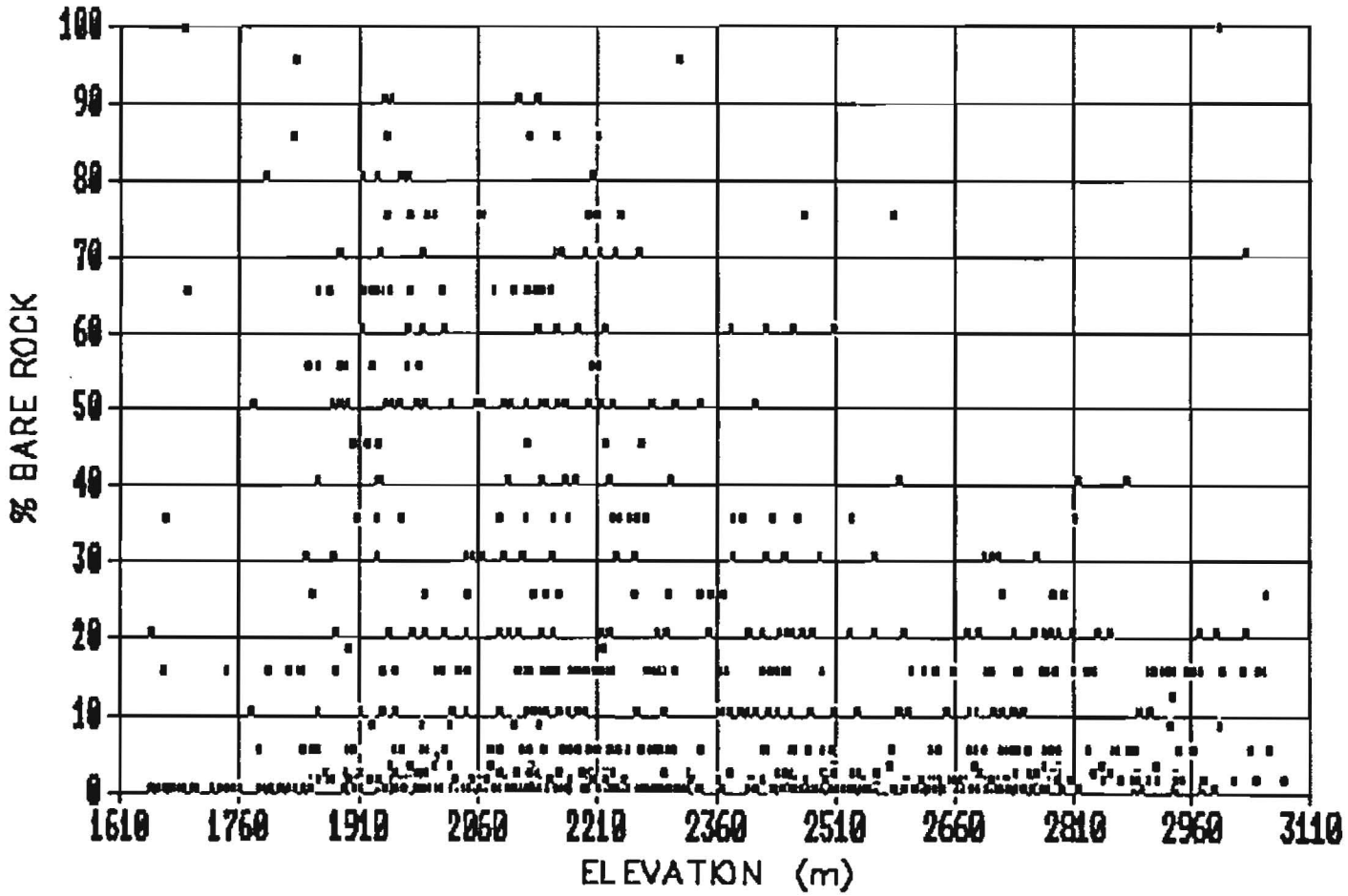


FIGURE 4-5. Percent bare rock versus elevation for all sample points.



of elevation in Figure 4-5. The percent bare rock ranges from 0 - 99%, with a median of 3%, mean of 13.9%, and a standard deviation of 21.6%. The concentration of values > 25% bare rock at mid to lower elevations reflects the steep, rocky slopes of the canyon walls as well as the thinner soils and erosion problems of the lower Pajarito Plateau.

B. DISTURBANCE REGIMES

Following White and Pickett (1985), disturbance is defined here as "any relatively discrete event in time that disrupts ecosystem, community, or population structure and changes resources, substrate ability, or the physical environment". Disturbances in the Jemez Mountains can be roughly divided into two categories, "natural" and anthropogenic. Natural disturbances would continue to occur in the absence of people, while anthropogenic disturbances depend upon human agency to occur. I recognize that classification into natural versus anthropogenic is ultimately a false dichotomy, due to the currently pervasive interactions of human activities with natural processes and the fact that human beings evolved from and continue to be a part of nature; I use such a classification here only to highlight new human influences on the landscape.

Distinguishing a disturbance event from "normal" environmental fluctuation requires a spatial and temporal frame of reference. My discussion of natural disturbance regimes will emphasize events that can alter landscape patterns and which appear as pulses when considered over a time scale spanning one to several hundred years. In contrast, many of the anthropogenic disturbances treated here are chronic and cumulative rather than relatively discrete events - nonetheless these

human activities have caused significant changes in local landscape patterns and processes. Still, identification of a disturbance remains somewhat ambiguous. For example, are high-frequency, low-intensity surface fires a disturbance to a forest that has experienced similar fires for centuries, or is an alteration of that fire regime by direct suppression a disturbance? Both types of "event" will be considered as disturbances here. Local natural disturbances include precipitation variability, fires, insect outbreaks, extreme weather events, floods, and volcanic activity. Local anthropogenic disturbances include livestock grazing, altered fire regimes, biocide use, biotic extirpations and introductions, forest cutting, development of roads and dams, and various forms of pollution.

The spatial and temporal characteristics of disturbances define disturbance regimes. Disturbance regime descriptors include spatial distribution and scale, frequency, and magnitude. Frequent interactions between disturbances complicates characterization of disturbance regimes (White 1979). While detailed disturbance descriptions are largely fragmentary or non-existent for the Jemez Mountains, the best available information is compiled here to describe the most important local disturbance regimes.

NATURAL DISTURBANCES

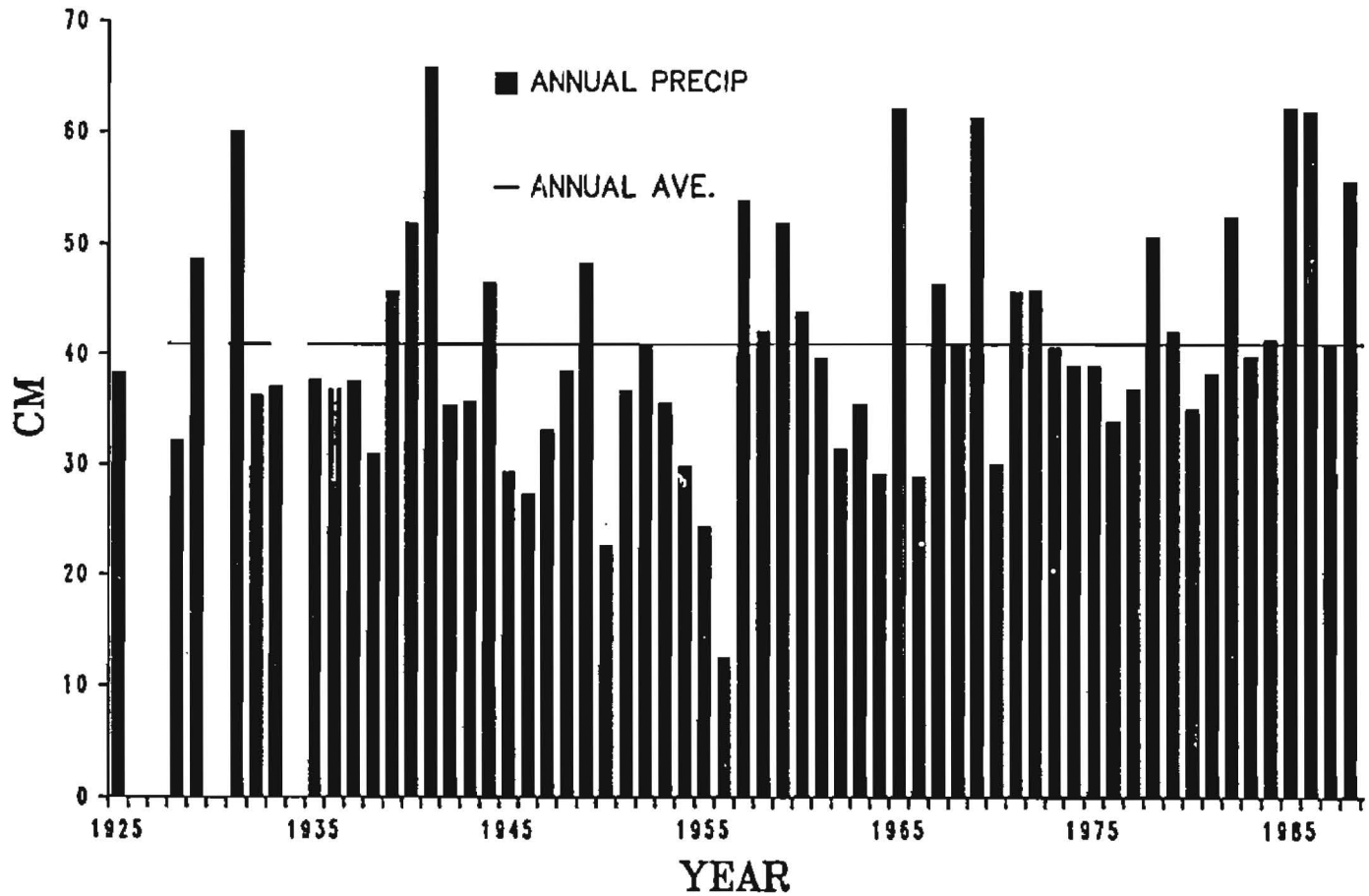
Precipitation Variability

Inter-annual precipitation variability appears as a pulsed disturbance regime when viewed on a time scale of decades, centuries, or millenia. During drought periods available moisture often becomes a limiting factor for primary plant productivity and reproduction in the

Southwest, with repercussions for entire landscapes. Drought often interacts directly with other landscape-wide disturbance regimes, such as fire and some insect outbreaks. Unusually wet years also have significant landscape impacts through interactions with biota and disturbance regimes. Precipitation variability was likely one determinant of prehistoric settlement and migration patterns in the Southwest (Euler et al 1979), and thus modulated prehistoric human landscape disturbances.

Precipitation variability in the Jemez Mountains is related to quasi-periodic El Niño events which bring wetter springs and summers to the Southwest about every 3-5 years (Andrade and Sellers 1988). Persistent clusters of consecutive dry and wet years also occur. Post-1924 droughts recorded at Bandelier include a moderate drought from 1932-1938, another moderate drought during 1942-1943/1945-1947, and an extreme drought during 1950-1951/1953-1956 (Figure 4-6). The driest year on record at Bandelier is 1956 when 12.5 cm of precipitation were recorded, versus the 1925-1987 average of 40.7 cm. At a coarser level of resolution, decadal maps of local tree-ring variability from Paliza Canyon in the southern Jemez Mountains (Dean and Robinson 1977) indicate 27 decades probably dominated by drought conditions between 680 A.D. and 1970 (based upon decadal tree growth less than or equal to one standard deviation below the overall mean). The 1950's had by far the lowest decadal growth index in the 1290 year record, 4.1 standard deviations below the mean; another extreme decade was the 1580's, at 3.1 standard deviations below the mean. Similar tree-ring growth patterns have been found at two other sites in the southwestern Jemez Mts. (T. Swetnam, Tree-Ring Lab - unpublished data). Annual and 5-year

FIGURE 4-6. Annual precipitation at Bandelier National Monument



running average graphs of the post-1500 portion of the Paliza Canyon tree-ring chronology (Dean and Robinson 1978) are shown in Figure 4-7. These graphs record relative tree growth, with a mean annual index value of 1.0 and a standard deviation of 0.446. Small tree-ring index values indicate low tree growth and probable drought conditions.

Clusters of years with above average precipitation have been recorded at Bandelier during 1939-1941, 1957-1959, 1967-1969, and from 1984-1988 (Figure 4-6). A significant, recent, wet period occurred locally from 1905-1921 (with some drier years interspersed), based upon unpublished tree-ring data from the western Jemez Mountains (T. Swetnam - personal communication) and Los Alamos weather records (B. Bowen, LANL meteorologist - personal communication); a similar generally wet period was found by Rose et al (1981) at nearby Arroyo Hondo, but only portions of it are confirmed by Dean and Robinson (1978) from the southern Jemez Mountains (Figure 4-7). Note that tree-rings are generally better predictors of dry years than wet periods in this region, as other factors may limit tree growth in moist years (Rose et al 1981). The wettest year on record at Bandelier is 1941, when 65.9 cm of precitation fell. Tree-rings indicate 28 probable decades of above average precipitation (decadal tree growth greater than or equal to one standard deviation above the mean) since 680 A.D. in the Jemez Mountains, with the highest decadal growth index recorded for the 1830's at 3.1 standard deviations above the mean (Dean and Robinson 1977). The 1925-1988 mean precipitation has been exceeded at Bandelier 8 of the last 11 years (1978-1988).

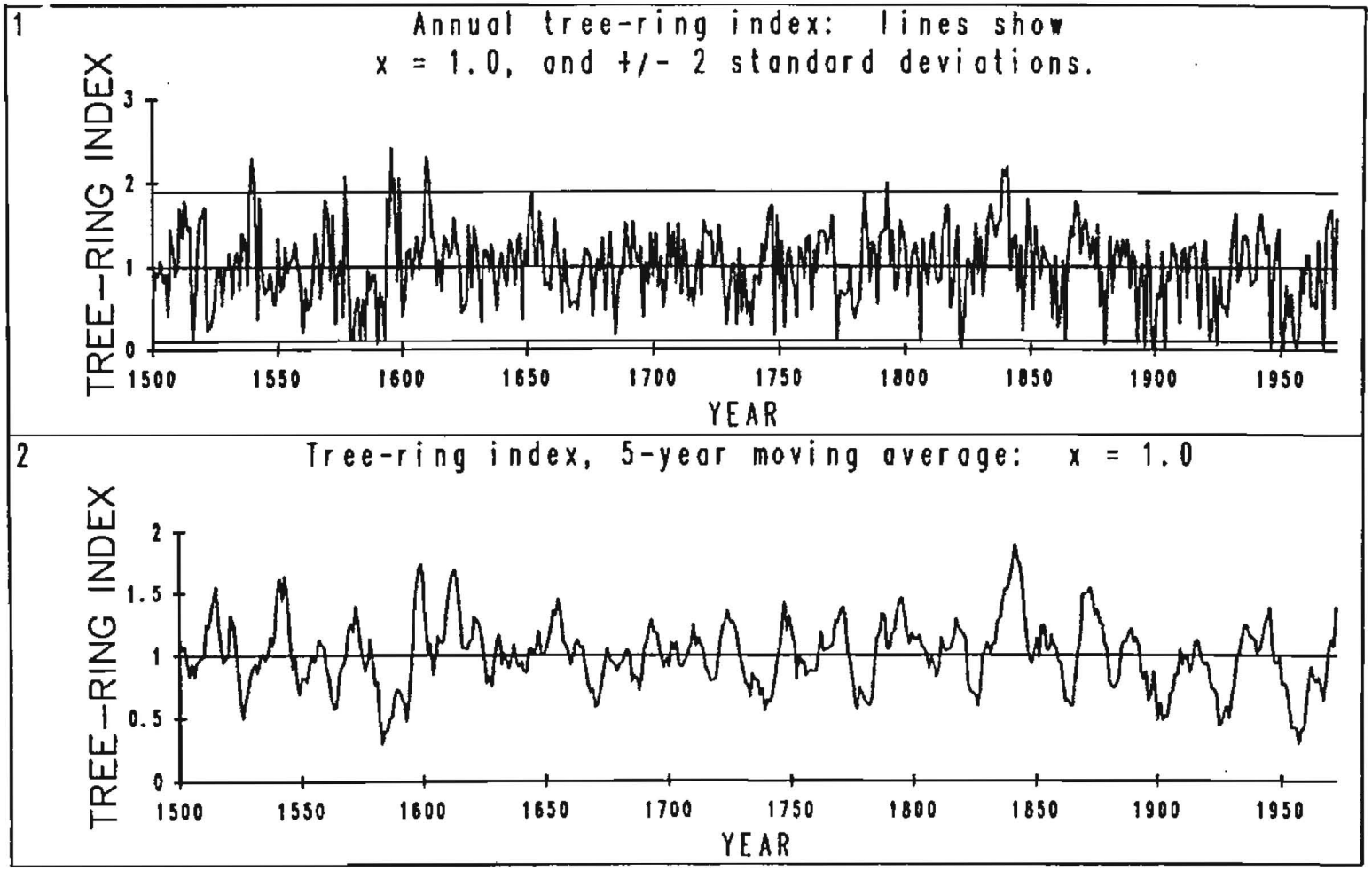


Figure 4-7. Annual and 5-year average tree-ring index values for Paliza Canyon in the Jemez Mountains.

Fire

Fire disturbance regimes affect ecosystems at scales ranging from site-level nutrient cycling (Woodmansee and Wallach 1981) to structuring of landscape vegetation patterns (Romme 1982, Franklin and Forman 1987). Fire is widely recognized as an integral and nearly ubiquitous element of forested landscapes in the Southwest (Komarek 1969, Swetnam - in press). Fire has apparently been an important process in the Jemez Mountains landscape for at least several thousand years - 21% of the 3.76 m length of a pollen core extracted from Alamo Bog in the central caldera is composed of charcoal lenses dating back to 4600 B.P. (Stearns 1981). In 1977 the La Mesa Fire burned 6184 ha centered on the Frijoles watershed, stimulating increased research on local fire history. Foxx and Potter (1978, 1984) examined the fire ecology of Bandelier ponderosa pine forests, while Potter and Foxx (1981) and Allen (1984-b) conducted limited fire history research in mixed conifer forests of the Frijoles watershed. This dissertation greatly extends previous research to document the fire history of most of the Frijoles watershed, demonstrating the central role of fire in shaping the past and present landscape of the Jemez Mountains.

Fire History of the Frijoles Watershed - General Characteristics of Sampled Fire Scars

Field sampling of the Frijoles watershed found old fire scars that predate the La Mesa Fire at 311 sites, with La Mesa Fire scars found at 63 points (Figure 4-8). The sample locations of the 75 trees, in five groups, with crossdated fire scars from the Frijoles watershed are mapped in Figure 3-3. Table 4-1 lists summary data for the five sample

FIGURE 4-8. Fire scar distribution of all sample points.

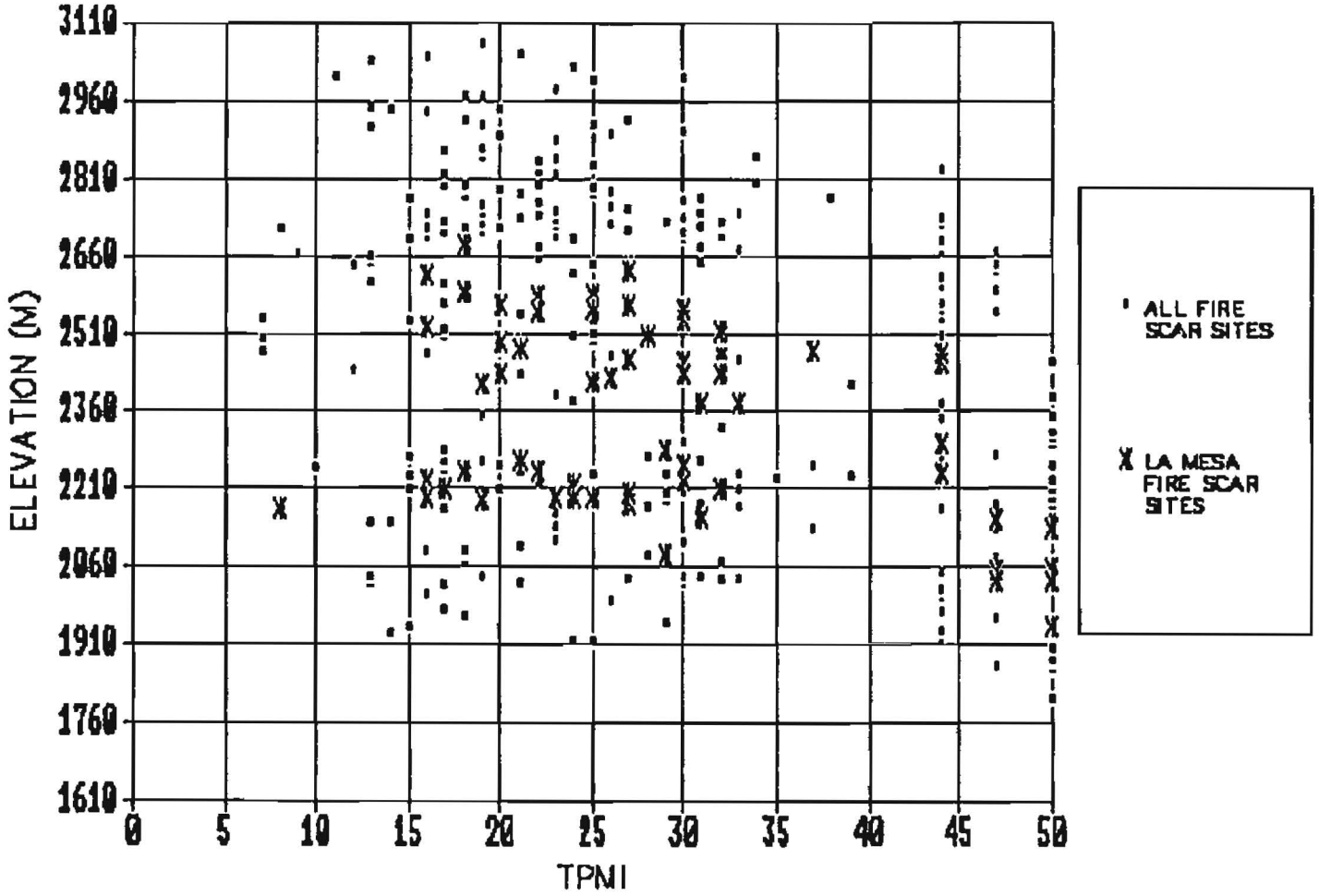


TABLE 4-1. Summary of fire scar sampling data by group.

	GROUP 1	2	3	4	5	Total
Veg. type	PPPJ	RIMC	PIPO/PPMC	PPMC/MC	PIPO	-
# trees	9	13	18	23	12	75
% trees	12	17.3	24	30.7	16	100
# scars	109	249	307	406	89	1160
% scars	9.4	21.5	26.5	35.0	7.7	100
elevation (m):						
mean	2032	2337	2520	2787	2218	2463
std. dev.	10.5	112.5	73.1	76.1	17.5	271.8
median	2031	2339	2516	2772	2211	2496
range	2016-2048	2172-2577	2416-2698	2681-2934	2202-2246	2016-2934

groups, while Table 4-2 lists summary data for each sampled tree. The groups sample five different overstory vegetation types, namely: open forests of ponderosa pine and piñon-juniper (PPPJ), riparian mixed conifer forest (RIMC), the transition from ponderosa pine to ponderosa pine/mixed conifer forest (PIPO/PPMC), ponderosa pine/mixed conifer grading into mixed conifer forest (PPMC/MC), and ponderosa pine forest (PIPO). Sampled trees grew between 2016 and 2934 m elevation. A complete listing of the data associated with each of the 1160 dated scars is contained in Appendix 1 of Caprio et al (1988).

The mean number of fire scars/tree (i.e. scars representing different fire years) for the entire data set (N = 75) is 10.25 scars/tree, with a standard deviation of 4.75, median of 10, and range from 1 to 31; if only the trees I sampled are considered (N = 64) the mean is 11.09 scars/tree, with a standard deviation of 4.52, median of

TABLE 4-2. Summary of data by tree. Gives tree sample #, group #, tree species, overstory vegetation type, date and type of inside and outside rings observed, number of scars dated on right and left sides of each sample section, and total number of different scar dates on a sample. For inside dates P = pith, I = innermost ring present, and + = innermost dateable ring present. For outside dates B = at bark, L = last ring present, and + = outermost dateable ring present. Tree species abbreviation contained in Table 3-1. Tree #'s 89-99 are from earlier collections (Robinson 1978, Foxx and Potter 1984).

TREE #	GROUP	TREE SP.	VEG. TYPE	INSIDE DATE		OUTSIDE DATE		# OF SCARS RIGHT	LEFT	TOTAL*
1	4	PIFL	MC	1624	I	1986	B	7	0	7
2	4	ABCO	MC	1820	I	1987	B	5	2	6
5	1	PIPO	PPPJ	1627	+	1987	B	11	3	11
6	1	PIPO	PPPJ	1650	P	1987	B	5	4	5
7	4	PIPO	PPMC	1617	P	1822	L	9	11	13
8	4	PIPO	PPMC	1587	P	1978	+	10	12	14
9	4	PIPO	PPMC	1783	P	1987	B	12	11	12
11	3	PIFL	PPMC	1557	P	1987	B	13	12	13
12	3	PIPO	PPMC	1775	P	1985	B	7	0	7
13	1	PIPO	PPPJ	1612	P	1987	B	12	5	12
14	1	PIPO	PPPJ	1762	I	1954	B	5	6	6
15	1	PIPO	PPPJ	1627	P	1987	B	8	8	8
16	1	PIPO	PPPJ	1660	P	1987	B	3	3	4
17	1	PIPO	PPPJ	1614	P	1987	B	13	0	13
19	1	PIPO	PPPJ	1625	P	1987	B	3	3	3
20	3	PIPO	PPMC	1783	P	1897	B	13	11	13
21	3	PIPO	PIPO	1649	P	1907	L	9	1	9
22	3	PIPO	PIPO	1665	I	1973	L	10	11	14
23	3	PIPO	PIPO	1615	P	1961	L	11	0	11
24	3	PIPO	PPMC	1827	P	1987	B	6	3	6
25	3	PIPO	PPMC	1595	-	1815	+	3	4	6
26	5	PIPO	PIPO	1637	P	1965	+	14	16	17
28	4	PIPO	MC	1655	P	1987	B	6	13	13
29	2	PIPO	MC	1649	P	1827	B	8	7	8
30	2	PIPO	MC	1648	P	1987	B	8	9	11
31	4	PIPO	MC	1539	P	1935	B	17	0	17
32	2	PIPO	RIMC	1680	-	1860	+	7	3	7
33	2	PIPO	RIMC	1659	P	1970	B	9	9	11
34	2	PIPO	RIMC	1706	P	1949	B	9	7	10
36	2	PIPO	RIMC	1580	P	1880	L	13	0	13
37	2	PIPO	RIMC	1644	P	1987	B	11	13	15
38	2	PIPO	MC	1595	P	1987	B	12	3	12
39	2	PIPO	RIMC	1643	P	1987	B	16	6	16
40	2	PIPO	RIMC	1644	I	1919	+	13	8	13
42	2	PIPO	RIMC	1705	P	1936	B	8	9	11
43	2	PIPO	RIMC	1642	P	1987	B	16	14	18
45	2	PIPO	RIMC	1650	P	1987	B	13	9	15
46	3	PIFL	PPMC	1459	I	1828	L	7	8	11
47	3	PIFL	PPMC	1543	P	1987	B	13	10	14
48	3	PIPO	PPMC	1541	P	1858	L	11	4	12

TABLE 4-2. Continued.

TREE #	GROUP	TREE SP.	VEG. TYPE	INSIDE DATE		OUTSIDE DATE		# OF SCARS RIGHT	LEFT	TOTAL*
49	3	PSME	PPMC	1561	P	1977	B	23	20	31
50	3	PIPO	PIPO	1574	P	1979	B	13	4	13
51	3	PIPO	PIPO	1627	P	1842	L	12	13	14
52	4	PSME	PPMC	1627	P	1987	B	17	14	19
53	4	PIPO	PPMC	1627	P	1987	B	9	6	9
54	4	PIPO	PPMC	1627	P	1933	B	10	7	10
55	4	PIFL	MC	1627	P	1987	B	5	6	7
56	4	PIFL	MC	1627	P	1873	L	8	7	8
57	4	PIPO	MC	1627	P	1975	+	9	0	9
58	4	PIPO	MC	1627	P	1902	L	8	10	11
59	4	PIPO	MC	1627	P	1941	B	8	7	8
60	4	PIPO	MC	1625	P	1806	L	13	14	18
61	4	PIPO	PPMC	1542	P	1958	B	5	11	11
64	4	PIPO	PIPO	1721	P	1887	L	4	3	5
66	3	PIPO	PPMC	1706	P	1987	B	8	8	10
67	3	PIPO	PIPO	1652	P	1873	L	8	3	8
68	3	PIPO	PIPO	1577	P	1797	L	9	9	9
69	1	PIPO	PPPJ	1655	I	1955	+	6	6	7
71	3	PIFL	PPMC	1755	I	1964	B	5	5	6
72	4	PIPO	PPMC	1686	P	1873	L	7	6	7
73	4	PIPO	PPMC	1646	P	1948	B	10	5	12
74	4	PIPO	PPMC	1588	P	1988	B	18	13	18
75	4	PIPO	PPMC	1588	P	1800	+	10	6	10
76	4	PIPO	PPMC	1652	P	1917	B	11	8	13
89	5	PIPO	PIPO	1910		1976	B	-	-	2
90	5	PIPO	PIPO	1913		1976	B	-	-	1
91	5	PIPO	PIPO	1774	P	1977	B	-	-	3
92	5	PIPO	PIPO	1770	P	1977	B	-	-	4
93	5	PIPO	PIPO	1774	P	1977	B	-	-	6
94	5	PIPO	PIPO	1768	P	1977	B	-	-	10
95	5	PIPO	PIPO	1780	P	1977	B	-	-	8
96	5	PIPO	PIPO	1767	P	1977	B	-	-	6
97	5	PIPO	PIPO	1774	P	1977	B	-	-	5
98	5	PIPO	PIPO	1773	P	1977	B	-	-	6
99	5	PIPO	PIPO	1777	P	1907	V	-	-	8

* Total number of scars is the total number of different scars on both the right and left sides of a sample section. Number given for either right or left may be for more than one section of the same side.

11, and range of 3 to 31. These trees apparently record generally low-intensity surface fires that sometimes ignited the pitchy edges of exposed, healing scar surfaces (cf. McBride 1983); high-intensity fires would have consumed much of the scarred area and killed the trees

rather than allowing them to persist and record multiple scars.

Most of the sampled trees are ponderosa pine, as this abundant species often develops large, resinous "catfaces" that record multiple fires and resist decay. Limber pine commonly develops similar scars, but was sampled less often since its relative scarceness translates into fewer specimens with exceptional scar records. However, limber pine is probably the longest-lived tree in the Jemez Mountains, and thus many of the earliest fire dates in this chronology came from limber pine. White fir and Douglas-fir are typically less effective recorders of multiple fires as their catfaces are less resinous than those found in the pines, and their scarred area usually leads to internal decay that may obscure the scar record and shorten the life of the tree (and thus the scar record). Yet the only two Douglas-fir trees sampled (#'s 49 and 52) had more individual fire scars than any other sampled trees. Both of these exceptional Douglas-fir trees displayed resinous, undecayed, fire-scarred cross-sections, although #49 was a small snag growing on an exposed, rocky site while #52 was a huge, vigorous tree growing on an unremarkable site.

The age at which a sampled tree was first scarred by a fire could be determined for 64 samples. The mean age when initially scarred was 43.66 years, with a standard deviation of 39.19, median of 31, and range of 8 to 169 years. Several high values skew the mean age, as 73% of the samples scarred by age 40, and 55% scarred between the age of 20 and 40 years. The age when first scarred in the Frijoles watershed is lower than the 96 years (std. dev. = 56.5) found for ponderosa pine in the Sierra Nevada by Kilgore and Taylor (1979) and the 76.8 years recorded by Madany and West (1983) for ponderosa pine in Zion National

Park. The young age at which trees in the Frijoles watershed typically first scarred is another indication that low-intensity surface fires were the dominant fire regime, as higher intensity fires would likely have killed such young trees. Note that these data slightly underestimate the true age to initial scarring, as sample cross-sections were typically collected at a height of 0.2 - 0.7 m.

Thirty-eight of the 64 dated specimens I collected consisted of snags, downed logs, stumps and other dead tree material. The highly resinous nature of trees that record multiple fires allows these trees to persist with little sign of decay for many years after the tree dies, especially if the tree avoids contact with the ground (and associated moisture) by standing as a snag. Charring of the external surface of the tree by subsequent fires, as well as the charred character of the catface itself, may also contribute to the preservation of the such specimens. The fire-scarred area in and around the catface itself is often extremely resistant to decay and may persist intact long after the rest of the tree has decayed around it. Twentieth century fire suppression has helped to protect these records of fire history from consumption by recent fires. Review of the outside dates seen in Table 4-2 shows that scarred trees can persist for well over 100 years, e.g. Tree #'s 7, 25, 29, 32, 46, 48, 51, 56, 60, 67, 68, 72, and 75. While most of these old snags lost portions of their outer wood due to physical weathering and/or charring by fires, such losses typically represent at most a few decades. Note that Tree #29 displays a bark date from 1829. Crossdating of such dead material allows the extension of this chronology farther back in time than most other Southwestern fire histories. For example, the earliest six dates

in this chronology, 1480 - 1544, come from a single limber pine snag (#46), and represent some of the oldest known fire dates from the Southwest. Most of the other fire dates from the 1500's also come from dead material or long-lived limber pines.

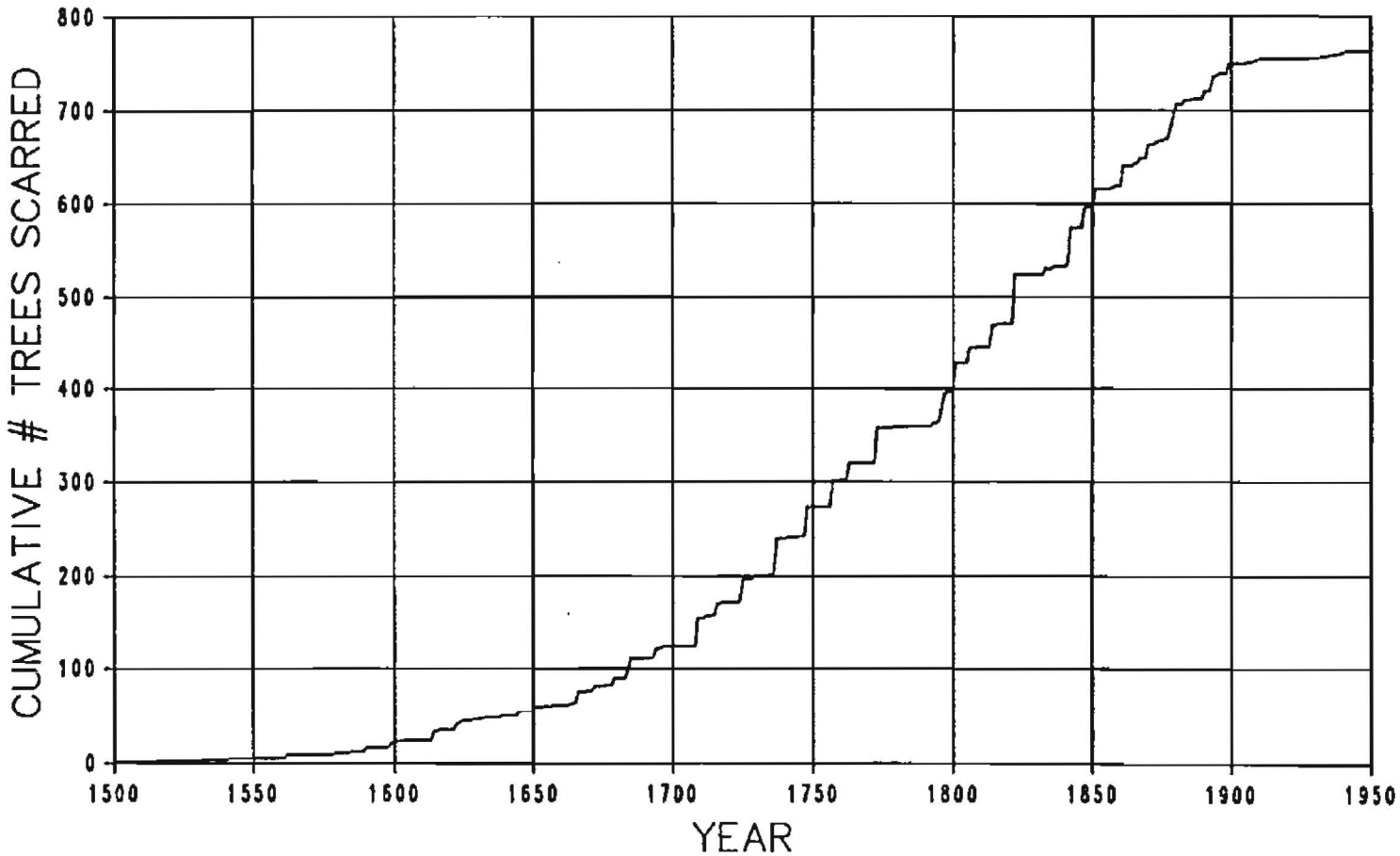
The cumulative number of fire scars through time is displayed in Figure 4-9. Of the 1160 total scars, 2 (0.2%) date to the 1400's, 25 (2.2%) date to the 1500's, 157 (13.5%) date to the 1600's, 424 (36.6%) date to the 1700's, 532 (45.9%) date to the 1800's, and 18 (1.6%) date to the 1900's. The increasing number of scars/century through the 1800's reflects the increasing number of previously scarred trees (sample depth) in the sample through time, while the abrupt decline in scars after 1899 reflects changes in the fire regime.

Table 4-3 shows the scar index ([# trees scarred X 100]/sample depth) for each group, and all groups combined, for all fire years. This table and associated scar index graphs (Figure 4-10) are slightly revised and updated from Caprio et al (1988). High scar index values indicate one or more widespread fires burned throughout the sample area in a given year, while low index values indicate more-restricted patchy fires, lower intensity fires, or simply the vagaries of whether individual trees scarred in a fire event (cf. Dieterich and Swetnam 1984). These fire history data are considered below for the entire Frijoles watershed, and then for each group separately.

Rito de los Frijoles watershed (all groups combined)

Mean fire intervals: The mean fire interval (MFI) for any site depends upon the time period (Romme 1980, Dieterich 1980-b) and threshold scar index value (Swetnam - in press) used in the

FIGURE 4-9. Cumulative annual distribution of sampled fire scars, 1500 - 1950.



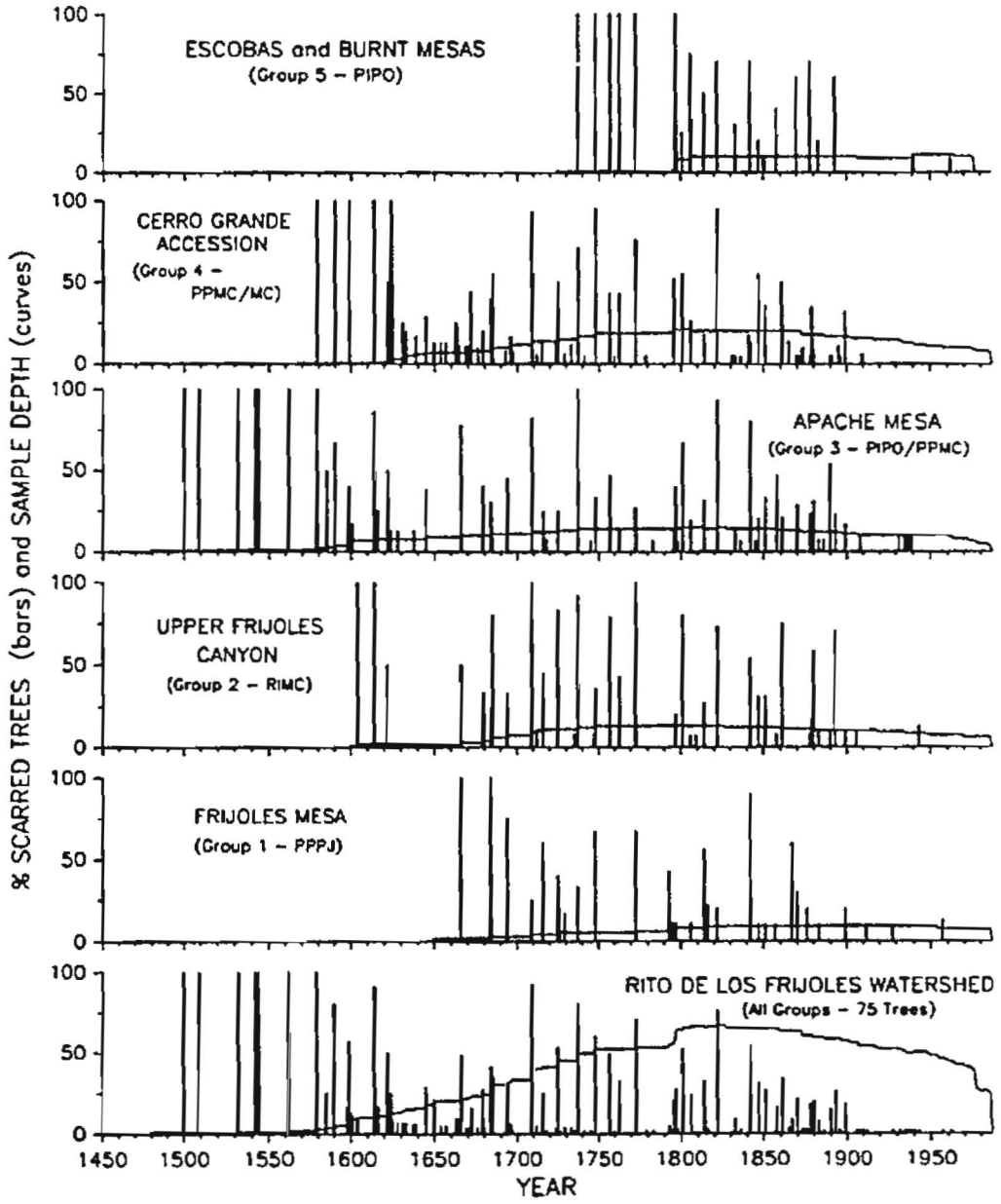


Figure 4-10. Fire-scar indices for each sample group in the Frijoles watershed.

TABLE 4-3. Scar index for each fire year for each sample group and all sites combined (ALL). First fire scar date for each group has a value of 0.

	GROUP							GROUP					
YEAR	1	2	3	4	5	ALL	YEAR	1	2	3	4	5	ALL
1480			0			0	1709	25	143	82	93		92
1500			100			100	1712		10		6		5
1509			100			100	1715				13		5
1532			100			100	1716	60	45	25			25
1542			100			100	1718			8			2
1544			100			100	1724				13		4
1562			100	0		300	1725	40	83	25	50	0	53
1579			100	100		100	1726	20					2
1585			50			25	1729	17			6		4
1590			67	100		80	1733				12		4
1598			25			17	1735		8				2
1599			40	100		57	1737	33	92	100	71	100	80
1600		0				13	1741				5		2
1601			17			11	1745			7			2
1604		100				10	1747		8				2
1614		100	86	100		91	1748	67	36	33	95	100	60
1616			25			1	1757		79	47	43	100	49
1622		50	50	50		50	1760				5		2
1623			13			8	1763		43	13	43	100	32
1624			13	100		25	1773	67	107	27	76	100	70
1625				50		8	1779				5		2
1628			13			7	1783			7			2
1631				25		7	1793	43					5
1633				20		7	1795	11					2
1638			13			6	1796			13	52		21
1639				17		6	1797	11	20	40		700	27
1645			38	29		29	1798			7		14	3
1650	0			13		21	1801		80	67	55	25	52
1654				13		5	1806	11	7	19	26	75	24
1657				13		5	1809		7				1
1663				25		9	1814	56	27	31	18	50	32
1664				22		9	1816	22					3
1665				11		4	1822	20	73	93	95	70	76
1666	100	50	78			48	1831				5		1
1669				11		4	1833			13	5	30	9
1671				11		4	1836			7	5		3
1672				44		16	1841				18		6
1676				10		4	1842	90	54	80	14	70	54
1679		33	40	20		27	1845			7			1
1680		33				4	1847	10	31	20	55	20	31
1684	100	25	30	40		41	1850					10	1
1685		80	9	55		35	1851	10	31	33	36	10	27
1693				8		3	1857	10					1
1694	75	33	45			30	1858		8	47		40	17
1696				17		6	1861		75	21	50		34
1697				8		3	1865				14		4

TABLE 4-3. Continued.

YEAR	GROUP 1	2	3	4	5	ALL	YEAR	GROUP 1	2	3	4	5	ALL
1867	60					9	1899	20	10	17	32		18
1870	30		29	5	60	21	1906		10				2
1871				5		1	1908			9			2
1873				9		3	1909				6		2
1874				10		3	1910				6		2
1876	20					3	1912	10					2
1878		8	23	5	70	18	1928	10					2
1879		17	23	35		18	1932			9			2
1880		58	31	10		20	1935			9			2
1883	10	10	8		20	8	1937			9			2
1886			8			2	1939			9			2
1890		10	54	5		15	1940					11	2
1891				5		2	1941					10	2
1893		70	23		60	26	1943		13				2
1895				11		3	1957	13					2
1896				5		2	1962					9	2

calculation. The pre-1900 MFI for all sampled portions of the Frijoles watershed combined ranges from 2.4 to 22.9 years, corresponding to threshold scar indices between 0 and 50 (Table 4-4). A threshold scar index $\geq$ 10 yields a MFI = 7.2 years from 1598-1899, while a scar index $\geq$ 25 shows a MFI = 11.6 years for the period 1598-1893. Given that many trees may not scar in any given fire (Dieterich 1980-b), a MFI calculated with a scar index $\geq$ 25 (MFI_{25}) likely gives a reasonable summary estimate of the frequency of extensive fire events in the Frijoles watershed. Table 4-5 shows that pre-1900 MFI_{25} values for each individual group vary from 9.6 to 15.5 years. These MFI's fall within the range found in other Southwestern fire histories in ponderosa pine and mixed conifer vegetation types (Weaver 1951, Ahlstrand 1980, Dieterich 1980-a, Madany and West 1983, Dieterich 1983, Swetnam and Dieterich 1985, Swetnam et al 1988, Savage 1988, Swetnam-in press).

TABLE 4-4. Mean fire intervals (MFI's) for the Rito de los Frijoles watershed (all groups combined).

	Threshold Scar Index			
	≥ 0 (all fires)	≥ 10	≥ 25	≥ 50
period*	1598-1899	1598-1899	1598-1893	1599-1842
MFI (years)	3.1	7.2	11.6	24.3
std. dev.	2.4	5.5	7.2	22.9
median	3	6	10	20
range	1-12	1-23	1-32	8-87
# fire-years	99	43	28	11

* Periods begin with the earliest year with reliable sample depth (N = 5, or fewer samples showing a consistent pattern). Periods end with the last fire year ≥ the threshold scar index or 1899, whichever occurs first.

TABLE 4-5. Summary of pre-1900 mean fire intervals (MFI's) for each group, (for more information see the individual group MFI tables).

	Threshold Scar Index			
Group	≥ 0	≥ 10	≥ 25	≥ 50
1	8.6	8.6	15.5	26.1
2	6.8	8.4	10.7	18.9
3	6.0	7.3	11.2	33.4
4	4.4	6.3	12.7	17.5
5	6.4	6.4	9.6	13.7
Combined	3.1	7.2	11.6	24.3

Comparison of mean and median fire intervals for the whole watershed (Table 4-4) shows that the distribution of fire intervals is slightly skewed toward large values, with a few large fire intervals exerting a disproportionate influence on the MFI values. Still, the mean and median values are not greatly divergent, and they correspond even more closely when MFI's are calculated at the individual group level.

Watershed-wide fire years: The indication that a widespread fire occurred in a year with a large scar index value (Figure 4-11) can be confirmed by comparing scar index values between groups (Table 4-3, Figure 4-10) and mapping the locations of individual samples scarred in particular years (Figure 4-12). Extensive fires with large scar indices burned across the sampled portions of the Frijoles watershed in the years 1684, 1709, 1725, 1737, 1773, 1814, 1822, and 1842. These fire years show up as distinct steps in the cumulative graph of fire scars (Figure 4-9) because large numbers of trees scarred in those years. Fire years with lower scar indices also occurred in all 5 groups in 1806, 1847, and 1851, while the years 1757, 1763, and 1801 recorded large fire events everywhere except in Group 1. The occurrence of watershed-wide fire years across this large and environmentally heterogenous area indicates that vegetation structure and fuel conditions previously allowed fires to spread extensively across this landscape. Watershed-wide fires cease after 1851, although a relatively extensive fire occurred as late as 1899.

Climate interactions with fire: Lightning is common in the Jemez Mountains, as the convective cumulonimbus clouds of the summer monsoon bring an average of 62 thunderstorm-days/year to Los Alamos (U.S. DOE,

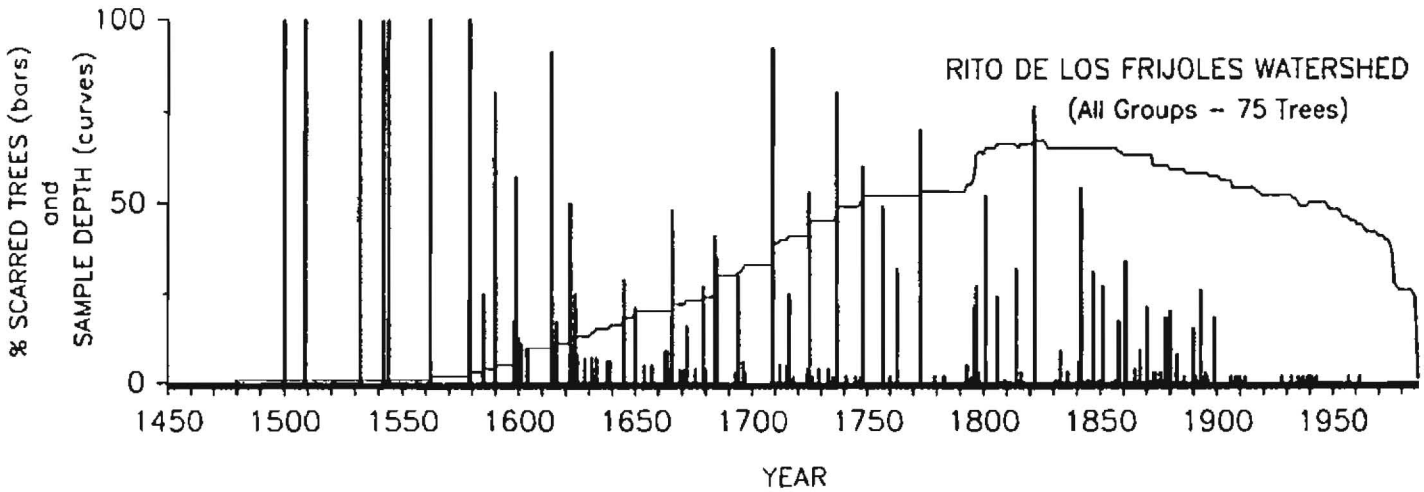
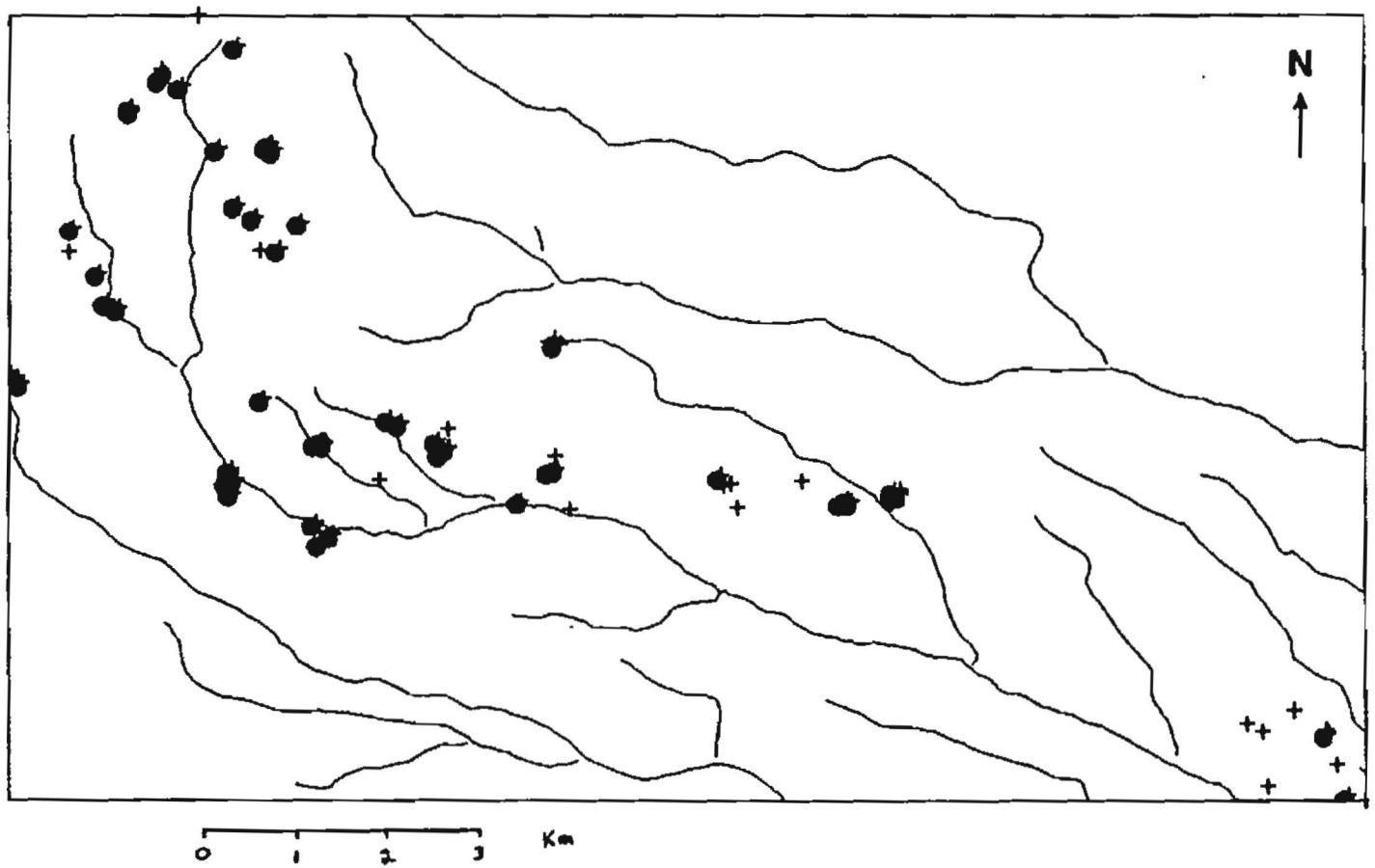


Figure 4-11. Combined fire-scar indices for the entire Rito de los Frijoles watershed.

Figure 4-12. Map showing all samples scarred by fire in 1822 (black circles), with unscarred samples = + . Drainage outlines are shown as lines.



1979). The Jemez Mountains experience one of the higher levels of lightning activity in the western U.S. (Reap 1986) - the BLM's automatic lightning detection system reveals that hundreds of lightning strikes can occur in the Jemez Mountains in a single day (BNM - maps from BLM on file). As a result the Jemez Mountains are subject to a high frequency of lightning-caused fires (Barrows 1978, Allen 1984-a).

There is typically a dry period of high fire danger from late April to early July, prior to the onset of the monsoonal rains. After thunderstorm activity declines in mid-September dry conditions usually recur, but the infrequency of lightning results in few ignitions. Foxx and Potter (1978) found that 86% of the fires recorded at Bandelier were lightning-caused, with a peak in July followed by June and August. Virtually all local fires occur between May and September. Barrows (1978) confirms this pattern of lightning ignitions. This overall climatic regime of spring drought followed by the onset of summer thunderstorms is thought to have persisted for about the past 8000 years in the Southwest (VanDevender and Spaulding 1979). Native Americans and Hispanic shepherds may have enhanced local fire frequencies in certain periods prior to this century, but the high incidence of lightning fire ignitions seems sufficient to explain pre-1900 fire intervals.

It seems probable that fires in the Southwest are more likely to occur in years of below-average moisture, although demonstrating a connection between dry years and fires has generally been difficult (McBride 1983), in part because even wet years display at least brief periods of dryness that permit fires to develop if ignitions occur. Linkages between fire occurrence and climate may also be obscured by

reliance upon historic fire records confounded by active fire suppression and changes in reporting technology. Swetnam (in press) has recently demonstrated that reduced fire activity occurs in the Southwest during years of "severe" El Niño events, when early summer and total precipitation are elevated (Andrade and Sellers 1988). The observed synchrony of large fire years throughout Arizona and New Mexico also indicates the presence of a regional climatic affect upon fire incidence (Swetnam - in press). Foxx and Potter (1978) found that historic fires in Bandelier tend to occur in years with below-average precipitation, especially following winters with subnormal precipitation.

Local tree-ring growth indices are particularly sensitive to the precipitation received during the previous winter and current spring (Rose et al 1981), which likely also affect the probability of fire incidence in the current year. Comparison of the tree-ring chronology from nearby Paliza Canyon (Dean and Robinson 1978, Figure 4-7) with the combined fire scar data from the Frijoles watershed for the period 1598-1899 reveals a strong connection between low growth (dry) years and and pre-1900 fires. Years with a scar index $\geq$ 25 (N = 27) display a mean tree-ring index of 0.644, standard deviation of 0.436, median of 0.65, and range from 0.02 to 2.05. By comparison, years with a scar index $<$ 25 (N = 275) have a mean tree-ring index = 1.080, standard deviation of 0.403, median of 1.12, and a range from 0.02 to 2.30. The probability that these fire years have lower tree-ring indices than non-fire years by chance alone is less than 0.00001 (Wilcoxon rank sum test, normal approximation corrected for ties). This is the strongest indication to date that pre-1900 fires typically occurred during dry

years in the Southwest.

Two small but noticeable gaps with minimal fire activity occur in this chronology from 1773-1793/6/7 and between 1822-1842 (Figure 4-10, 4-11). Other southwestern fire chronologies display a lack of fires in the 1830's (Swetnam - in press), likely related to the regionally wet conditions which prevailed throughout this decade - the 1830's show the highest decadal tree-ring growth index in the entire Jemez Mountains record (Dean and Robinson 1977, Figure 4-7). The 1780's fire gap is more puzzling, as this gap is not observed in other Southwestern fire chronologies and dry conditions probably occurred from 1773-1782, although 1783-1795 were likely quite wet (Dean and Robinson 1978, Figure 4-7). A variety of explanatory scenarios can be envisioned. For example, perhaps the extensive fires of 1773 (watershed scar index = 70) reduced the fuels so that ignitions in subsequent dry years did not spread well, and then the wet years in the latter part of the period inhibited widespread fires. Or, perhaps dry conditions and lightning ignitions failed to coincide during this period. Alternatively, this gap may simply represent a random departure from observed mean fire intervals. We still have much to learn about the timing and interactions between precipitation, lightning occurrence, fuel conditions, vegetation structure, and fire behavior in the Southwest.

Seasonality of past fires: The relative positions of fire scars within their annual tree ring of occurrence were determined for 847 fire scars, 73 % of the total sample (Caprio et al 1988). As described in Chapter III, each fire scar was assigned to one of the following categories: dormant season (D); early-earlywood (EE); earlywood (E),

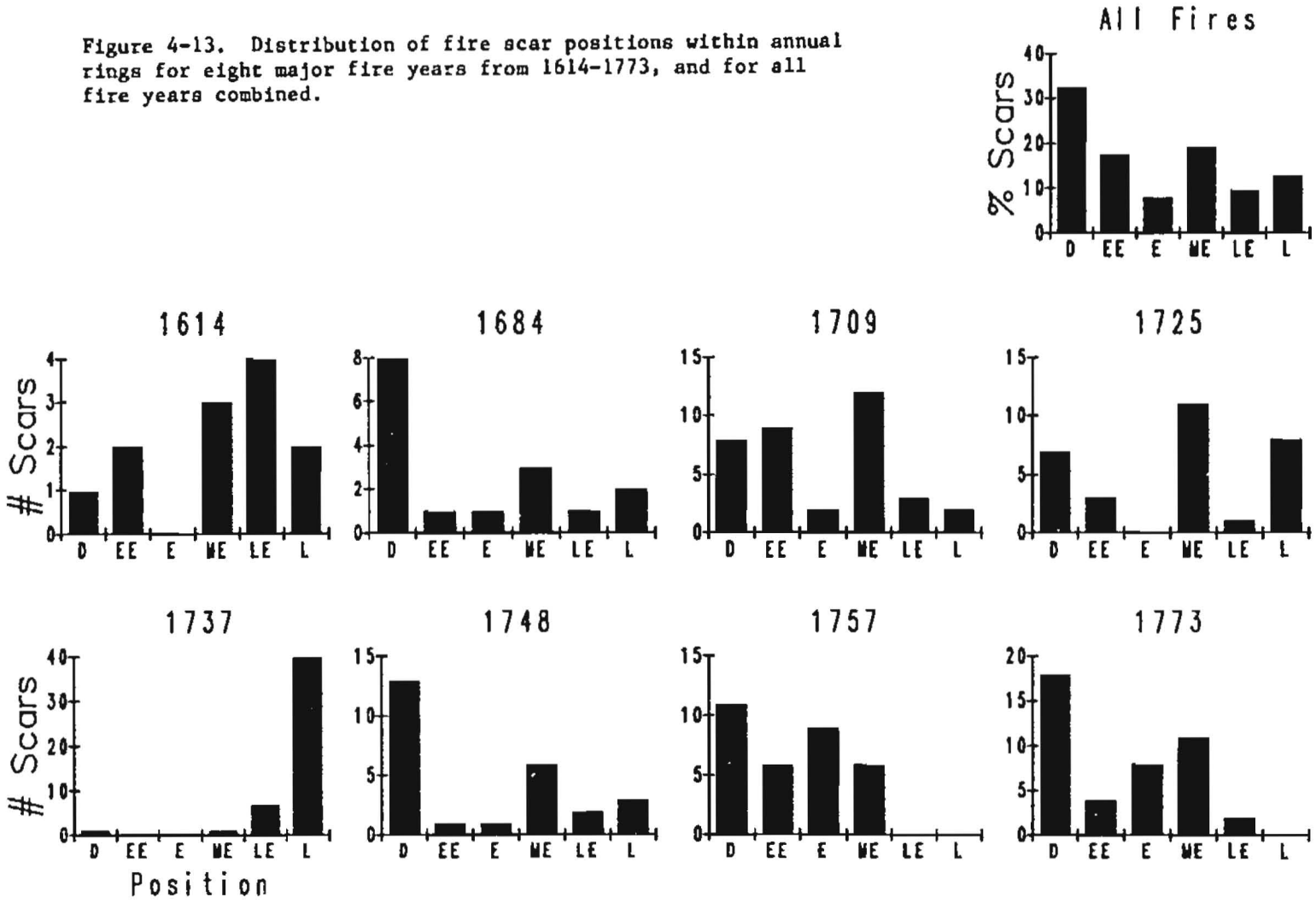
TABLE 4-6. Relative position within annual tree-rings of all fire scars by group. The percentages of total scars with position data for each group are listed, with the number of fire scars in parentheses. Percentage totals do not always equal 100 when added due to rounding variability.

Scar position	GROUP 1	2	3	4	5	Total
D	63 (47)	29 (56)	24 (53)	36 (116)	14 (3)	33 (276)
EE	8 (6)	20 (38)	21 (48)	17 (56)	4 (1)	18 (149)
E	5 (4)	8 (16)	8 (19)	8 (25)	11 (3)	8 (67)
ME	7 (5)	20 (39)	24 (53)	19 (61)	21 (6)	19 (164)
LE	9 (7)	9 (18)	12 (27)	7 (24)	21 (6)	10 (82)
L	8 (6)	14 (27)	12 (26)	13 (42)	29 (8)	13 (109)
Total (with position data)	100 (75)	100 (194)	100 (226)	100 (324)	100 (28)	100 (847)
U	(34)	(55)	(81)	(82)	(13)	(265)
no data	-	-	-	-	(48)	(48)
Total (all scars)	100 (109)	100 (249)	100 (307)	100 (406)	100 (89)	100 (1160)

mid-earlywood (ME); late-earlywood (LE); latewood (L); or uncertain (U). No position data exist for 48 scars from Group 5 that were previously analyzed by other workers (Robinson 1978). The convention of assigning D scars to the year of the subsequent ring seems appropriate for the Frijoles data, as D scars are typically associated in the chronology with earlywood scars of some sort in the subsequent year, and in no cases were latewood scars observed in the preceding year. The distributions of all scars by position and group are displayed in Table 4-6 and the "All Fires" graph in Figure 4-13.

Swetnam et al (1988) review the relationship between scar position

Figure 4-13. Distribution of fire scar positions within annual rings for eight major fire years from 1614-1773, and for all fire years combined.



and seasonal timing of recorded fires in southeastern Arizona, based largely upon work by Fritts (1976) in the Santa Catalina Mountains. At 2590 m in the Santa Catalinas cambial activity in ponderosa pine generally begins in late April, increases to a July/August peak, and stops by mid-September, although mature xylem cells may not develop until June (Fritts 1976). Thus dormant season scars usually occur between late September and April, although May fires could be recorded as D in some cases. Most earlywood scars (EE, E, and ME) likely occur in May and June, while LE and L scars likely occur in August and September. The seasonal distribution of Frijoles watershed fire scars will be considered in the context of this general pattern from southeastern Arizona since both areas have similar climatic regimes and timing of tree growth. Still, data on the phenology of local tree growth across the range of site conditions sampled in the Jemez Mountains are needed to make more precise statements about the seasonal timing of past fires here.

Spring and early summer fires dominate the Frijoles watershed scar record (Table 4-6, Figure 4-13). One-third of all scars with position data are from the dormant season (D), likely recording April and May (mostly) fires. Since another 18% of the scars are EE, 51% of the recorded fire activity probably occurred before the end of June. Mid-summer (July) fires likely center on the distribution of ME scars (19%), while probable August and September fires (LE, L) comprise 23% of the sample. This interpretation of the seasonal occurrence of past fires corresponds closely to modern monthly patterns of lightning fire ignition in this area (Barrows 1978).

The distribution of relative fire scar position by group generally

follows the overall watershed pattern, with the exception that early fires may be relatively more important at the low-elevation Group 1 site (Table 4-6). The proportion of late fire scars may be overestimated at Group 5 due to selective recording of this relatively distinctive scar type during the incidental analysis of scar position that most samples in this group received, or due to chance because of the small sample size; there is no apparent reason to suppose proportionately twice as many late fires burned at Group 5 compared to the adjacent Groups 1, 2, and 3.

The relative scar positions of 14 widespread fires between 1614 and 1899 are graphed in Figures 4-13 and 4-14. These graphs show that there was a great deal of variability in the probable seasonal pattern of burning between fire years, and none of these individual fire years matches the overall pattern for all fire years. Early fires predominated in 1684, 1757, 1822, 1842, 1861, and 1899, while late fires were prominent in 1614, 1725, and 1737. The most distinctive distribution pattern is observed with the fire(s) of 1737, which clearly occurred unusually late in the growing season. Fires burned throughout most of the fire season in many of these big fire years, with relatively even seasonal distributions observed in 1709 and 1801. These data support the belief that fires persisted through wet periods and spread during dry spells over the course of many months in the absence of human fire suppression (Swetnam - in press).

Changes in the distribution of relative scar positions may indicate a decrease in the relative importance of late fires and a reciprocal increase in the proportion of early season fires in the Frijoles watershed between the 1600's and the 1800's (Figure 4-15).

Figure 4-14. Distribution of fire scar positions within annual rings for six major fire years from 1801-1899, and for all fire years combined.

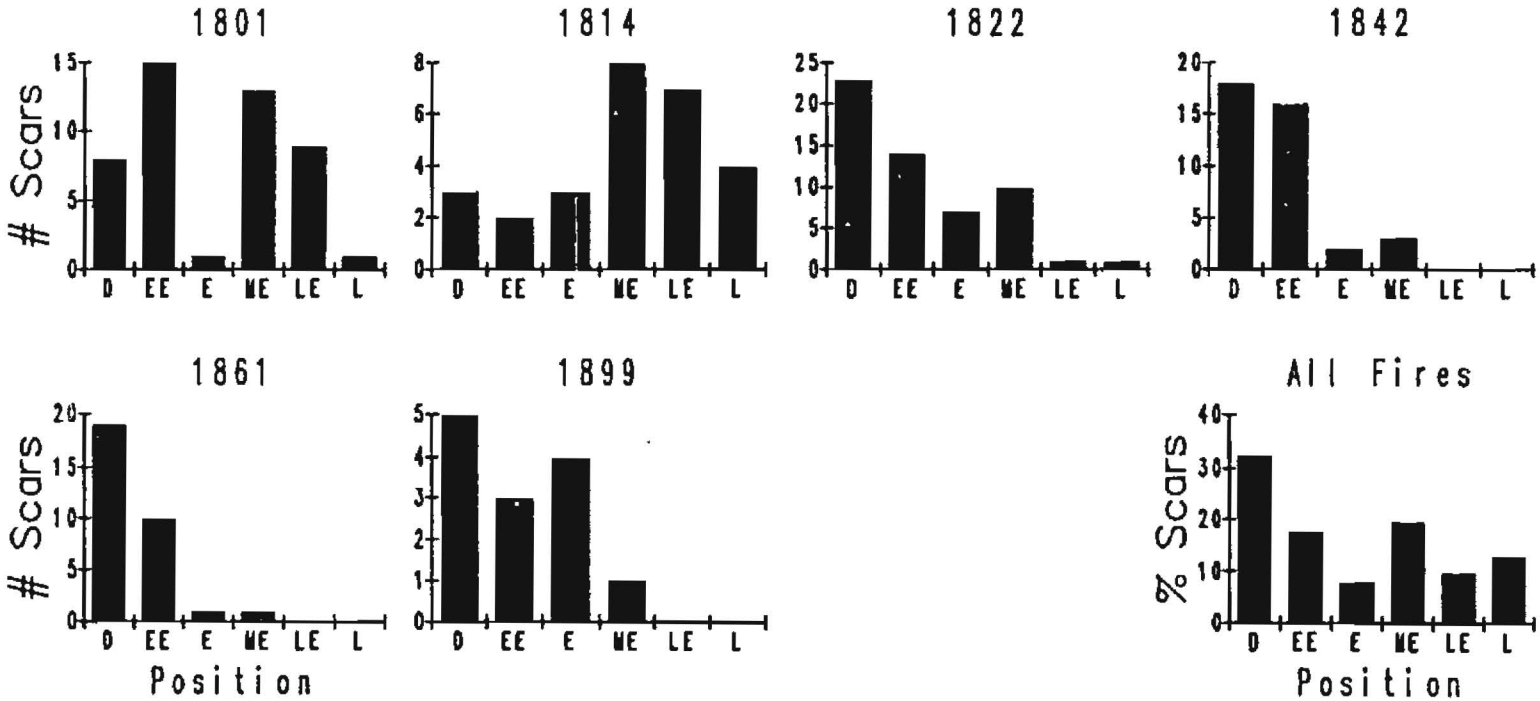
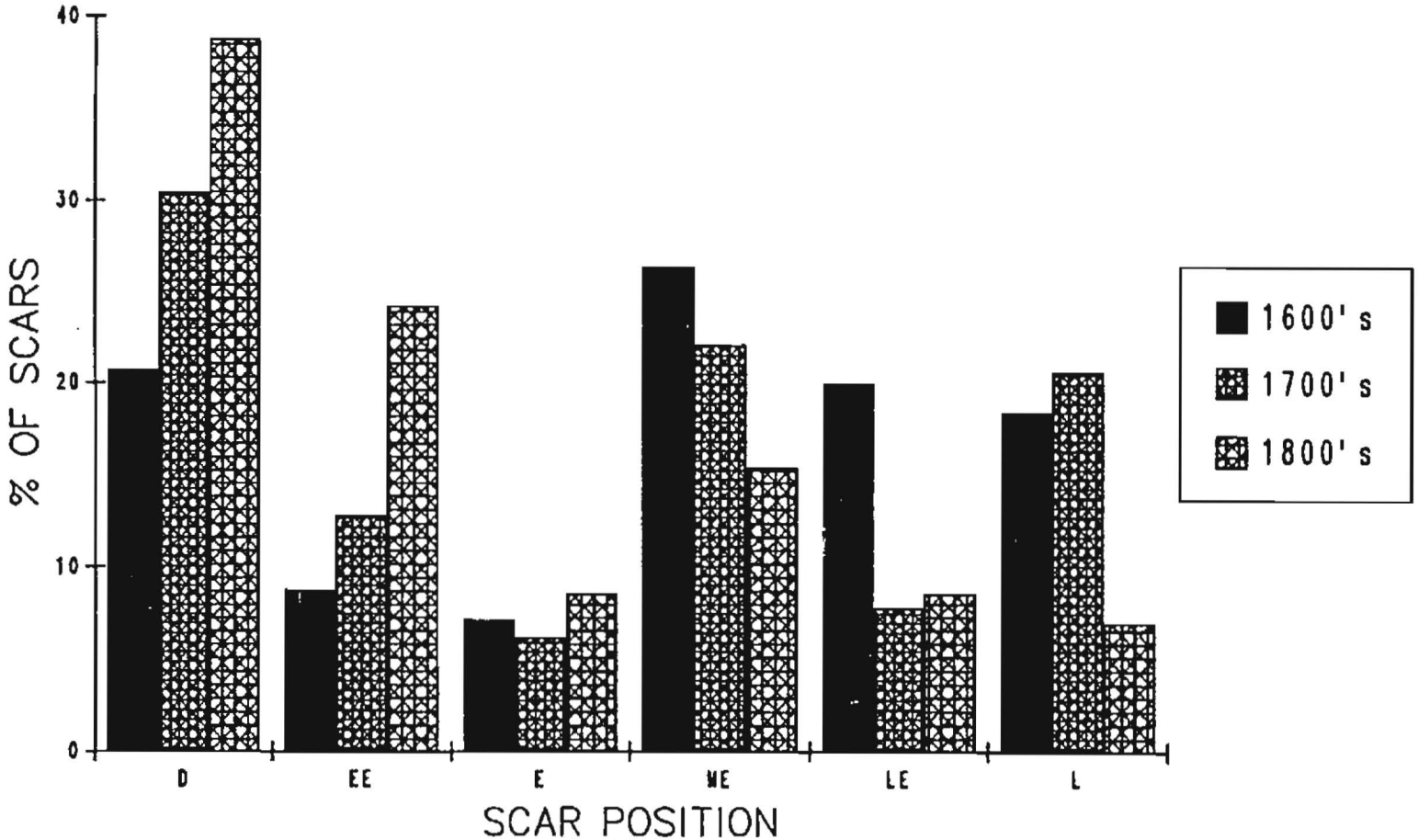


FIGURE 4-15. Relative scar position by century for all fire scars, 1600 - 1899.



These 3 centuries contain 97% of all scars with position data, with 127 scars in the 1600's, 321 scars in the 1700's, and 371 scars in the 1800's. This trend of changing scar position can also be seen in Figures 4-13 and 4-14. The probability that the 17th, 18th, and 19th centuries have the same overall distribution of scar positions is $<<$ 0.0005 (chi- square test), with $p << 0.0005$ that these 3 centuries have the same distribution of late fire scars (chi-square test), and $p < 0.0005$ that the three centuries have the same distribution of dormant season fire scars (chi-square test). Reasons for such a change in fire seasonality are not apparent, although it is possible that more early ignitions resulted from increased human activity in the Jemez Mountains after ca. 1850.

Changes in local fire regimes: The most obvious change in the sampled fire regimes of the Frijoles watershed is a sharp drop in fire frequency and extent after 1899 (Figure 4-10). A closer look reveals that fire size may have been declining in the last half of the 19th century, as 1842 was the last year with a watershed scar index $\geq$ 50, and 1851 was the last year that fires were recorded at all 5 sample sites. These relatively abrupt changes in long-standing fire regimes which took place in the late 1800's are remarkable in that they occurred well before human fire suppression became locally effective in the late 1910's (Allen 1984-a, deBuys 1985).

Similarly, abrupt interruptions of fire chronologies in the late 1800's are observed at numerous other locations in the Southwest, and at many of these sites the reductions in fire frequencies correspond to dramatic increases in domestic livestock populations that apparently reduced the herbaceous fuels supporting these ground fire regimes

(Leopold 1924, Cooper 1960, Komarek 1969, Ahlstrand 1980, Dieterich 1980-b, Madany and West 1983, Swetnam and Dieterich 1985, Bahre 1985, Savage 1988, Swetnam et al 1988, Swetnam - in press). Immense livestock populations also developed in the Jemez Mountains during the last several decades of the 1800's (Denevan 1967, Scurlock 1981, Rothman 1989 - discussed further below in **ANTHROPOGENIC DISTURBANCES: Livestock Grazing**). The overgrazing characteristic of this period likely caused great reductions in the quantity and continuity of herbaceous fuels, inhibiting the ability of surface fires to spread widely as well as reducing fire intensity. Decreased fuel continuity may explain the failure of fires to occur throughout the Frijoles watershed after 1851, with a complete collapse of the surface fire regimes eventually occurring as huge livestock populations spread throughout the landscape. When resolved at the group level, fires with a scar index $\geq$ 25 cease earliest at the lowest elevation site (1870 at Group 1) and latest at the highest elevation site (1899 at Group 4), consistent with increased livestock utilization near valley settlements first. It is possible that the small fire gaps observed in the 1780's and the 1830's in the Frijoles watershed chronology are at least partially attributable to the impacts of livestock grazing, as both gaps coincide with known periods of relative quiescence in Navaho raiding activity and probable increases in livestock numbers in northern New Mexico (Denevan 1967).

Thus inadvertent fire suppression by intensive livestock grazing in the late 1800's likely caused the initial drop in fire frequency and extent that persists through the 20th Century scar record. High livestock populations persisted at least into the 1930's (Smith 1953),

keeping herbaceous fuel loads at low levels. As woody fuel loads began building up in the early 1900's land management agencies began to make a concerted effort to control wildfires, which came to be viewed primarily as a destructive force (Pyne 1982); fire control efforts in the Jemez Mountains probably became effective by the late 1910's (Allen 1984-a, deBuys 1985). A locally wet period from 1905 to 1921 (T. Swetnam - unpublished tree-ring data from the western Jemez Mountains) may have contributed to low fire frequencies during this interval of early fire suppression, although this wet period is barely discernable at Paliza Canyon (Figure 4-7).

While confounded by increased human presence and improvements in reporting technology, it appears that the annual number of fire ignitions occurring in the Jemez Mountains has been increasing over the past 50 years (Allen 1984-a), a trend seen throughout the Southwest (Barrows 1978, Swetnam - in press). This may be due to the buildup of woody fuel loads that has been occurring with fire suppression (Dodge 1972, Foxx and Potter 1978, Allen 1984-c, Brown 1985). However, site-specific fire frequencies have been kept low by the early containment of formerly wide-spreading fires with active auppression, and by the fragmentation of the primeval landscape with roads which provide human access and act as firebreaks. Acreage burned in the Jemez Mountains also remains low on average because of successful suppression of almost all ignitions, although the building ladder fuel loads may be promoting increasingly large and intense fires in recent years (Foxx and Potter 1978, Allen 1984-a, Swetnam - in press). Former high-frequency, low-intensity fire regimes have been converted into low-frequency, high-intensity fire regimes by human suppression efforts.

For example, in 1977 the La Mesa Fire burned 6184 ha centered on the Frijoles watershed, including 4305 ha on BNM (Foxx 1984). Certainly the most widespread fire to occur in the watershed since at least 1899, the intense La Mesa Fire burned as a crown fire across many of the dense thickets of ponderosa pine that then characterized much of this area. Field sampling of the Frijoles watershed found that 402 of 969 points (41.5%) were burned by this fire, with partial or total kill of the former overstory forest at 329 points. Foxx and Potter (1978) showed that the intensity of this fire was proportional to length of time since the last fire, with crown fires occurring on the sites with the densest tree cover and highest fuel loads, sites that had not experienced fire since the late 1800's. This "unnatural" fire (cf. Kilgore 1985) created large treeless grasslands in the heart of the ponderosa pine zone of BNM and adjacent lands (Figure 4-16).

Prescribed fire is being increasingly used to reduce fuel loads and reintroduce a more natural fire regime into wilderness areas in the Jemez Mountains. For example, the adjoining NPS Bandelier and the USFS Dome Wilderness areas have a joint 'natural prescribed fire' program "to perpetuate the wilderness resource unimpaired for future use and enjoyment..." and "to restore and/or maintain the <u>primeval</u> character of the wilderness as realistically as possible" (BNM - fire managment plan on file). However, the seasonality, intensity, and areal extent of prescribed fires may differ markedly from natural fire regimes in order to minimize the probability of prescribed fires escaping from control. Local prescribed fires are often ignited in the fall months, while the fire scar record indicates that before 1900 most natural fires occurred during the extreme weather conditions of late spring and early summer

Figure 4-16. Map of 1981 ecosystem patches in the Bandelier National Monument area which originated in crown fires. Grassland = black, grass/shrubland = red, shrubland = gray, aspen stands = blue, and the Bandelier National Monument boundary is the black line. The 1977 La Mesa Fire created the large grass/shrubland and grassland patches.

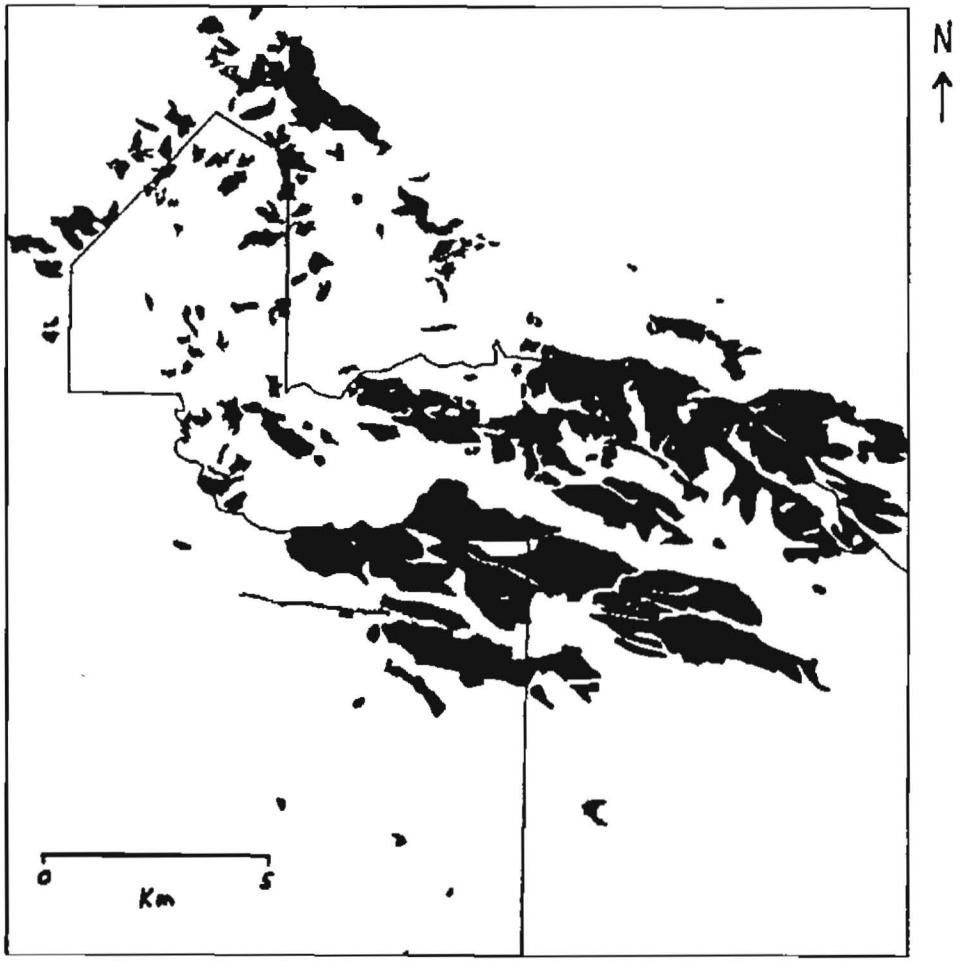


Figure 4-16

avoided by prescribed burning; fall fires were rare. Even in Bandelier's wilderness 'natural prescribed fires' continue to be carefully controlled to avoid the possibility of burning out of prescription, and fires that start naturally outside of the prescription (e.g. in early summer) are suppressed immediately, greatly reducing the naturalness of the resultant fire regime. Such a change in season of burning affects the intensity of resultant fires, thereby altering ecological effects. Altered seasonality of burning clearly has the potential to disrupt the life cycle of organisms whose phenology and life history are attuned to early summer fires. Modern prescribed fires cover only a few to several tens of hectares and only a minute fraction of the landscape (<< 1%) can be prescribe-burned in any year; in contrast, many natural fires covered hundreds of hectares, and in some years most of the Frijoles watershed burned (Figure 4-12).

Group 1: Ponderosa Pine/Piñon-Juniper (PPPJ) Transition, Frijoles Mesa

Group 1 samples the extreme lower end of the mesa-top distribution of ponderosa pine, where it interfingers with piñon-juniper woodlands. I am not aware of any published fire histories from this vegetation type (cf. Mastrogiuseppe et al 1983). Sample tree elevations ranged from 2016 to 2048 m. Here it was difficult to find trees or dead specimens that displayed more than 3-5 visible fire scars. At least partial explanations are that a portion of the potential fire history material was removed or damaged by earlier logging, snags were systematically felled for fire protection purposes after the extensive bark beetle mortality of the 1950's, and that scarred trees were used

for fuelwood since the Group 1 area is close to a park campground. Subsequent intensive exploration of this area has revealed several old stumps and logs with promising-looking catfaces that could be used to extend the fire history at this site. Many samples from this group were also difficult to date due to the galleries in the wood left by beetle larvae; these rendered one collected sample undatable (Caprio et al 1988). While the fire scar data for this site are not as numerous or as early as some of the other groups, they still provide a good representation of site fire history back to the late 1600's (Figure 4-17), with the earliest recorded fire in 1650 (Table 4-3).

Group 1 displays the most idiosyncratic fire history of the five sampled groups. Big fire years in the Frijoles watershed were usually recorded at all five sites, but extensive fires in 1757, 1763, 1801, and 1861 left no trace in Group 1 samples, while Group 1 experienced major fires in 1793 and 1867 that were not recorded at any other site. The last fire year exceeding a scar index of 25 occurred in 1870.

The pre-1900 MFI for this site (Table 4-7) ranges between 8.6 and 26.1 years, depending upon the threshold scar index value used in the calculation. Using a scar index $\geq$ 25 yields a MFI_{25} = 15.5 years for Group 1, the longest MFI_{25} for the Frijoles watershed (Table 4-5). The conventional wisdom states that most presettlement fires in ponderosa pine systems spread extensively in open grassy understories (Cooper 1960), but it seems probable that herbaceous fuels were relatively sparse and patchy at this low elevation, piñon-juniper transition site compared to more mesic ponderosa pine sites; this would reduce the probabilities of fire spread and tree scarring, and thus lengthen MFI's. Fire scars in this area tend to be shallower and typically do

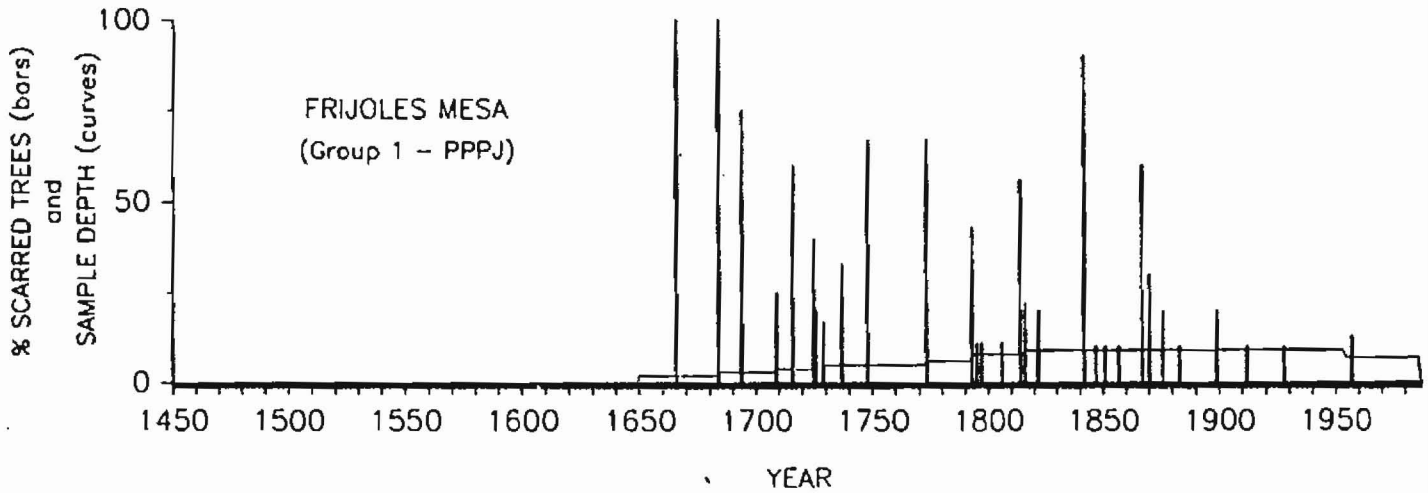


Figure 4-17. Fire-scar indices for Group 1, Frijoles Mesa.

TABLE 4-7. Mean fire intervals (MFI's) for Group 1, Frijoles Mesa.

	Threshold Scar Index			
	≥ 0 (all fires)	≥ 10	≥ 25	≥ 50
period*	1684-1899	1684-1899	1684-1870	1684-1867
MFI (years)	8.6	8.6	15.5	26.1
std. dev.	6.3	6.3	8.1	9.5
median	7	7	15	25
range	1-25	1-25	3-28	10-41
# fire-years	26	26	13	8

* Periods begin with the earliest year with reliable sample depth (N = 5, or fewer samples showing a consistent pattern). Periods end with the last fire year ≥ the threshold scar index or 1899, whichever occurs first.

TABLE 4-8. Mean fire intervals (MFI's) for Group 2, Upper Frijoles Canyon.

	Threshold Scar Index			
	≥ 0 (all fires)	≥ 10	≥ 25	≥ 50
period*	1680-1899	1680-1899	1680-1893	1680-1893
MFI (years)	6.8	8.4	10.7	18.9
std. dev.	5.6	5.9	6.3	4.4
median	5	8	10	19
range	1-24	1-24	1-28	12-28
# fire-years	33	27	21	13

* Periods begin with the earliest year with reliable sample depth (N = 5, or fewer samples showing a consistent pattern). Periods end with the last fire year ≥ the threshold scar index or 1899, whichever occurs first.

not extend as high up the trunk from the base of the tree as scars at other sites in this watershed (personal observation). This supports the idea that fire intensities were lower in Group 1 than the other groups. The MFI's for all fires and for fires with a scar index value $\geq$ 10 are the same here due to the small sample depth in this group.

Group 2: Riparian Mixed Conifer (RIMC) Forests, Upper Frijoles Canyon

The fire histories of Southwestern canyon forests are little known and thus largely speculative (Moir 1980). Group 2 samples the riparian zone of Upper Frijoles Canyon, which is dominated by mixed conifer species. Sample tree elevations range from 2172 to 2577 m, with all but sample #'s 29 and 30 below 2361 m. All samples were located on level floodplains or terraces along Frijoles Creek except for #'s 29, 30, 36, and 38 which occurred on canyon sideslopes less than 30 m above the canyon bottom. One sample (#35), an old-looking snag, remains undated due to its complacent ring pattern which fails to match the master chronology ring series. The earliest fire date in this group is 1600 (Table 4-3), with sufficient samples by 1680 to provide a reliable chronology (Figure 4-18).

The pre-1900 MFI for this group (Table 4-8) ranges from 6.8 to 18.9 years, with a MFI_{25} = 10.7 years. Most Group 2 fires displayed high scar index values ($\geq$ 25) and occurred in major watershed fire years which tended to be locally dry years (discussed above). Small fires were occasionally recorded solely in Group 2 (e.g., 1680, 1735, and 1747), but major Group 2 fire years were always extensive fire years in other groups. Examination of the spatial pattern of fires, as shown in Table 4-3 and Figure 4-10, indicates that Group 2 trees

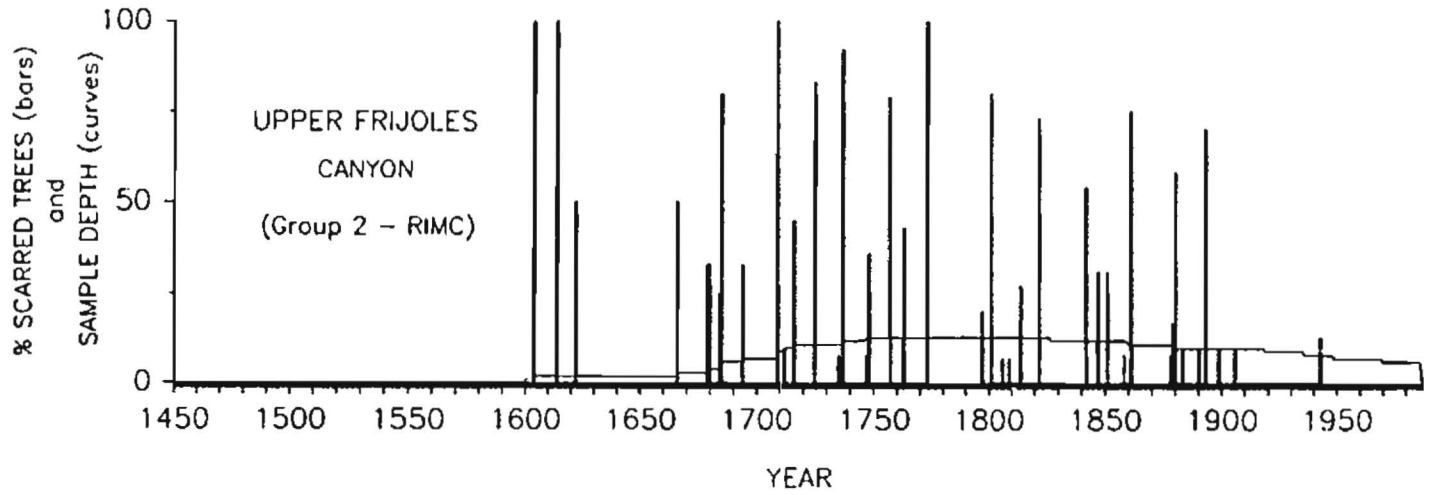


Figure 4-18. Fire-scar indices for Group 2, Upper Frijoles Canyon.

sometimes scarred in the same year as the second-most mesic group, Group 4 (e.g., 1685, 1763, 1861), but more often Group 2 burned in synchrony with Group 3 on adjacent Apache Mesa (e.g., 1666, 1694, 1716, 1797, and 1893). In some years only the lowest Group 2 sample (#45) scarred, recording fire dates that match with adjacent lower Group 3 and Group 5 but which are absent from the higher portions of Group 3 and from Group 4 (e.g., 1797, 1858, 1878, and 1893). These patterns may indicate that Group 2 fires were typically secondary ignitions that spread into Upper Frijoles Canyon from adjacent upland areas in dry years, and then burned through the canyon bottoms. The last major fire burned through Upper Frijoles Canyon in 1893.

There appear to be fewer fire-scarred trees in the lower reaches of Frijoles Canyon. Sampling these lower portions would reveal whether fires burned throughout the length of the canyon in major fire years or if the mid-canyon narrows acted as a firebreak, and if lower reaches of the canyon burned in synchronony with adjacent mesa-tops (e.g., Group 5, Group 1).

The Frijoles Canyon MFI_{25} of 10.7 years is lower than the 12.5 to 19.2 years found at Rhyolite Canyon in the Chiricahua Mountains (Swetnam et al 1988) and the 50 year interval from Boot Canyon in the Chisos Mountains (Moir 1982). This Group 2 MFI_{25} is also the second lowest of the five groups sampled in the Frijoles watershed, which defies expectations and raises the question of whether Frijoles Canyon is unique or if relatively high fire frequencies occurred in other Southwestern canyon systems. Numerous fire scars are evident in at least portions of Alamo Canyon and throughout the length of Capulin Canyon, suggesting that these adjacent canyons to the south also have

significant fire histories, although no sampling has occurred in these drainages. The Rhyolite Canyon fire history also indicates that frequent and extensive fires occurred along this canyon in southeastern Arizona, although complementary data from the surrounding uplands has not yet been obtained (Swetnam et al 1988).

This study answers some of the questions raised by Moir (1980) about the fire ecology of canyon woodlands in general and Frijoles Canyon in particular. Given the moist environmental conditions and mesic vegetation characteristic of such narrow riparian areas Moir expected "very local" fires and little spread of fires from adjacent uplands; neither supposition is supported by this pre-1900 fire chronology, although Moir's observation that the La Mesa Fire skipped across this canyon's bottomlands is certainly correct. Fire chronologies are needed from other Southwestern canyons to test the correctness of Moir's assertion that the fire histories of different canyon woodlands will be quite individualistic, although my study indicates that the fire history of at least one canyon is more closely related to the fire regimes of adjacent uplands than to the idiosyncracies of the canyon itself.

Group 3: Ponderosa Pine/Ponderosa Pine - Mixed Conifer Forests (PP/PPMC), Apache Mesa

The sampled area on Apache Mesa grades from pure ponderosa pine forests on the lower portions to ponderosa pine with a significant component of mixed conifer species in places on the upper reaches. The La Mesa Fire burned across this sample area in a patchy pattern of crown and surface fires. Sample tree elevations ranged from 2416 to

TABLE 4-9. Mean fire intervals (MFI's) for Group 3, Apache Mesa.

	Threshold Scar Index			
	≥ 0 (all fires)	≥ 10	≥ 25	≥ 50
period*	1598-1899	1598-1899	1598-1890	1590-1890
MFI (years)	6.0	7.3	11.2	33.4
std. dev.	4.4	5.1	6.0	17.4
median	6	7	10	28
range	1-21	1-23	1-24	8-64
# fire-years	51	42	27	10

* Periods begin with the earliest year with reliable sample depth (N = 5, or fewer samples showing a consistent pattern). Periods end with the last fire year ≥ the threshold scar index or 1899, whichever occurs first.

2698 m (Table 1). The presence of a number of trees over 400 years old in this sample allowed the earliest group estimates of MFI (Table 4-9), with the oldest fire dates in the Frijoles watershed (Figure 4-10) obtained from Tree #46 (pith date of 1459).

The MFI for Apache Mesa ranges from 6.0 to 33.4 years with a MFI_{25} = 11.2 years (Table 4-9), comparable to MFI's found in similar Southwestern forest types (Swetnam - in press). The scar index graph (Figure 4-19) shows that both large and small fires burned here. Major fire years in this group are well-represented in other groups (Table 4-3). The small valley that separates the upper six samples from the lower 12 samples apparently was not an effective fire break, although fires are only recorded on the lowest samples in 1814, 1833, and 1870 when fires were more widespread in lower elevation groups (Table 4-3). Apache Mesa could be considered the modal site of the five groups

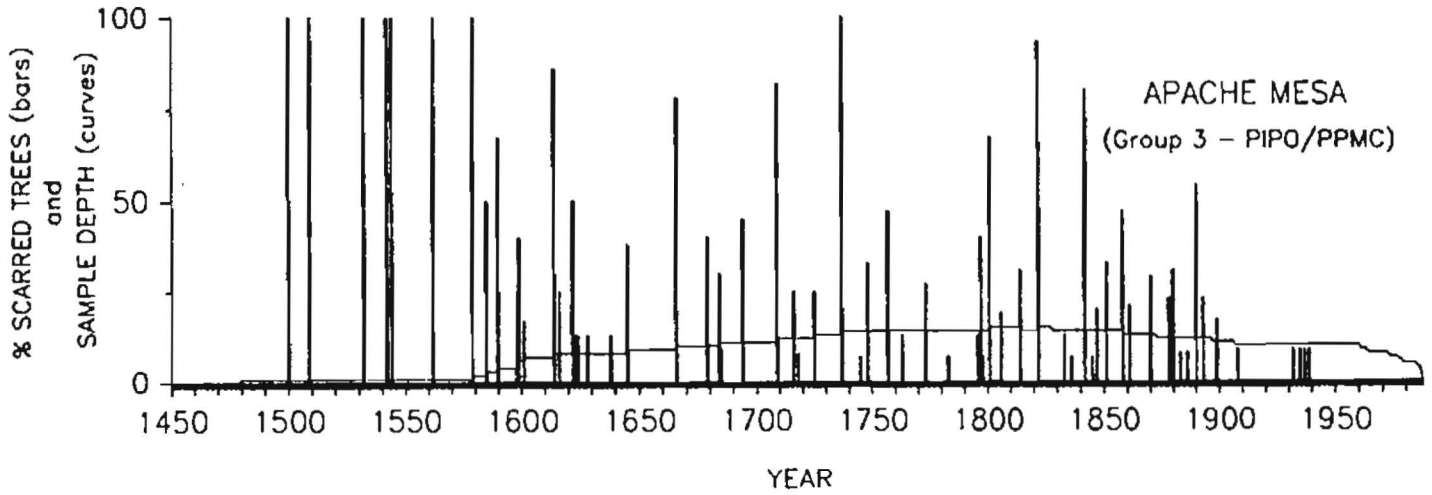


Figure 4-19. Fire-scar indices for Group 3, Apache Mesa.

sampled as this site most closely reflects the overall burning pattern of the whole Frijoles watershed (cf. Table 4-5, MFI_{10} and MFI_{25}), likely due to its intermediate elevational position and vegetation composition. The last fire with a scar index $\geq$ 25 burned in 1890, although a 1893 fire had a scar index of 23.

The lack of recorded fires between 1645 and 1666 may indicate another small gap in the fire chronology, or it may simply reflect the modest sample depth present during this period. Major fires were also absent from the only other group (4) with reasonable sample depth during this interval, although low scar index events are recorded (Figure 4-10). Climatic (Figure 4-7) and historic correlations with this possible gap are not apparent.

Group 4: Ponderosa Pine - Mixed Conifer/Mixed Conifer Forests (PPMC/MC), Cerro Grande Accession.

Group 4 samples the south-facing slopes and level surfaces of the Frijoles drainage headwaters. The vegetation at most of the sample sites consists of an overstory of ponderosa pine - mixed conifer, with mixed conifer species dominant in the understory. Sample elevations ranged from 2681 to 2934 m (Table 4-1). Adequate sample depth exists to construct a fire history back to the mid-1600's for this group (Figure 4-20, Table 4-10).

The MFI for Group 4 ranges from 4.4 to 17.5 years, with a MFI_{25} = 12.7 years. As might be expected (Dieterich 1980-a), the MFI_{25} value is longer here than at the lower elevation ponderosa pine sites in Groups 3 and 5 (Table 4-5), although a fire regime of frequent surface fires is still indicated. The MFI's of this PPMC/MC forest resemble

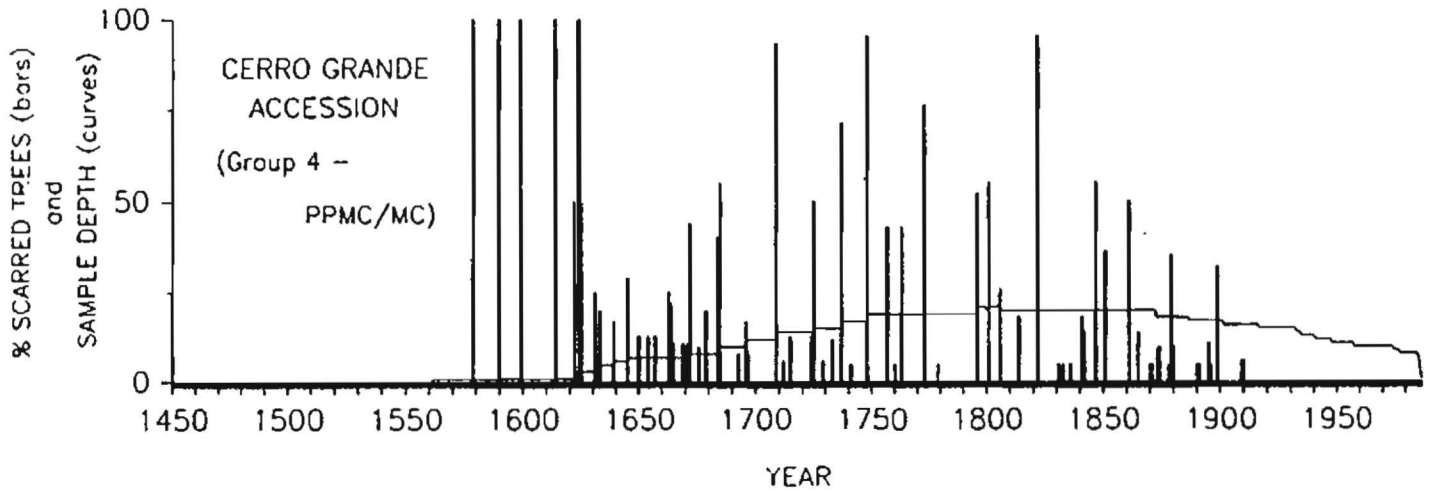


Figure 4-20. Fire scar indices for Group 4, Cerro Grande Accession.

TABLE 4-10. Mean fire intervals (MFI's) for Group 4, Cerro Grande Accession.

	Threshold Scar Index			
	≥ 0 (all fires)	≥ 10	≥ 25	≥ 50
period*	1639-1899	1639-1899	1645-1899	1685-1861
MFI (years)	4.4	6.3	12.7	17.5
std. dev.	3.3	4.9	7.1	7.2
median	4	5	12	21
range	1-17	1-23	1-25	4-25
# fire-years	59	42	21	11

* Periods begin with the earliest year with reliable sample depth (N = 5, or fewer samples showing a consistent pattern). Periods end with the last fire year ≥ the threshold scar index or 1899, whichever occurs first.

the few fire chronologies that exist from other Southwestern mixed conifer forests. Dieterich (1983) found pre-1900 MFI's ranging from 3 years for all fires to 22 years for watershed-wide fires in a mixed conifer forest in the White Mountains of Arizona. Ahlstrand (1980) found pre-suppression MFI's between 4.7 years for all fires and 17.4 years for "major" fires in mixed conifer forests of the Guadalupe Mountains on the Texas/New Mexico border. Kilgore and Taylor (1979-Table 3) indicate pre-1875 MFI's between 8.4 and 17.8 years from "clusters" of trees in the mixed conifer forests of the Sierra Nevada in California. The last major fire in Group 4 burned in 1899.

In an earlier study (Allen 1984-b) I compiled the fire history of a mixed conifer forest on Sawyer Mesa (elevation ca. 2685) adjacent to

the BNM boundary on Santa Fe National Forest land. Thirty-seven fire-scarred stumps, exposed by a logging operation, were sampled. Un-crossdated ring counts indicated a MFI_0 = 9.6 years for the period 1797-1893. The accuracy of the 1984 study can be checked by comparison with the Group 4 chronology, which includes Sample #'s 55-57 from adjacent to the earlier Sawyer Mesa study area. Comparison reveals a precise correspondence of fire years back to 1861, but earlier dates in the un-crossdated chronology are obviously 1 to several years off of the true fire date. Recalculation for the period 1801-1893 yields a MFI_0 = 11.5 years for the 1984 data set. This comparison confirms the conclusions of Madany et al (1983) that crossdating of fire-scarred samples improves data accuracy and is necessary if precise identifications of fire years are needed.

It must be noted that crossdating fire-scarred specimens can be a time-consuming and difficult procedure. The Tree-Ring Lab was unable to date three samples collected from the Cerro Grande Accession due to ring complacency. Potter and Foxx (1981 - Table 2) provide fire dates for nine scarred samples they collected and crossdated (when necessary) from the Cerro Grande Accession. Only four of their listed samples revealed pre-1900 fire dates, and these years do not correspond to the internally consistent dates found in the Group 4 chronology from the same area. Their results are also curious in that they show numerous 20th Century fires (23 of 45 total scar dates) continuing into the 1970's, whereas this study found only 18 of 1160 total scars dating to the 1900's. Resolution of the discrepancies between these sets of results is unlikely without examination of the 1981 samples, which apparently were not preserved (T. Foxx - personal communication).

Group 5: Ponderosa Pine Forests (PP), Escobas and Burnt Mesas

Group 5 consists of eight samples from Escobas Mesa and a cluster of 4 samples from across a 35 m deep valley on adjacent Burnt Mesa (Figure 3-3). Group 5 is separated from Apache Mesa (Group 3) by a 120 m escarpment along the Pajarito fault zone and from Frijoles Mesa (Group 1) by distance and another 40 m deep valley. Sample elevations varied between 2202 and 2246 m. The 1977 La Mesa Fire burned intensely through the dense ponderosa pine forests which covered these mesas, creating large grassy patches across the sample area (Figure 4-16). All Group 5 samples (except for tree # 26) were collected by other researchers and crossdated at the Tree-Ring Lab in the 1970's (see CHAPTER III. METHODS), explaining the lack of relative scar position data for most scars in this group (Table 4-6). I analyzed these earlier scar data by the same methods used for the remainder of my 1988 collections. Foxx and Potter (1978, 1984) used data from the earlier collections, along with fire dates they independently obtained from another 15 samples, to develop a fire history for this area.

The MFI for Group 5 ranges from 6.4 to 13.7 year, with a MFI_{25} = 9.6 years (Table 4-11). This is the lowest MFI_{25} value of the 5 sampled groups (Table 4.5), consistent with the frequent surface fire regimes found in other Southwestern ponderosa pine forests (Weaver 1951, Cooper 1960, Dieterich 1980-b, Swetnam and Dieterich 1985, Savage 1988). Sample depth is inadequate to calculate reasonable scar indices prior to 1797 (Figure 4-21) because the earlier collections did not emphasize dead and older material. However, Sample # 26 extends this groups's chronology back in time as it individually records 17 fires from 1725-1893 for a MFI of 10.5 years. Foxx and Potter (1984) provide

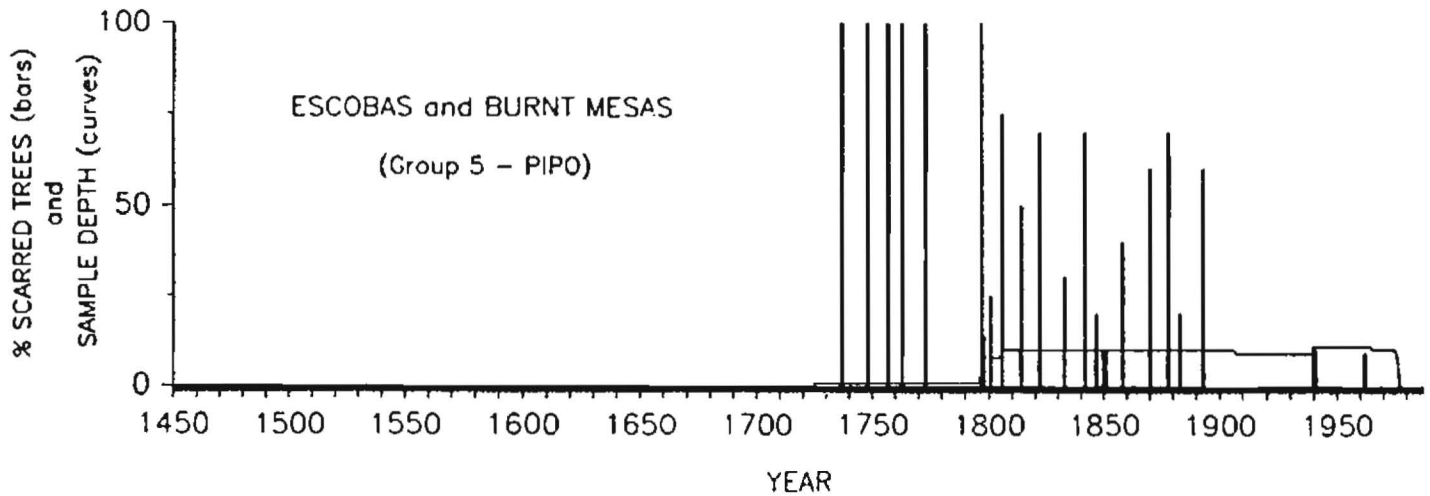


Figure 4-21. Fire-scar indices for Group 5, Escobas and Burnt Mesas.

TABLE 4-11. Mean fire intervals (MFI's) for Group 5, Escobas and Burnt Mesas.

	Threshold Scar Index			
	≥ 0 (all fires)	≥ 10	≥ 25	≥ 50
period*	1797-1893	1797-1893	1797-1893	1797-1893
MFI (years)	6.4	6.4	9.6	13.7
std. dev.	3.5	3.5	3.9	7.8
median	7	7	9	9
range	1-12	1-12	4-16	8-28
# fire-years	16	16	11	8

* Periods begin with the earliest year with reliable sample depth (N = 5, or fewer samples showing a consistent pattern). Periods end with the last fire year ≥ the threshold scar index or 1899, whichever occurs first.

a number of fire intervals for the pre-suppression period 1797-1893 in this area, including 12.0 years for MFI_{20} and 24.0 years for MFI_{30}. The Group 5 chronology indicates that the MFI_{20} = 8.0 years and the MFI_{30} = 10.7 years. Apparently Foxx and Potter's inclusion of additional specimens with non-matching fire years dropped some of the fire years included in my calculations (Table 4-3, 4-11) below the threshold scar index when figured with their complete data set.

The high scar index values in this group indicate that fires were typically extensive in these ponderosa pine forests. Fire years in Group 5 always corresponded with fires in Group 3 (with 1 exception, a small fire in 1850), especially with samples from the lower end of Apache Mesa. A previously noted connection exists between Group 5

fires and the nearest (lowest) Frijoles Canyon sample (#45). The last recorded extensive fire burned in 1893.

Unsampled Landscape Elements

Although the fire histories of several important vegetation/landform types characteristic of the Jemez Mountains have never been studied, it is possible to make limited statements about the pre-1900 fire regimes of these landscape elements.

Extensive piñon-juniper woodlands and juniper savannas cover the skirting mesas of the Jemez Mountains below the lower edge of the ponderosa pine zone. It is widely believed that surface fires once burned through the grassy understories of open piñon-juniper woodlands and savannas in the American West, preventing the trees from completely dominating most sites (Leopold 1924, Johnsen 1962, Burkhardt and Tisdale 1976, Young and Evans 1981, Evans 1988). The general scenario indicates that massive overgrazing of these woodlands in the 19th century greatly reduced the herbaceous understory, causing the surface fire regime to collapse and allowing piñon and juniper trees to greatly increase in density (Rogers 1982, West and Van Pelt 1987). A similar scenario has likely occurred in at least portions of the piñon-juniper woodlands of the Jemez Mountains. Group 1 (PPPJ, Frijoles Mesa) and Group 5 (PP, Escobas/Burnt Mesas), with known histories of frequent surface fires, spatially bracket some stands of piñon-juniper woodland, with no appreciable topographic barriers to prevent the spread of fires into the woodlands. Further, patches of old woodland are interspersed throughout the Group 1 sampling area such that fires must have burned into many woodland patches which are adjacent to fire-scarred ponderosa

pines. Certainly a pre-1870 MFI_{25} of about 15 years was operative along the ponderosa pine/piñon-juniper ecotone, and these fires would have spread with the prevailing winds down the mesas from this ecotone into the piñon-juniper zone if the pre-livestock grazing surface fuels could have supported them. Just as MFI's become longer between Escobas Mesa and Frijoles Mesa, woodland MFI's likely became longer with decreasing elevation due to declines in surface fuel quantity and contiguity. Thus pre-1870 woodland MFI_{25}'s of 15 to perhaps 40 years seem plausible, yet few fire scars or other evidence of past fires are found in the heart of the piñon-juniper zone below the range of ponderosa pine. There is currently insufficient herbaceous ground cover to carry surface fires except in restricted woodland localities. Until the local piñon-juniper woodlands and juniper savannas receive more intensive investigation their fire histories will remain speculative.

Fire scar data do not exist from the Jemez Mountains for mixed conifer forests on mesic sites, such as the north-facing slopes of the caldera rim peaks. The Group 4 chronology (PPMC/MC, Cerro Grande Accession) indicates that prior to 1900 a high-frequency (MFI_{25} = 12.7 years) surface fire regime occurred on the south-facing slopes and level surfaces of the upper Frijoles watershed. Denser mixed conifer forests, interspersed with large aspen stands and only infrequent ponderosa pine, blanket the adjoining north-facing slopes. Fires were apparently less frequent and sufficiently intense (at least on occasion) to crown and initiate succession to aspen stands. Pre-1900 MFI_{25}'s may have ranged from 20 years to 100+ years on extremely moist, protected locations. The potential exists to reconstruct fire

histories in these mesic mixed conifer forests by combining information from the relatively few fire scars observed on such sites with age structure data.

Montane grasslands are found on the upper south-facing slopes of most of the caldera rim peaks (Allen 1984-a). Old, fire-scarred trees are generally absent from these grasslands, but the fire histories of adjacent forests can be reconstructed and used as reasonable surrogates for grassland fire frequencies. Fire-scarred samples were collected in Group 4 on the slopes of Cerro Grande from locations immediately below and adjacent to the remnants of a large montane grassland. Fires that scarred these trees certainly must have spread upslope into the open grassland through the dense grass fuels that still persist in the open forest understories across this slope. Seventeen samples from the slopes of Cerro Grande, a subset of Group 4, were used to calculate MFI's for this site. These data yield pre-1900 values of MFI_0 = 4.9 years, MFI_{10} = 6.5 years, MFI_{25} = 10.8 years, and MFI_{50} = 15.1 years for the south slopes of Cerro Grande. This MFI_{25} value is lower than the 12.7 years found when all 23 Group 4 samples are used in the calculations because the subset samples a more homogenous environment (the Cerro Grande slopes) by deleting the six non-adjacent samples from upper Sawyer Mesa. Prior to 1900 the Cerro Grande grassland probably burned about every 10 years, on average, although higher mean frequencies could have occurred. As in the total Group 4 analysis, extensive fires ceased by 1900.

Engelmann spruce/corkbark fir forests are found on the north-facing slopes of the highest peaks in the Jemez Mountains, immediately above the mesic mixed conifer forests described above. Low-frequency,

high-intensity, crown fire regimes have been described for other spruce-fir forests in the Rocky Mountains (Peet 1981), with MFI's of 300 to 500 years likely (Romme and Knight 1981, Aplet et al 1988). The presence of discrete, large, aspen stands within many Jemez Mountains spruce-fir forests indicates that infrequent crown fires likely occur on these sites too. Given the short pre-1900 MFI's observed in adjacent Jemez Mountains vegetation types, and the recurrence of many probable drought periods over the course of centuries in this region (Figure 4-7), I suggest somewhat lower MFI's of 150 to 350 years for local spruce-fir forest sites. Such long pre-1900 MFI's suggest that 20th century fire suppression has had less ecological impact on spruce-fir forests than any other landscape element of the Jemez Mountains (cf. Romme 1982).

In summary, fire scar analysis conclusively demonstrates that past fires were frequent and widespread in the Jemez Mountains landscape. Fire has probable been more important in structuring local landscape patterns than any other natural disturbance. The pervasive impacts that have attended the cessation of the former high-frequency, low-intensity fire regimes typical of much of this landscape are developed below under <u>LANDSCAPE PATTERNS THROUGH TIME</u>.

Insect Outbreaks

Insect outbreaks occur when insect populations rise dramatically above their "normal" background levels so that they become apparent as a cause of significant defoliation and/or mortality to host plants. Such disturbances range from local to widespread, and may last for

years before the outbreak ultimately collapses and populations return to endemic levels (Barbosa and Schultz 1987). Insect outbreaks of sufficient magnitude to alter the gross structure of vegetation and landscape patterns have occurred in the Jemez Mountains. Partial outbreak histories have been pieced together for seven insect species in the Jemez Mountains and are reviewed here.

Western Pine Beetle (Dendroctonus brevicomus)

Arizona Five-Spined Engraver (Ips lecontei)

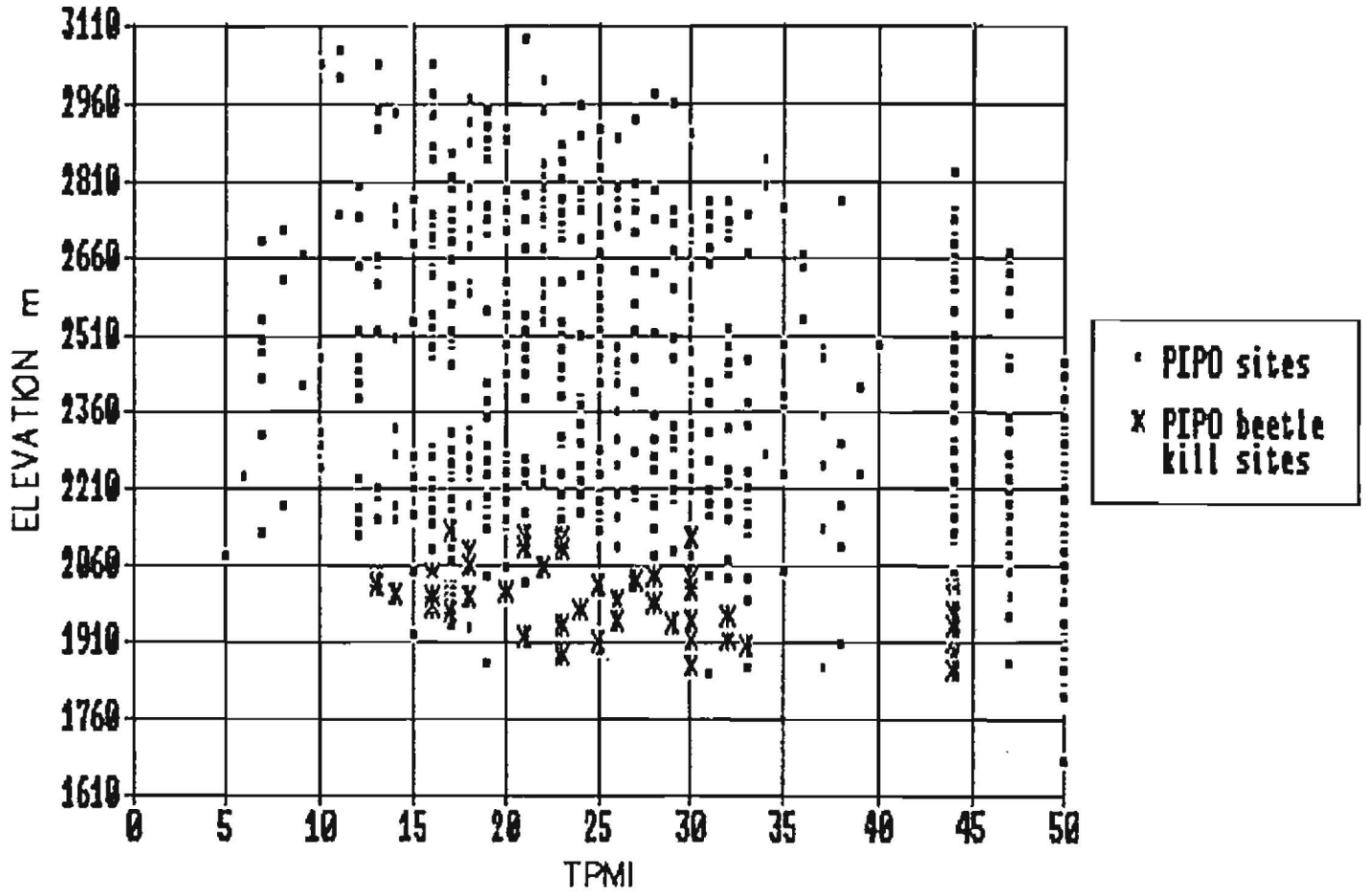
At background levels these two bark beetles kill senescing, individual ponderosa pine trees and small groups across this landscape. Group kills triggered by bark beetle attraction to and successful colonization of lightning-struck trees are described as important in structuring Texas pine forests (Rykiel et al 1988), but remain unstudied here.

A severe bark beetle outbreak involving both of these species occurred around Bandelier from 1955-1958 (BNM - annual forestry reports and other records on file), part of a larger region-wide outbreak (Choate 1966). Drought conditions, commonly implicated in insect outbreaks (Mattson and Haack 1987), are thought to have caused this outbreak by stressing the host ponderosa pine trees. Some mortality over background levels was apparently occurring in the early 1950's, with a serious outbreak recognized by 1955. A 1955 aerial survey estimated 5000 and 18,460 acres of Bandelier and Los Alamos lands, respectively, were infested, with 3237 heavily-infested brood trees and another 3000 previously attacked trees found in a 5% ground survey of 5000 acres of forest land and a total survey of the Los Alamos townsite

(BNM - F. Yasinski, Nov. 1955 report on file). A 21 Oct., 1957, memo by the Regional Forester states: "During the past three years an epidemic infestation of Southwestern (Western) Pine and the Arizona Five-spined Engraver Beetles have destroyed a majority of the ponderosa pine at Bandelier" (BNM - S. Carlson, on file). Control efforts were undertaken in 1957 and 1958. The outbreak peaked at Bandelier in 1957 despite the break in the drought, continued into 1958, and then collapsed by late 1958/early 1959 when it became "practically impossible to find Ips, or Dendroctonus ..." (BNM - W. McCambridge, 13 April, 1959, memo on file). Around Bandelier the outbreak was most severe along the lower fringes of the ponderosa pine zone on the mesas of the Pajarito Plateau, with "mortality in some sections of the monument (being) 95% of all age classes" (BNM - S. Carlson, 21 Oct., 1957, memo on file). Paul Judge, the park superintendent during the outbreak years, recalls that "it looked awful for awhile" due to the large number of snags left by the beetles (personal communication). Using the thousands of ponderosa pine logs and fewer snags that remain as evidence, field sampling found 66 points where ponderosa pine had been reduced by at least one abundance value class in the lower ponderosa pine zone of the Frijoles watershed (Figure 4-22), almost certainly by this outbreak. The elevational zone of beetle-kill sites ranged from 1850 m to 2123 m, with a mean elevation of 1992 m (std. dev. = 59 m). At 28 of these sites ponderosa pine was lost as a co-dominant vegetation component.

Fragmentary information exists for earlier bark beetle outbreaks in the area. A map dated January, 1935, displays 39 points where "beetle control" was carried out in the lower ponderosa pine zone of

FIGURE 4-22. Sample points where ponderosa pine trees were apparently killed by bark beetles.



Bandelier (BNM - map on file). No other information on this control effort, or the conditions that prompted it, has been found to date. While the insect situation on the Santa Fe National Forest was reported as "normal" in the regional summary for 1934, it also notea that "Drought throughout the Region during the past year has allowed the building up of the insect population. This is particularly true in the pine type, where rainfall has been abnormally low during the year" (USFS Southwest Regional Office - 6 Nov., 1934, memo on file). The mid-1930's were atypically dry at Bandelier.

A 1947 memo (BNM - on file) states that mortality of ponderosa pine had been increasing in Bandelier since 1945, with group kills confined to low-quality sites and decadent or lightning-damaged trees. "Severe" losses of about 200 piñon trees/year were also reported from the Otowi section for 1946-1947. The dry years of 1942-1943/1945-1947 likely contributed to this increased bark beetle activity at Bandelier, as well as to the Ips outbreak noted in the southern portion of the Jemez Mountains in 1946 (USFS Southwest Regional Office - 27 Feb., 1947, memo on file). Otherwise, USFS annual regional reports of insect control activities on the Santa Fe National Forest (covering the years 1925-1939/1943/1945-1947/1949-1950) do not mention any local bark beetle outbreaks.

Piñon Ips (Ips confusus)

In endemic conditions piñon ips attack primarily injured and uprooted piñon trees (Furniss and Carolin 1980). A severe outbreak of these engraver beetles occurred in the Bandelier area during the 1950's drought, concurrent with the bark beetle outbreak in ponderosa pine

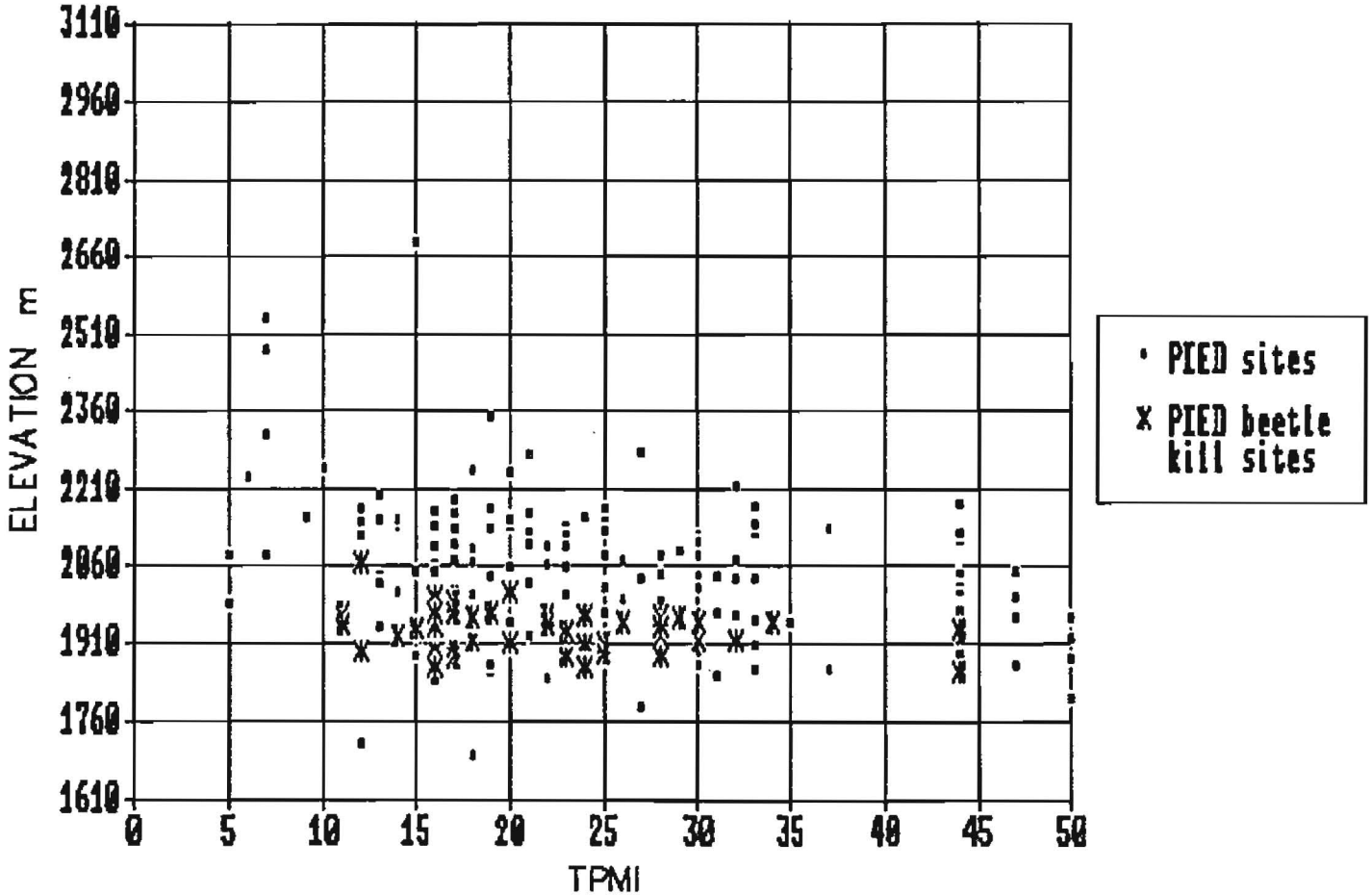
noted above (Bandelier Nat. Mon.- records on file). Reports at the time implicated Ips lecontei in the piñon mortality, but the primary attacker was likely the closely related Ips confusus. This 1950's outbreak in Bandelier is probably the infestation illustrated by Chansler (1964). This episode killed 100% of the piñon trees on many of the drier, piñon-juniper sites in the Frijoles watershed, where the piñon trees were presumably under water stress. The annual forestry report for 1957 (BNM - on file) states: "The effects of beetle attack increased the mortality rate of all age ponderosa and pinyon pine to the point of near extinction on the lower elevations". This mortality prompted control efforts in 1957 and 1958. The characteristic feeding galleries of this engraver beetle can still be seen on some long-dead piñons that litter these sites. Marked reductions of piñon abundance were observed at 58 field sample points (Figure 4-23), with an elevational range of 1841 m to 2062 m and a mean of 1940 m (std. dev. = 42 m). Piñon was eliminated as a codominant at 37 of these points.

Spruce Beetle (Dendroctonus rufipennis)

Engelmann spruce is the primary host for spruce beetle in the Jemez Mountains. These beetles prefer downed trees, but single trees and groups are attacked and killed in spruce forests under endemic conditions (Schmid and Frye 1977). Major outbreaks originate in material from blowdowns or logging operations.

An outbreak occurred on the north slopes of the highest summit in the Jemez Mountains, Chicoma Peak, and on adjacent Polvadera Mesa between 1968 and 1971. Scattered windthrow in these stands may have spawned this outbreak (Schmid and Frye 1977). Continual beetle buildup

FIGURE 4-23. Sample points where pinon trees were apparently killed by bark beetles.



in secondary windthrow around the margins of the sanitation/salvage clearcuts used to control the initial outbreak caused management problems throughout the 1970's (Schmid 1981, E. Collins, USFS ranger-personal communication).

Western Spruce Budworm (Choristoneura occidentalis)

Outbreaks of spruce budworm, the most destructive defoliator in the West, may last for over a decade and affect large geographic areas (Furniss and Carolin 1980). Spruce budworm primarily attacks white fir and Douglas-fir in the Jemez Mountains. Recent tree-ring studies by Swetnam (1989) indicate that widespread spruce budworm outbreaks have occurred in the Jemez Mountains since at least the 1700's. In upper Frijoles Canyon Swetnam found evidence of budworm outbreaks in the early 1710's, early 1750's, late 1760's, late 1820's, ca. 1840, early 1870's, late 1910's, late 1930's, 1950's, and finally the early 1980's. The last two outbreaks are confirmed by USFS records (Brown et al 1986). The 1980's outbreak caused significant mortality of understory white fir (especially) and Douglas-fir in parts of the upper Frijoles watershed, with probable budworm mortality observed at 24 field sample points between 2448 and 2974 m in 1987. The intensity and synchroneity of spruce budworm outbreaks may be increasing across large areas in the southern Rocky Mountains, including the Jemez Mountains, likely due to changes in forest stand structure and increases in the budworm's primary host species with historic fire suppression and logging activities (Swetnam and Lynch - in review).

Western Tent Caterpillar (Malacosoma californicum)

Fall Webworm (Hyphantria cunea)

Principal hosts for the western tent caterpillar in the Jemez Mountains are species in the genera Populus, Alnus, Betula, and Prunus. Tree growth loss or mortality from these defoliators is usually minor, with the exception of outbreaks in quaking aspen (Furniss and Carolin 1980). USFS regional insect control summaries between 1925 and 1950 record significant tent caterpillar outbreaks in aspen stands on portions of the Santa Fe National Forest in 1931-1932, 1935, 1942, and 1949-1950. Control efforts were implemented for tent caterpillars in the riparian deciduous forests of lower Frijoles Canyon in the 1950's (BNM - records on file). A severe outbreak affected aspen throughout much of the Jemez Mountains in the early 1980's, when annual and even biannual defoliations occurred for several consecutive years in many aspen stands (personal observation). Tree-rings reveal that minimal growth took place during this outbreak, and some mortality resulted in the eastern Jemez Mountains.

Fall webworm is another common defoliator that typically causes little damage to its principal local hosts, alders and cottonwoods. Park records on file at Bandelier show that periodic efforts were made to control fall webworm for aesthetic reasons in lower Frijoles Canyon during "outbreaks" in 1953, 1955, and 1958.

Tree Diseases

Tree diseases will be discussed as chronic environmental influences on distributions of individual tree species in subsequent sections rather than as pulsed disturbance regimes here. Major

diseases considered include dwarf-mistletoe (Arceuthobium spp.) on ponderosa pine and true mistletoe (Phoradendron juniperinum) on one-seed juniper.

Large Vertebrates

While even small animals can have large cumulative effects upon their environment (Huntly and Inouye 1988), large vertebrates often create readily visible impacts that qualify as disturbances, such as deer browsing a preferred food item to near-extinction (Alverson et al 1988) or beaver altering landscape patterns (Johnston and Naiman 1987). The degree of disturbance typically depends upon the fluctuating population levels of the species in question. Several large vertebrates may act as disturbances in the Jemez Mountains landscape.

Beaver (Castor canadensis)

Beaver alter landscapes by cutting trees and shrubs and by impounding stream courses (Johnston and Naiman 1987). As in much of North America, by the 1930's beaver populations in northern New Mexico had been decimated by past trapping. Early wildlife reports at Bandelier (BNM - on file from 1934/1934/1940) state that there was no evidence of past or present beaver use of the canyon tributaries of the Rio Grande, except for occasional use at the mouth of Frijoles Creek. Subsequent memos record that two beaver were planted in the upper portions of Frijoles Creek in 1941, and by 1948 the park beaver population was estimated to be ten individuals. But by 1957 beaver in the park were "very scarce", and annual population estimates ranged between none and four through 1965, when two more beaver were stocked

in Upper Frijoles Canyon. Beaver have not been seen above Upper Falls (a 20 m high barrier located 1.5 km above the Rio Grande confluence) since 1977, although a wet meadow persists in Upper Frijoles Canyon at the former site of a beaver dam. In 1987 beaver returned to the mouth of Frijoles Creek, cutting many narrowleaf cottonwood trees and building a dam. Beavers rapidly reduce their food supply in the narrow riparian zone of Frijoles Creek, thus prompting me to speculate that they are unable to sustain long-term populations in the isolated headwater canyons of this area.

Porcupine (Erithizon dorsatum)

Porcupine can cause considerable damage to commercial forest trees. A 1934 report states that "porcupine have been poisoned at various times in the past and are now rare" in Bandelier (BNM - B. Thompson, 18 April, 1934 report on file). With protection their populations apparently rebounded so that by the 1950's low-level concern was expressed in the park over the damage that porcupines were inflicting on trees in developed areas (BNM - annual wildlife reports and other memos on file). Field sampling in 1987 recorded porcupine damage at 53 sample points in the Frijoles watershed; 87% of these points were below 2530 m, likely due to the increased visibility of affected tree crowns in the more open vegetation of lower elevations.

Feral Burro (Equus asinus)

Feral burros have been controversial in Bandelier since at least 1940, when W. McDougall described an increasing population of 15-18 burros and claimed that they "do a considerable amount of damage to the

range... ...they should be eliminated from the monument" (BNM - 7/40 report on file). Concern has been expressed more or less continuously since then over the impacts of burros on the vegetation and soils of the dry mesas and canyons of the southern half of the monument. Primarily grazers, local burros also browse shrubs in direct competition with mule deer (Potter and Berger 1977). Several studies in the 1970's tied declining range trends and accelerated soil erosion across large areas of Bandelier to overgrazing by burros (Koehler 1974, Morgart and Ohmart 1976, Potter and Berger 1977, Morgart 1978).

In the first burro reductions, carried out from 1946-1948, 67 individuals were killed (BNM - F. Binnewies, 12/8/48 report on file). Despite repeated expressions of concern from 1957 onward in various memos and despite burro population estimates of 100+ by 1965, no further reductions occurred until 1975 when supportive research findings became available (Koehler 1974). The burro population was then estimated to be about 130 animals before 52 were shot. Subsequent reductions include 16 shot in 1976, 9 captured and 66 shot in 1977, 37 shot in 1980, 29 captured and 22 shot in 1983, 3 shot in 1985, and 5 shot in 1987. Legal battles from 1980-1983 with two animal protection groups eventually supported NPS management actions. Bandelier burro populations are thought to have been below ten individuals since 1983, but ongoing vigilance is required since this alien species continually reinvades from the adjacent Santa Fe National Forest where it is protected under the federal Wild Horse and Burro Act (J. Lissoway, BNM resource manager - personal communication). Minimal ground cover and excessive erosion persist in the backcountry despite greatly reduced burro populations (see discussion below under ANTHROPOGENIC

DISTURBANCES: Accelerated Soil Erosion).

Mule Deer (Odocoileus hemionus)

Mule deer are almost exclusively browsers in Bandelier (Potter and Berger 1977), and like other deer elsewhere can greatly reduce preferred browse species when at high population levels (Alverson et al 1988). Potter and Berger (1977) found that mountain mahogany (Cercocarpus montanus) comprised half or more of the volume in deer fecal pellets at most locations in all seasons, with four wing saltbush (Atriplex canescens) and oak (Quercus) species also important. Pajarito Plateau populations are resident (Eberhardt and White 1979): fewer deer seem to inhabit higher elevations in the Jemez Mountains and many of these migrate to lower elevations in winter to avoid deep snows.

A 1934 wildlife survey found mule deer to be scarce throughout Bandelier and the Jemez Mountains, although some tracks were seen (BNM - A. Borell, Nov., 1934, report on file). This scarcity was attributed to hunting pressure and the past abundance of predators such as mountain lions and wolves. With protection from predation and adequate food, deer populations can grow rapidly. In 1940 W. McDougall noted that, "Mountain mahogany is quite abundant in places and most of it seems to have been very severely browsed" (BNM - July, 1940, report on file), although trespass cattle and burros may have supplemented deer browsing. By 1945 the park custodian states "mule deer seem to be getting more numerous every year in the vicinity of Frijoles Canyon Headquarters and it may be not many years before we have a 'deer problem' (BNM - C. Thomas, 9/30/45 report on file). While clearly

imprecise, deer population estimates in annual wildlife reports on file at BNM show a perception of markedly increased deer numbers by the 1950's:

Year	Population
1942	120
1943	100
1948	270
1951	1162
1953	865
1957	1780
1960	600
1961	400
1962	450
1963	550
1964	550+
1965	500+
1966	500+

In 1960 J. Spillett emphasized that: "The greatest wildlife problem at present in Bandelier is that of too many deer... Almost the entire deer range in Bandelier is over-browsed and in poor condition" (BNM - 16 Sept., 1960, report on file). Deer census, control, and browse condition studies were proposed in 1961, 1963, and 1964 (BNM - records on file), but apparently not carried out. By the early 1960's a similar deer population buildup and resultant control efforts had occurred on adjacent LANL lands (Eberhardt and White 1981) and in the National Forests of New Mexico (Findley 1987). A corresponding increase in mule deer numbers in the Intermountain West was attributed to increases in shrub dominance due to livestock impacts and fire suppression (Gruell 1986).

More recently Potter and Berger (1977) reported heavily browsed mountain mahogany in some areas of Bandelier. An airplane overflight in 1987 counted 121 deer in Bandelier, indicative of a significantly larger total population (BNM - New Mexico Dept. Game and Fish tally sheet on file). Fieldwork in 1987 shows that mesa-tops in the Frijoles

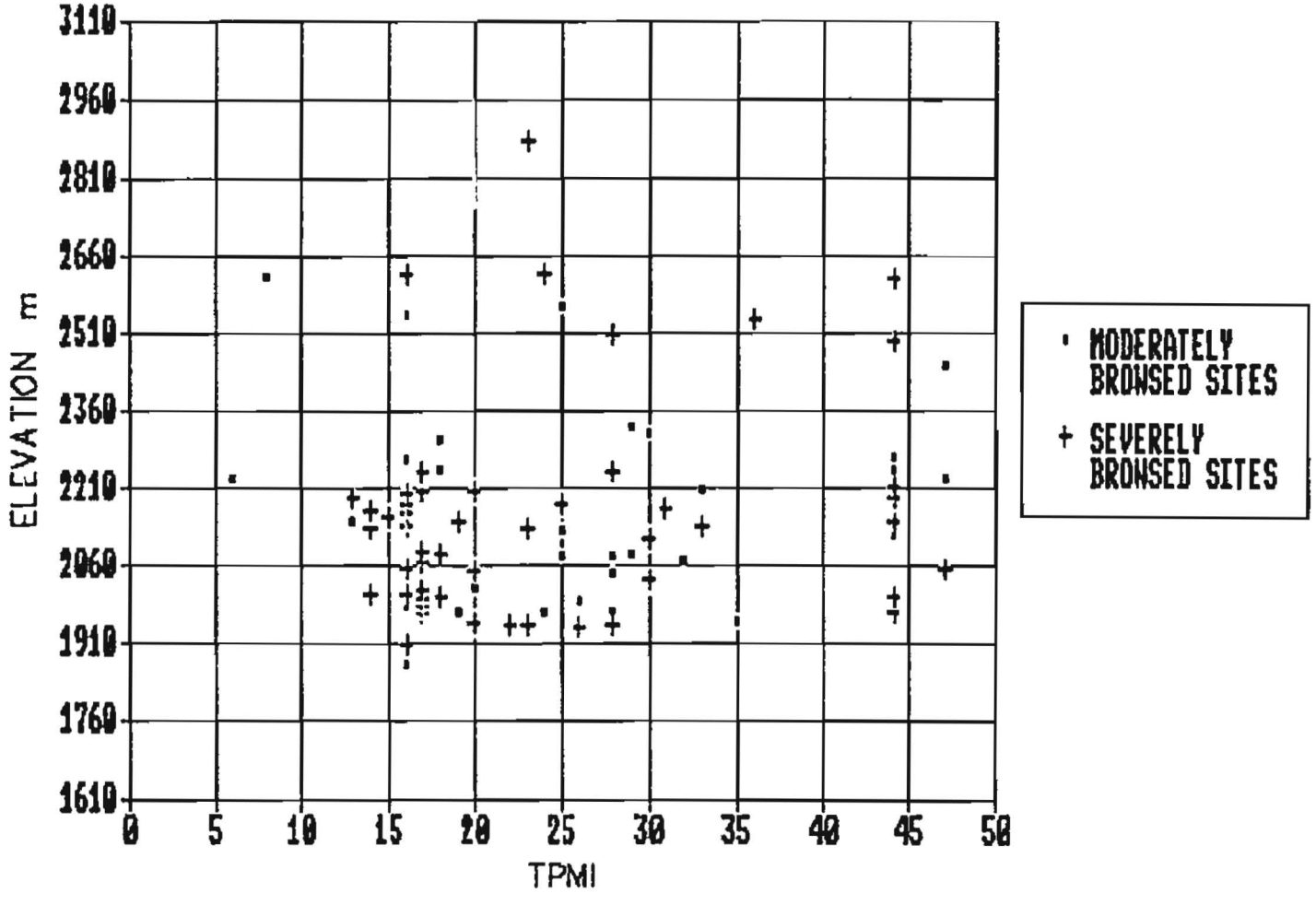
drainage continue to display extreme browsing of mountain mahogany and other species (Figure 4-24) - 97 sample points record "moderate" or "severe" browsing of shrubs. The bulk of this browsing (81.4%) was found across a 300 m mesa-top elevational zone, between 1937 and 2237 m. Browsing by mule deer populations may be seriously inhibiting the reproduction of mountain mahogany at many mesatop sites, although vigorous, unhedged stands may be found on steep canyon walls where browsing pressure is presumably reduced (personal observation).

Rocky Mountain Elk (Cervus elaphus nelsoni)

In the Jemez Mountains elk are primarily grazers, although they also use woody browse species (Rowland et al 1983). Local elk use is concentrated on non-forest or early successional forest areas (White 1981). In the Jemez Mountains elk spend most of the year at high elevations, with the bulk of the population seasonally migrating to lower elevation mesa sites to avoid deep winter snows.

Populations of Rocky Mountain elk were present in the Jemez Mountains into the late 19th century, but by 1909 all elk populations in New Mexico were considered extirpated (Findley 1987). In 1948 the New Mexico Game and Fish Department released 21 cows/calves and 7 bulls of C. elaphus into the Jemez Mountains; they had been brought from Yellowstone National Park (BNM - S. Keefe, 25 Sept., 1948, report on file). By 1961 the Game and Fish Dept. estimated a population of 200+ in the Jemez Mountains, all descendants of the 28 founders (BNM - 19 Dec., 1961, letter on file). Yet from 1948-1965 elk were rarely seen in Bandelier, with park population estimates ranging from 0 to 6 individuals. In 1964-1965 another 58 elk from the Jackson Hole area

FIGURE 4-24. Severely and moderately browsed sample sites.



were released into the mountains of Los Alamos County adjacent to Bandelier (White 1981).

The elk population in the Jemez Mountains has possibly doubled in the last decade, to its current level of roughly 6000-8000 individuals, with about 3500 elk summering on the Baca Location (R. Isler, New Mexico Dept. of Game and Fish - personal communication). Hunter kill statistics and observations by local inhabitants indicate that the Jemez Mountains elk population may actually be closer to 10,000 (J. Elsen, USFS range and wildlife specialist - personal communication). Since the 1977 La Mesa Fire the population of the Bandelier area elk herd has increased dramatically due to the creation of about 6000 ha. of grassy winter range. In 1979 about 200-400 elk wintered in the La Mesa Fire area (Rowland et al 1983) - wintering populations on Bandelier and adjacent LANL and USFS lands are now estimated to be 1000-2000 individuals (R. Isler, New Mexico Dept. of Game and Fish-personal communication). White (1981) expressed concern over the potential for elk overpopulation and resultant vegetation damage, soil erosion, and automobile accidents in the Jemez Mountains. Every year more elk seem to be wintering farther down the mesas of the Pajarito Plateau, thus increasing utilization of piñon-juniper woodland areas. Given the already minimal herbaceous ground cover present on these sites the potential for negative impacts from elk utilization similar to those documented for feral burros exists. For example, the shattered rosettes of Yucca baccata record one elk food preference from piñon-juniper woodlands (personal observation).

Elk can inflict considerable damage on aspen by eating the bark in winter or by direct browsing of regenerating shoots (DeByle 1985).

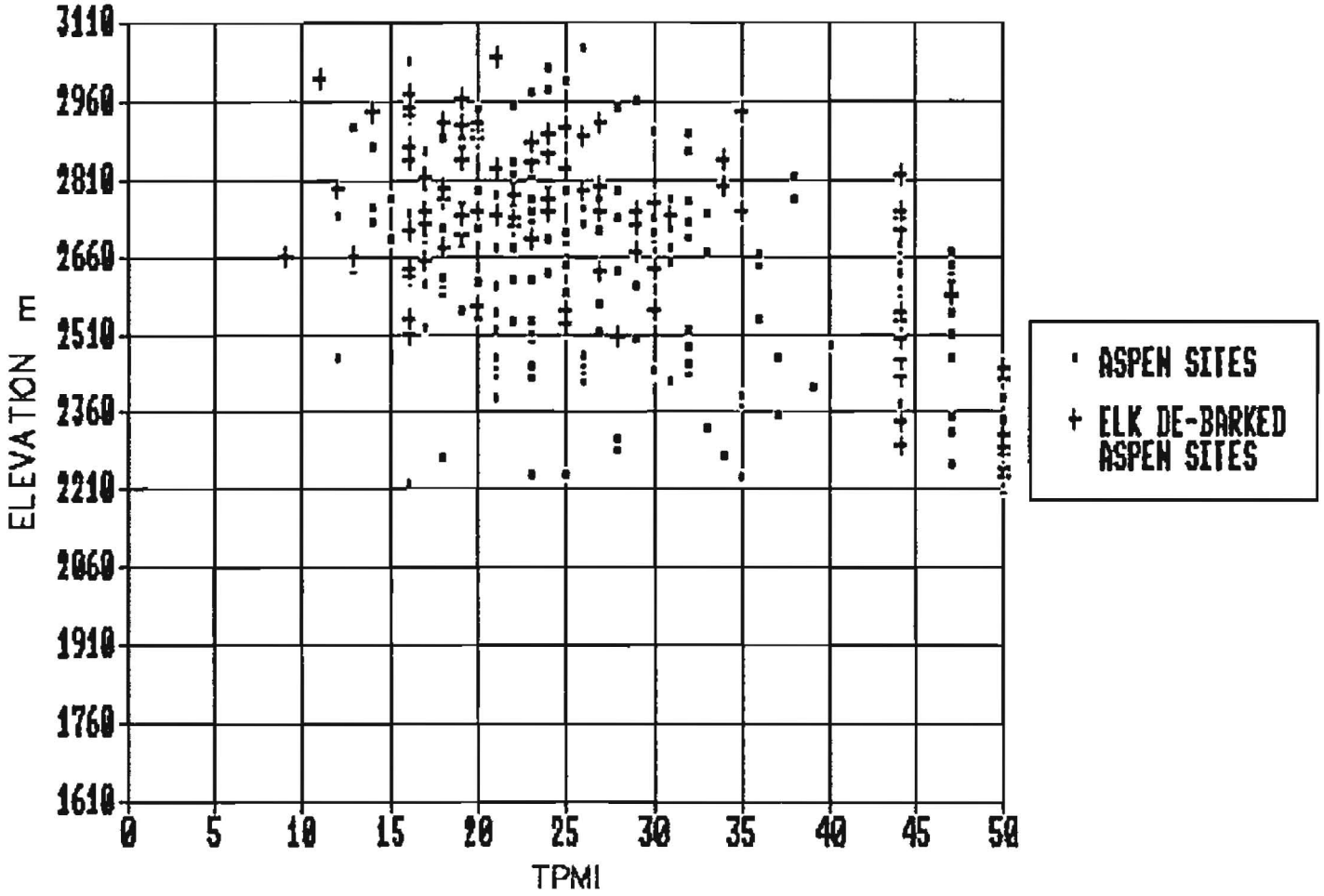
Fieldwork documents significant elk de-barking at 90 points (Figure 4-25), 33% of the 269 points with aspen abundance values of 2 or 3 in the Frijoles watershed. I have observed aspen stands heavily de-barked by elk throughout the eastern half of the Jemez Mountains. Atop Cerro Grande a small aspen clone was apparently killed by a combination of severe elk de-barking and fungal pathogens. In several places on Apache Mesa aspen stands that resprouted after the La Mesa Fire have been browsed so heavily by elk that the continued survival of the clone appears questionable.

Buckrubs from elk (primarily) and deer serve as another source of mortality for young trees, particularly invasive trees in or at the edge of open meadows and grasslands above 2400 m. This process plays a small role in slowing the succession of open areas to forests. Saplings of limber pine seem to be preferentially selected as buckrubs, most noticeably within forest interiors. Buckrubs were observed at 55 points between 1944 and 3052 m in the Frijoles watershed.

Windthrow

Windthrow is an important disturbance regime in some forested landsapes (White 1979, Canham and Loucks 1984, Franklin and Forman 1987). The tornadoes, downbursts, and hurricanes that cause much wind disturbance elsewhere are absent in the Jemez Mountains, although locally strong winds do occur in association with frontal storm systems and thunderstorm cells (U.S. DOE 1979, B. Bowen, LANL meteorologist-unpublished data). Recorded maximum wind gusts at LANL indicate about a 2-year return interval for a 62 mph gust, and a 100-year return interval for a wind gust of about 80 mph (B. Bowen - personal

FIGURE 4-25. Elk de-barked aspen sample points.



communication). Widely-distributed, individual, windthrown trees are found throughout the Jemez Mountains, likely occurring due to some combination of localized wind gusts with tree senescence, root rot, and soil conditions.

One major windthrow event is recorded for this area. Strong winds following heavy rains on the night of April 29, 1942, caused extensive blowdown damage in ponderosa pine forests in and around Bandelier (BNM - C. Richey, 15 May, 1942, memo on file). The monthly climatological summary for New Mexico stated, "A severe south wind on the 29th caused considerable blowdown damage in northern forested areas" (Cameron 1942). Albuquerque recorded a 68 mph southerly wind gust that evening, which is about a 10-year event for that station (Simiu et al 1979). This windstorm occurred at the end of the wettest April on record at Bandelier (8.6 cm of precipitation versus the mean of 2.1 cm), which likely exacerbated the severity of the blowdown. A series of memos from subsequent months (BNM - on file) describe an estimated 5,000,000 board feet of timber blown down within the park, with at least another 2,500,000 board feet felled on adjacent USFS lands; restoration of phone service and clearing of the Dome Road took weeks. The hardest hit areas were the headwater mesas and canyons of the Frijoles, Alamo, and Capulin drainages. Up to 90% of the trees were blown down in spots, with an estimated 15,000 trees down in one 3000 acre portion of the mesa between Frijoles and Alamo Canyons. The fire hazard and insect outbreak potentials were considered to be so high that the park staff advocated salvage logging most of the down timber despite its remote location. Mesa del Rito was roaded and salvage logged by the USFS at this time, but wartime labor shortages and second thoughts

scuttled NPS salvage efforts inside the park.

Another local windthrow event may have occurred in the 1930's (H. Pickens - in Foxx and Potter 1978). Small blowdown events recorded in park files include 20 cottonwood trees in the headquarters area of Frijoles Canyon in July of 1961, and scattered dead snags on the mesa tops (likely relicts of the 1950's bark beetle outbreak) in March of 1963.

Scattered windthrow in Englemann spruce stands on Polvadera Mesa may have contributed to a local spruce beetle outbreak in 1968-1971 (see **Insect Outbreaks**: Spruce Beetle above). Windthrow problems also occurred around the margins of salavage-logged clearcuts in this area. I have observed "windthrow gashes" in the spruce-fir forests along the ridge extending west of adjacent Chicoma Peak, where narrow, linear, windthrow areas develop perpendicular to the abrupt ridge-crest ecotone between grassland and forest. It appears that the blowdown of one or a few trees may lead to a preferential funneling of turbulent winds into the slit-like gap, causing other newly exposed trees to fall and extending the gap into the forest until the effect fades. Such windthrow gashes form a definite pattern of nearly parallel, linear slices into the forest that is visible on air photos or on the ground. The profuse spruce reproduction found in some of these gaps indicates that these windthrow gashes may allow the forest to continually regenerate along this ridge (cf. Sprugel 1976)

Lightning

Lightning impacts on forests are reviewed by Taylor (1971). Lightning disturbance in the Jemez Mountains is a low-probability,

high-intensity cause of mortality for individual trees. Lightning-scarred snags or live trees were observed at 152 sample points throughout the Frijoles watershed. Lightning strikes may interact with bark beetles to create patch kills in pine stands (Rykiel et al 1988), although this is not documented locally. Most importantly, the Jemez Mountains experience a high frequency of lightning-caused fires (Barrows 1978, Allen 1984-a, reviewed above) which have been vital determinants of local landscape patterns.

Hail

Hail commonly accompanies summer thunderstorms, especially at higher elevations. Elevations in the Frijoles watershed, above roughly 2600 m, may be pounded by several hail storms each growing season (personal observation). Thick blankets of hail frequently found in the mountains in the middle of the growing season may limit the distribution of some organisms. The high probability of damage to broadleaf trees may restrict their ability to successfully compete with needleleaf trees where hail is so common. The trembling habit of aspen leaves may reduce hail damage, and when combined with aspen's capacity to sprout new leaves the same season after defoliation, this may partially explain its presence as the only broadleaf canopy dominant of higher elevations in the Jemez Mountains. Obvious hail damage was observed at five sample points in 1987.

Snow

Broadleaf plants are often structurally damaged by snowfall when they are leafed out. Thus the timing of unusual snowfalls may affect

the distributions of some broadleaf trees. For example, an unusually early, wet, heavy snowfall occurred October 10-12, 1986, causing extensive damage to narrowleaf cottonwoods in Frijoles Canyon. The trees had not yet shed their leaves. Field sampling in 1987 found evidence of snow-induced damage to broadleaf trees at 13 points in this riparian forest, concentrated along 4 km of the lower end of the canyon between 1745 m and 1877 m where cottonwoods are most common. Heavy spring or fall snows can also damage high-elevation aspen stands (Jones and DeByle 1985-a), like a contorted stand found near the summit of Polvadera Peak in the Jemez Mountains.

Flooding

Flooding is an important disturbance regime in most riparian zones (cf. Johnson et al 1985). Flooding along the Rio Grande, primarily associated with spring snowmelt, was formerly a critical determinant of successional patterns in the riparian zone (Hink and Ohmart 1984, Dick-Peddie et al 1984). Regulation of the river's flow by upstream and downstream dams has greatly altered the previous flooding regime.

Flood disturbance in the canyon riparian situations of the Pajarito Plateau has received little study. The June 1977 La Mesa Fire severely burned about half of the Frijoles Creek watershed, which promoted unusually rapid surface runoff until ground cover redeveloped; as a result summer thundershowers caused repeated and extreme flooding in Frijoles Creek in 1977 and 1978 (Purtymun and Adams 1980). Similar extreme, post-fire, runoff events are also known from the Lummis and Alamo Canyon drainages (Purtyman and Adams 1980). Field sampling in 1987 recognized evidence of overbank flows (flooding) at 46 sample

points along 9 km of Frijoles Canyon. Future utilization of fluvial terraces, laminated sediments which contain charcoal layers and buried tree stumps, and probable flood scars on trees could improve our knowledge of the past flooding history of this watershed.

A major flood event may have occurred ca. 1640 A.D. in Upper Frijoles Canyon, based upon the synchroneity of establishment dates observed among the trees sampled for fire scars here (Table 4-2). Sample trees 37, 39, 43, and 45 all grew on the floodplain along a 6.5 km stretch of the canyon bottom and have pith dates between 1641 and 1650. Sample 40 likely established during this time interval also. The only two older trees sampled here grew on the slopes above the upper floodplain terrace. A flood might have promoted the establishment of ponderosa pine by providing a mineral seedbed and reducing competition with herbaceous plants. This regeneration pulse is not observed in the other sample areas, and post-fire reproduction can be discounted by the lack of any scars from the 1620's through 1670's on the two older trees sampled in this area, and the absence of a major fire in the 1630's or early 1640's in the scar record of the adjoining Apache Mesa site (which has a closely correlated fire history). Increasing the number of known dates of tree establishment in the floodplain by increment coring trees would help resolve or reject this hypothesized flood event.

Landslide

Landslides are important disturbances in some landscapes (White 1979, Garwood et al 1979). In the Jemez Mountains landslide phenomena are largely restrict to the steep slopes of the extensive canyon

systems. Documentation on local landslides is sparse. Talus deposits are present at the base of cliffs above the Rio Grande in White Rock Canyon, and rock slides are found on some steep slopes. In the Frijoles watershed a landslide occurred below the Lower Falls in April, 1942, destroying about 150 yards of trail and blocking access to the Rio Grande (BNM - C. Richey, 14 May, 1942, memo on file). This landslide occurred in the same month as the previously described large blowdown event, and similarly may have been promoted by the record precipitation of that month. Heavy rains on June 19, 1948, again triggered several rock slides in this steep and narrow section of Frijoles Canyon (BNM - F. Binnewies, 23 June, 1948, memo on file). Another landslide event occurred in this area just below Lower Falls in the mid-1970's (J. Lissoway, BNM resource manager - personal communication). Field sampling in 1987 found one fresh mudslide in Upper Frijoles Canyon near point # 675.

Volcanic Activity

Volcanic activity, while outside my primary time-frame of interest, merits recognition as one of the central disturbance regimes that has repeatedly shaped the landscape of the Jemez Mountains through the ages (Burton 1982). Although the last significant volcanic event, represented by deposits of El Cajete pumice, is now dated at ca. 140,000 years B.P. (J. Hawley, New Mexico Bureau of Mines and Mineral Resources geologist - personal communication), low-frequency but extremely high-intensity volcanic disturbances will likely recur in the Jemez Volcanic Field someday (Burton 1982).

ANTHROPOGENIC DISTURBANCES

Livestock Grazing

The grazing of large numbers of domestic livestock may be viewed as a widespread, high-frequency, moderate-magnitude disturbance regime that has had large cumulative impacts upon the Jemez Mountains. It is certain that sheep-raising became the economic focus of the early historic period locally, but the early grazing history of the Jemez Mountains landscape remains uncertain (Denevan 1967). The Spanish introduced livestock into the adjoining lowlands in 1598, and it is possible that regional sheep populations grew to large numbers during a "golden age of relative peace and prosperity" between roughly 1790 and the 1830's (Denevan 1967). Yet, in general, the dangers of Navaho and Apache attacks seem to have restricted intensive livestock utilization of the Jemez Mountains to peripheral areas near settlements like Cañada de Cochiti and those of the Jemez, Chama, and Rio Grande valleys prior to the early 1860's (Scurlock 1981, Rothman 1989). Area shepherds were still being killed by Navahos in 1856, and U.S. military campaigns against the Navaho and Apache continued locally into the 1860's (Scurlock 1981). Minimal livestock utilization of this landscape prior to 1860 is supported by the almost complete absence of historic human artifacts from that time period on the Pajarito Plateau, despite its nearness to lowland settlements (R. Gauthier - personal communication).

After Indian attacks subsided, livestock grazing spread across the landscape and into the interior portions of the Jemez Mountains. The development of local railroad links to external markets by 1880 led to a boom in livestock numbers (Rothman 1989). The Ramon Vigil Grant on the Pajarito Plateau was leased to a Texas stockman from 1885 to 1887

who grazed 3000 cattle on 13,000 ha., about ten times its modern carrying capacity (Foxx and Tierney 1984). Statewide livestock populations rose to about 5,000,000 sheep and 1,000,000 cattle in the late 1880's (Denevan 1967, Carlson 1969). Livestock grazing was apparently ubiquitous across the unfenced forests and rangelands of the Jemez Mountains landscape. While undated, past grazing is indicated by the thick layers of sheep dung commonly found in natural caves and Anasazi cavates where sheep were penned at night by shepherds throughout low elevation canyon portions of the Pajarito Plateau (personal observation). Large livestock populations persisted into the early 20th century; e.g., up to 200,000 sheep and several thousand cattle were grazed annually during the decade before 1918 in the 40,500 ha Baca Location No. 1 in the center of the Jemez Mountains (Scurlock 1981).

Since about 1920 there has been a transition from sheep to cattle grazing in the Jemez Mountains for economic reasons, as well as reductions in stocking rates associated with the increasing control asserted by the U.S. Forest Service over much of the area (Allen 1984-a, deBuys 1985). The lands of LANL have been closed to grazing since 1943, and in recent years 3000 - 4000 head of cattle have utilized the Baca Location at overall stocking levels below that of adjacent USFS allotments (Rowland et al 1983), although the Baca grazing is relatively uncontrolled and thus particularly damaging to riparian zones and water quality (J. Piatt, New Mexico Environmental Improvement Division - personal communication). Most of the Jemez Mountains landscape continues to be grazed by cattle, including the USFS San Pedro Parks and Dome wilderness areas.

Livestock grazing was officially discontinued at Bandelier when the NPS assumed control from the USFS in 1932. When the NPS took over it found that "the whole area was heavily grazed", with "15 or more corrals, watering tanks, drift fences and other facilities to encourage grazing" present in the main section of the monument (BNM - 13 June, 1945, land ownership record on file). Trespass livestock grazing was a major park concern throughout the 1930's and 1940's, with the greatest problems occurring along the Rio Grande and tributary canyon mouths, the area south of Alamo Canyon, and the Otowi Section (BNM - records on file). Boundary fencing has reduced the numbers and impact of trespass livestock, but incidents continue to the present date. For example, in October of 1979 71 head of trespass cattle were rounded up, and in January of 1989 four cattle were observed in the park (BNM - records on file).

Livestock grazing has acted as a disturbance in three major ways in the Jemez Mountains. First of all, the extremely high historic stocking rates apparently led to gross alterations in the species composition of local vegetation associations, as they have elsewhere in the Southwest with similar grazing histories (Leopold 1924, Buffington and Herbel 1965, McGraw 1985). Cool-season grasses and other preferred forage species underwent reductions (Bohrer 1975), while unpalatable and weedy species like snakeweed (Gutierrezia sarothrae) increased in importance (Koehler 1974, Potter 1985, Gonzales et al - manuscript). Livestock also altered vegetation composition by serving as a vector for the introduction of alien plant species throughout the landscape (discussed further below). For example, grazed montane meadows in the Jemez landscape are now often dominated by European plants like

Kentucky bluegrass (Poa pratensis), red clover (Trifolium pratense), and dandelion (Taraxacum officinale) (personal observation, also c.f. deBuys 1985).

Secondly, the continuous, high-intensity grazing that occurred in the past also led to marked reductions in herbaceous plant and litter ground cover - thus overgrazing has been widely considered a major cause of soil erosion and arroyo cutting in the Southwest in general (Leopold 1924, Bahre and Bradbury 1978) and the upper Rio Grande Basin in particular (Cooperrider and Hendricks 1937, Smith 1953, deBuys 1985), although climate fluctuation has been considered an important or even over-riding co-factor by some authors (Leopold 1951, Hastings and Turner 1965, Denevan 1967, Graf 1986). The severe soil erosion observed across piñon-juniper portions of the Jemez Mountains (discussed further below) was likely initiated by the disturbance of past livestock grazing (Earth Environmental Consultants 1978, Potter 1985, Gonzales et al - manuscript, Rothman 1989).

Finally, the fire history record for the Frijoles watershed strongly suggests that overgrazing in the late 19th and early 20th centuries effectively suppressed previous surface fire regimes throughout this landscape (discussed above in **NATURAL DISTURBANCES: Fire**). The striking cessation of frequent surface fires several decades before active anthropogenic fire suppression began (Fig. 4-10) probably occurred due to marked reductions in the biomass and continuity of herbaceous fuels induced by extreme grazing pressures. This landscape-wide reduction in fire events apparently testifies to the ubiquity of local livestock impacts.

Altered Fire Regimes

Fire regimes in the Jemez Mountains have been greatly altered by human influences (primarily discussed above in **NATURAL DISTURBANCES: Fire**). This heading is retained here only to emphasize the importance of altered fire regimes as an anthropogenic disturbance with pervasive ecological repercussions throughout this landscape that are treated in detail below under LANDSCAPE PATTERNS THROUGH TIME.

Biocide Use

Large quantities of the chlorinated hydrocarbon DDT have been applied across the Jemez Mountains since 1950. USFS operations against western spruce budworm sprayed 1,133,622 pounds of DDT on 478,422 ha of the Santa Fe and adjacent Carson National Forests between 1955 and 1963, including an unspecified fraction on the Jemez Mountains (Brown et al 1986). The repeated use of biocides, especially DDT, to control insect outbreaks in Bandelier between 1952 and 1962 is displayed in Table 4-12. The public controversy that followed the 1962 publication of **Silent Spring** by Rachel Carson led to a brief ban on NPS use of chlorinated hydrocarbon biocides in 1963, and a permanent ban in 1964 (H. Baker - 12 June, 1964, memo on file at BNM). Yet, at least as late as 1967 chemical control of fall webworm was being advocated in Bandelier (P. Buffman - memo on file). Since then the NPS has moved to a policy that uses biocides only as part of an integrated pest management approach, and which avoids all use of biocides where possible (USDI National Park Service 1989). Since 1984 the USFS in the Southwest Region has been operating under a memorandum of understanding, stipulated by an out-of-court settlement of a lawsuit,

TABLE 4-12. Known use of biocides for control of forest insects, Bandelier National Monument, 1952-1967.

Note:

These park records are complete for chemical control efforts on forest insects from 1957 to 1966. More detailed information is available for most of these years, including in many cases the concentration of DDT applied. However, these records are fragmentary pre-1957 and post-1966.

Date	Target Pest	Compound	Notes
5/52	? (but see below)	DDT	Lower Frijoles Canyon, aerial

A 8/14/53 document claims that DDT was sprayed annually for "several years" before 1953 in Frijoles Canyon "to protect the scenic value of the broad-leaved trees" from tent caterpillars and fall webworm. At first a hydraulic sprayer was used, but "in the last few years aerial spraying has replaced the ground operation".

5/18/53	tent caterpillar fall webworm	DDT	Lower Frijoles Canyon, aerial, 300 acres, 85 gallons formulated
5/15/55 - 6/15/55	tent caterpillar fall webworm, boxelder bug	DDT	Lower Frijoles Canyon, aerial
5/30/57	Dendroctonus	DDT	headquarters, 455 PP trees
7/15/57	Dendroctonus	DDT	headquarters, 745 PP trees, 38 acres
5/58	Dendroctonus	DDT	headquarters, 1340 PP trees
5/58	Ips	DDT	headquarters/Otowi, 3530 pinyon trees
7/58	Dendroctonus	DDT	headquarters, 1340 PP trees
1960	Dendroctonus	EDB	park "area", 2 trees only
1961	"bark beetles"	EDB	campground, 2 trees only
5/22/61-5/26/61	Ips	EDB	LANL land along Hwy 4, 6 acres, 200 trees
7/30/62	fall webworm	DDT	Frijoles Canyon, aerial (helicopter), 50 acres
8/18/66	fall webworm	Sevin	Frijoles Canyon, 1 colony

that brings aerial spraying of chemical pesticides more in line with an integrated pest management approach (Brown et al 1986). Carbaryl is currently the chemical insecticide most likely to be used by the USFS for control of spruce budworm (Brown et al 1986).

Past biocide applications in the Jemez Mountains have left persistent impacts. Treating the "symptoms" of severe spruce budworm outbreaks with biocide suppression promoted insect resistance (Dahlsten 1983) and ignored the origin of outbreaks in altered forest structure and species composition (Swetnam and Lynch - in review) - an approach which sets up increasingly severe future outbreaks (Brown et al 1986). Peregrine falcons, recorded as nesting in the Bandelier portion of Alamo Canyon in 1934 (B. Thompson - 18 April, 1934, report on file), suffered massive reproductive failure because of egg shell thinning due to bioconcentration of DDT. While peregrines have not nested within Bandelier for decades, the population has recovered somewhat in surrounding portions of the Jemez Mountains (T. Johnson - personal communication); still, nesting success remains precarious due to continued use of chlorinated hydrocarbons on the falcons' Latin American wintering grounds. Sediment and fish samples of Lower Frijoles Creek taken in 1976 revealed significant levels of DDT contamination, indicative of an ongoing pollution problem from the headquarters area that has received no further attention (M. Fletcher-17 Nov., 1976, report on file). Chemical analysis of recent bat collections from a breeding colony of Tadarida brasiliensis in Bandelier reveals significant concentrations of DDT and other contaminants in some individuals (D. Dwyer - 16 Dec., 1988, memo on file); these concentrations are also likely to have been imported to

the Jemez Mountains from overwintering sites in Latin America. The full impacts of past and potential biocide use in the Jemez Mountains remain largely unassessed.

Human-Caused Extirpations

Historic overhunting, systematic predator control, and habitat alteration have led to the local extinctions of a variety of animal species in historic times. In 1943 Bandelier's custodian listed 13 species of terrestrial vertebrates that had probably been extirpated from the park (C. Thomas - 3 March, 1943, memo on file); five of those species are now found in the park due to recolonization, reintroduction, and expansion of park boundaries. Overhunting, combined with livestock competition and diseases, had eliminated elk and bighorn sheep (Ovis canadensis) in the Jemez Mountains by the early 1900's; elk have been successfully reintroduced (see **Large Vertebrate Disturbances** above), and bighorn sheep reintroduction to White Rock Canyon has been studied recently (Grunigen et al 1980). Pronghorn (Antilocapra americana) were thought to have been present in historic times on the Pajarito Plateau (Marchiando 1977). While its presence here was never confirmed (U.S. DOE 1980), the black-footed ferret (Mustela nigripes) may have been locally extirpated by prairie dog (Cynomys gunnisoni) control efforts. Trapping and habitat alteration may have decimated three other mustelid species for which no firm Jemez Mountains records exist: the pine marten (Martes americana) from high-elevation conifer forests (New Mexico Dept. of Game and Fish 1985); the mink (Mustela vison) from the Rio Grande (Guthrie and Large 1980); and the otter (Lutra canadensis) from the Rio Grande (C. Thomas - 3

March, 1943, memo on file at BNM).

Perhaps the most ecologically significant local extinctions involved large carnivores. Hunting and predator control programs eliminated the grizzly bear (Ursus horribilis) in the Jemez Mountains (Marchiando 1977) sometime before statewide extirpation in the 1930's (Findley 1987). The last gray wolf (Canis lupus) in northern New Mexico was reputedly killed in the Jemez Mountains in 1932 (Scurlock 1981), eliminating an important regulator of ungulate populations. Vigorous efforts were also made to eliminate the mountain lion (Felis concolor); nine lions were killed around the headwaters of Frijoles Creek in one week in 1932, and at least 50 lions were killed in the Jemez Mountains between 1931 and 1934 (A. Borell - Nov., 1934, report on file at BNM). Mountain lions persist in low numbers in the Jemez Mountains, despite continued hunting pressure that killed at least six lions locally from 1987-1988 (New Mexico Dept. of Game and Fish records on file). The decimation of large predators leaves responsibility for control of local deer and elk populations largely in human hands.

Plant extirpations from the Jemez Mountains are undocumented. However, Jacobs (personal communication) finds that six plant species have been lost from Bandelier due to habitat alteration along the Rio Grande by the Cochiti Reservoir, namely the western cardinal flower (Lobelia cardinalis), helleborine orchid (Epipactis gigantea), water smartweed (Polygonum amphicum), silverweed (Potentilla anserina), yerba-mansa (Anemopsis californica), and mountain water-parsnip (Cymopterus montanus).

Introduction of Alien Species

Deliberate and inadvertent human introduction of alien species has greatly altered the ecology of the Jemez Mountains landscape. The significant landscape-wide impacts that have resulted from the introductions of large populations of sheep, cattle, and burros are described above. Small wild horse herds exist in the Caja del Rio and around Polvadera Peak.

Widespread introduction of alien trout species has greatly diminished local populations of the native Rio Grande cutthroat trout (Salmo clarki virginalis), although relatively pure genetic populations persist in Polvadera Creek and Peralta Canyon. It is possible, but not certain, that native cutthroat trout existed prehistorically in Frijoles Creek. The magnitude of alien fish introductions into the Jemez Mountains can be appreciated by reviewing the history of fish planting in Bandelier streams. New Mexico Dept. of Game and Fish records on file at BNM show that 36,750 brook trout (Salvelinus fontinalis), 82,740 rainbow trout (Salmo gairdneri), and 368,404 cutthroat trout of Yellowstone origin were planted in Frijoles Creek between 1912 and 1955. Alamo Creek received 13,000 brook trout, 4,000 rainbow trout, and 6,000 cutthroat trout between 1919 and 1931, while Capulin Creek received 10,500 brook trout, 17,000 rainbow trout, and 1,500 cutthroat trout between 1922 and 1931. Undocumented introductions of brown trout (Salmo trutta) have also occurred in these streams. Cutthroat are no longer found in the park, while alien trout species persist as top carnivores with undocumented impacts on the ecology of these streams and their rich aquatic invertebrate fauna (Pippin and Pippin 1980, 1981).

Alien plant introductions are even more numerous and ubiquitous in the Jemez Mountains. The persistence of vegetation overstories dominated by native woody species masks a great deal of unrecognized compositional change in the understory flora of this landscape. The pervasive conversion of montane meadows into alien-dominated pastures by livestock grazing is discussed above. Botanist Brian Jacobs (personal communication) estimates that 20% of the 720 species of vascular plants found in Bandelier are aliens. Alien species of special concern at BNM include the tree-of-heaven (Ailanthus altissima) at several canyon sites, tamarisk (Tamarix pentandra) along the Rio Grande, and common mullein (Vervascum thapsus) across much of the park (BNM - 1989, draft Statement for Management on file).

Alien plant species and genotypes continue to be introduced across this landscape. Range improvement and erosion control projects on USFS and private lands routinely use alien species to establish ground cover. Tree planting efforts on USFS and DOE lands in the late 1970's introduced non-local genotypes of ponderosa pine across extensive areas. Similar introductions of non-native genotypes almost certainly occurred in Bandelier with seeding of developed areas, seeding of several small burns in 1965, and reforestation efforts in 1963 and 1964; these introductions are mentioned without details in annual forestry reports on file at BNM. Within one month of the 1977 La Mesa Fire the NPS aerially seeded the 4305 ha of burned BNM land with 44,242 kg of grass seed to minimize expected erosion problems; this was an application rate of 538 - 646 grass seeds/m^2 (Potter and Foxx 1979-a). While every effort was made to secure seed of native species, the seedstock was not from the Jemez Mountains, and it was later discovered

that the sheep fescue (Festuca ovina) in the 6-species mix may have been a European cultivar. The seeded slender wheatgrass (Agropyron trachycaulum) and sheep fescue now dominate many portions of the burn area (Potter and Foxx 1979-a), with sheep fescue especially prominent on Apache Mesa (personal observation). Further, the mix contained 0.19% unspecified weeds, which translates into 84 kg of weed seed dumped across one-third of the park with unknown impacts. Numerous species of alien agricultural weeds from upstream areas currently dominate the area along the Rio Grande flooded by the high waters of the Cochiti Dam reservoir in 1987. Horses apparently contribute to the spread of alien species in Bandelier, e.g. awned dogtail (Cynosurus echinatus) and joint goatgrass (Aegilops cylindrica), as these are only found near the park stable and along certain park trails (B. Jacobs, personal communication).

Forest Cutting

Significant anthropogenic cutting of piñon and juniper trees for cooking, heating, and building purposes almost certainly occurred during periods of Anasazi occupation in the Jemez Mountains (c.f. Kohler and Matthews 1988), but these prehistoric impacts remain locally undocumented. Ongoing archeological survey and excavation work in Bandelier may shed some light on this question.

Commercial logging began in peripheral portions of the Jemez Mountains in the late 1800's and continues to the present date throughout this landscape. The Ramon Vigil Grant on the Pajarito Plateau was intensively logged from 1897-1903 by H.S. Buckman (Foxx and Tierney 1984, Rothman 1989). Early logging activities are documented

by Scurlock (1981), Foxx and Tierney (1984), and Rothman (1989). The now-dissolved New Mexico Timber Company was an important agent of Jemez Mountains logging activity for decades, including operations in and around the upper portions of current Bandelier land in the 1940's (C. Thomas - 3 July, 1942, memo on file at BNM) and 1950's (BNM - March, 1979, briefing statement on file). The New Mexico Timber Company also used a timber rights lease to engage in the destructive clearcutting and roading of large portions of the Baca Location in the 1960's and early 1970's (USDI National Park Service 1979), which was finally stopped after legal action by the landowner led to an out-of-court settlement. Most logging activity in the Jemez Mountains today occurs on USFS lands. The Santa Fe National Forest Plan calls for harvesting 39 million board feet of timber annually through 1997, with the bulk of it coming from the Jemez Mountains (USDA Forest Service 1987-b).

Logging acts as an ecological disturbance at scales ranging from short-term and site-specific to cumulative and landscape-wide in the Jemez Mountains. Specific logging impacts include: soil compaction (Clayton **et al** 1987); increased surface water runoff and erosion (Swanson **et al** 1982); logging road effects (see below); drastically altered patterns of nutrient cycling (Johnson **et al** 1982), including diminished site nutrient pools due to export of harvested trees (Stark 1988) and the loss of nutrient capital, ectomycorrhizal activity, cation exchange sites, and nitrogen fixation associated with diminished pools of soil organic materials (Harvey **et al** 1987); increased incidence of insect (Dahlsten and Rowney 1983) and disease problems (Hessburg and Beatty 1986, McDonald **et al** 1987); narrowed genetic diversity of tree species (Ledig, 1986, Rehfeldt 1987); altered

disturbance regimes (Franklin and Forman 1987); and habitat loss for endangered species (Morse et al 1986). Logging impacts to archeological sites are a significant issue in the Jemez Mountains - a moratorium on new USFS timber sales in the Jemez Mountains occurred in 1985/early 1986 until a suit filed by the State of New Mexico and other appellants over logging damage to archeological sites was resolved. Currently the impact of logging upon the landscape patterns (Harris 1984, Franklin and Forman 1987), structural characteristics (Franklin 1988), dependent organisms (Simberloff 1987), and associated processes (Maser and Trappe 1984) of old-growth forests is emerging as one of the greatest ecological concerns in the Jemez Mountains (Audubon Society et al 1988) and other western landscapes (The Wilderness Society 1988, Maser 1988).

Extensive forest and woodland cutting has occurred within Bandelier's current boundaries over the past 90 years, although the selective nature of past tree cutting, subsequent forest growth, and the obliteration of many old stumps by the La Mesa Fire have acted to obscure much evidence and minimize the impression of past forest harvesting activities within the park. My 1987 field sampling of the Frijoles watershed (Figure 4-26) found 146 points where selective logging had occurred over 20 years ago, 29 points where old selective woodcutting had taken place, 21 points where snags (presumably) had been felled over 15 years ago, eight points where Bandelier management activities had cut trees (primarily invasive trees in grasslands and meadows), six points where trees where cut as part of fire-fighting efforts, 24 points on USFS land where salvage logging had occurred after the La Mesa Fire (in 1977 on Mesa del Rito), and 19 points where

Figure 4-26. Map of 1987 sample point locations in the Frijoles watershed with evidence of past tree cutting. The gray patch is the entire sample area, with drainages outlined in blue. Selective logging = green points, woodcutting of piñon and juniper species = magenta points, tree cutting to fight fires = black points, salvage logging on SFNF lands after the 1977 La Mesa Fire = yellow points, recent SFNF timber sales = red points, and BNM tree cutting for management purposes (old felling of snags at low elevations and grassland/meadow maintenance at high elevations) = blue points.

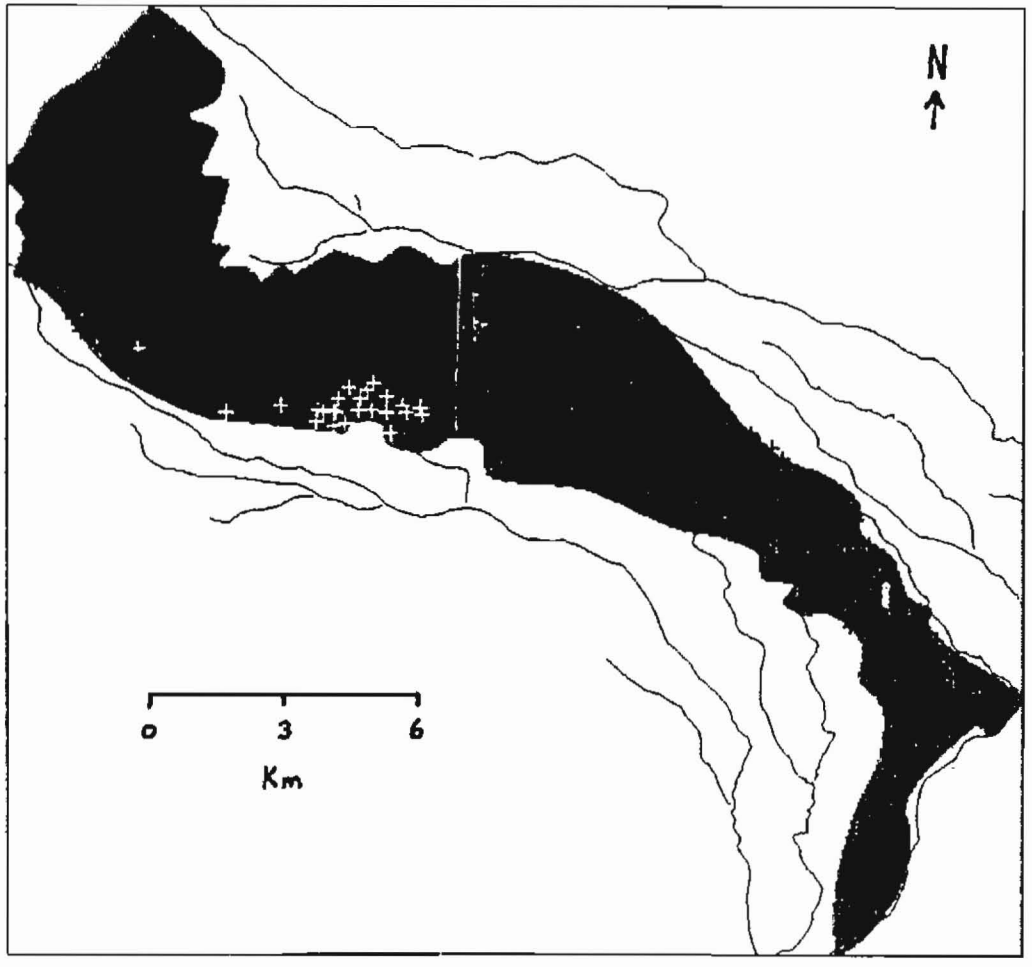


Figure 4-26

USFS timber sales had taken place in the previous five years. Overall, 289 of the 969 sample points (29.8 %) displayed evidence of past tree cutting. Ponderosa pine forests north of Frijoles Canyon and as high as Apache Mesa were high-graded in the Buckman era. Fire-scar Sample #20 is a stump from an elevation of 2452 m. with a cutting date of 1897; the cutting of this tree just west of the Ramon Vigil Grant that Buckman leased may reflect illegal cutting or other logging operations close to a nearby former wagon road (now Highway 4). Thinning operations in dense ponderosa pine reproduction were conducted by the Soil Conservation Service in 1937 on portions of the Ramon Vigil Grant north of the rim of Frijoles Canyon (W. Yaeger - 1 July, 1937, memo on file at BNM). The dating of trees growing in old logging roads or inside of old stumps and barely discernible traces of skid roads on the 1935 air photos indicate that selective logging was occurring by the 1930's on upper Apache Mesa and the southeast flank of Cerro Grande. Fire-scar sample #'s 73 and 61 are stumps from the south-central portions of Cerro Grande that were cut in 1948 and 1958 respectively-much of the logging in this area appears to have occurred in the 1950's. A 1933 trail report (W. Attwell - 13 May, 1933, on file at BNM) details the widespread and intensive woodcutting of juniper (primarily) and piñon across the detached Otowi Section of Bandelier. As late as 1962 the newly acquired area north of the Frijoles Canyon rim was fenced along State Highway 4 in part to deter "wood haulers" (BNM - 1962 annual forestry report on file). Annual forestry reports reveal that beetle-killed snags were felled each year from 1957 - 1966 in order to reduce the perceived fire hazard. While most of the cut snags were left where felled, the annual forestry reports show that

these snags provided a yearly average of 10,537 board feet of timber for park use as fuelwood and lumber from 1957 - 1965. Tree cutting in Bandelier today is basically restricted to individual hazard trees in developed areas or along trails, trees cut in fire-fighting operations, and limited efforts to maintain remnant open grasslands and meadows free of modern tree invasion.

Bandelier continues to be affected by logging activities on adjacent lands. Since 1983 the USFS-managed headwaters of Alamo and Capulin Canyons west of the park boundary have been logged, with another 2+ million board feet of timber tentatively planned to come out of this area and the headwaters of Sanchez Canyon 1991. In 1983 Bandelier was forced to permit the USFS to open and use an access road across the park to cut 1.1 million board feet of old-growth mixed conifer forest adjacent to the monument boundary (BNM - numerous memos on file), while additional USFS logging has occurred in the 1980's along the north boundary of Bandelier on Apache Mesa. Logging has occurred along about 11 % of Bandelier's main unit boundaries in the last 12 years.

Commercial logging will continue to be one of the most significant and controversial influences on the the Jemez Mountains landscape in coming years. Table 14 in the approved Santa Fe National Forest Plan (USDA Forest Service 1987-c) calls for logging "treatments" to remove 455 million board feet of timber from 43,037 ha in the first decade of implementation, which is 30.5% of the area identified as suitable for harvest, or 8.34% of the non-wilderness portion of the forest. Low-resolution USFS maps of the timber sale planning areas received by a local environmentalist under the Freedom of Information Act (S. Hitt-

personal communication) indicate that an even larger fraction of the Jemez Mountains portion of this national forest will be logged by 1997 (Figure 4-27). The difficulties inherent in meeting its conflicting mandates to both extract resources and maintain site productivity/diversity/aesthetics make it challenging for the USFS to meet its overall resource management goals (USDA Forest Service 1987-b, 1987-c) with this level of forest entry.

Roads

The landscape disturbance effects of road development can include greatly accelerated erosion rates (Burroughs and King - unpublished manuscript), increased stream sediment loads (Rice et al 1979), reduced landscape productivity due to replacement of natural sites by sterile roads (Maser 1988), avoidance behavior by deer and elk (Rost and Bailey 1979), decreased abundances of some bird species (van der Zande et al 1980), and increases in avian nest predation (Small and Hunter 1988). Human road traffic in the Jemez Mountains directly disturbs certain nesting raptors like peregrine falcons (Johnson 1986) and zone-tailed hawks (G. Schmidt, New Mexico Game and Fish - personal communication), while roads and associated road-kill mortality may inhibit the dispersal of some small animals like certain mice (Mader 1984) and salamanders (C. Painter, New Mexico Dept. of Game and Fish Biologist-personal communication). Roads act directly as fire breaks and allow widespread access for fire suppression. The extensive road systems in the Jemez Mountains were largely developed to support logging activities; these roads are heavily used by off-road vehicle enthusiasts, hikers, campers, and hunters. In general, roads can be

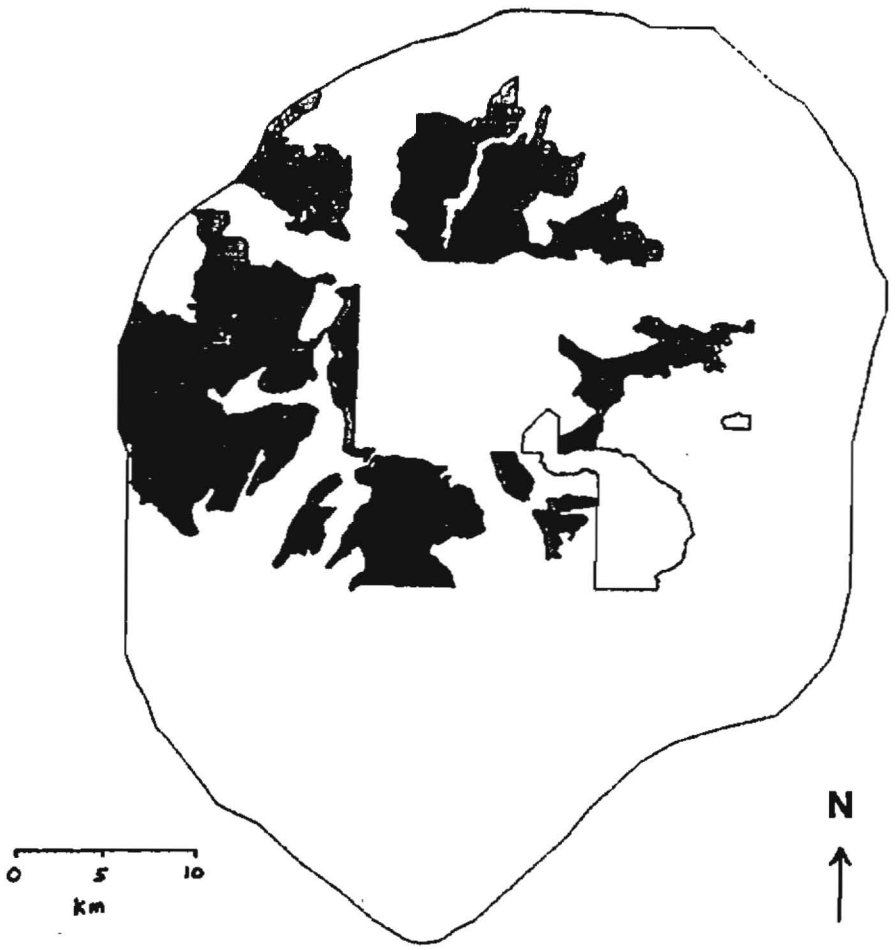


Figure 4-27. Map of planned timber sale areas on the Santa Fe National Forest in the Jemez Mountains between 1988 and 1997. The outlines of the Jemez Mountains and Bandelier National Monument are shown with lines, and the planned timber sale areas are displayed as shaded patches.

viewed as a signature of certain human impacts on this landscape. The striking development of road networks in the Jemez Mountains is discussed below under LANDSCAPE PATTERNS THROUGH TIME, MAPPED LANDSCAPE CHANGES: 1935-1981, Road Networks.

Accelerated soil erosion

Accelerated soil erosion in the upper Rio Grande Valley, including the Jemez Mountains landscape, has long been recognized to be a result of anthropogenic disturbance, attributable in large part to overgrazing by livestock (Cooperider and Hendricks 1937, Smith 1953, DeBuys 1985, Rothman 1989). Such soil erosion is especially prevelant at lower elevations on piñon and juniper woodland sites throughout the Jemez Mountains (B. Simms, USFS hydrologist - personal communication).

Excessive soil erosion has been recognized as a problem at Bandelier for decades. A 1948 assessment of "needs for soil and water conservation" lists 6000 total acres of "problem areas", with sheet erosion on 4000 acres, "excessive run-offs" on 1000 acres, "potentially serious gullies" on 1500 acres, "large or critical gullies" on 200 acres, and impacts from past overgrazing on 2000 acres (BNM - report on file). This assessment lists 2000 total acres in need of improvement work, and it calls for 1000 check dams across 2000 acres, "waterspreading" on 500 acres, "contouring" on 100 acres, "seeding-planting" on 50 acres, and "brush matting" on 50 acres. An accompanying memo from the park superintendent states:

> "Over grazing by wild burros along the Rio Grande has materially reduced the grass cover in that area of the Monument and there is some erosion starting as a result of this... There is some sheet erosion on the tops of the mesas and a few gullies that have started on the sides of the canyons. It is estimated that there is approximately 6,000 acres in the Monument that is in need of

> attention although none of it is severe at present" (F. Binnewies - 13 Jan., 1949, memo on file at BNM).

Apparently no action resulted from this assessment. Field sampling did find one sturdy check dam in a Burnt Mesa drainage (near pt. # 96), but this area was not added to the park until 1959.

From at least 1955 to 1966 Bandelier coordinated Boy Scout placement of discarded local Christmas trees in gullies in the Otowi/Tsankawi area for erosion control purposes (BNM - annual forestry reports on file). Storm runoff on June 18, 1965, flooded the visitor center, leaving sediment and debris throughout the building (BNM - 1965 annual forestry report on file). This flood prompted the establishment of an agreement with the Jemez District of the Soil Conservation Service to provide technical assistance to the park (BNM - 1966 annual forestry report). The only known outcome of this agreement was a preliminary study of erosion control on the canyon slopes above the headquarters complex (BNM - 1966 annual forestry report on file).

Bandelier's erosion concerns in the 1970's were associated with impacts from the large feral burro population (see discussion above). A soil survey of the affected piñon-juniper portions of the park made a crude soil erosion rate estimate of 0.53 cm/year, or 79.3 megagrams/ha/year (35.7 tons/acre/year) (Earth Environmental Consultants 1978). This estimate was based upon assumed rates of pedestalling and may not be accurate, but the erosion rate it implies of 53 cm /100 years reflects the obviously unsustainable net soil loss that was observed in many areas.

Erosion continues to be a problem at Bandelier, even in areas north of Frijoles Canyon that were never affected by significant burro populations. The ongoing archeological survey began recording erosion

impacts to archeological sites during the 1988 field season, and they found 307 of 390 total sites surveyed being affected by erosion, with 145 sites affected by gullying (Orcutt and Powers 1989). My fieldwork in the Frijoles watershed found evidence of ongoing accelerated soil erosion at 121 points, 12.5% of the total sampled, and recently-stabilized past erosion at an additional 16 points (Figure 4-28). Sheet erosion is occurring across much of the piñon-juniper woodland and juniper savanna areas in the park, with gullies present where the soils are deep enough to allow them to form. Pumice soils display less erosion. Southern portions of the park not covered by my point sampling display even worse erosion, except for the increasingly large areas where little more than bedrock remains (personal observation). Evidence of recently-stabilized past erosion was observed at 35 total points, largely at sites severely burned by the 1977 La Mesa Fire. Some erosion is observed from sites disked for site preparation by the USFS on Mesa del Rito. Most excess erosion at higher elevations in the Jemez Mountains today appears to be associated with roads and logging activities, although overgrazing of montane grasslands on Polvadera Peak and near Cerro Pelado has caused recent erosion (Allen 1984-a). Little soil erosion is currently observed from heavily vegetated high-elevatation sites in the Frijoles watershed, although there are scars of old gullies in the montane grasslands around the caldera rim that hint of accelerated erosion during previous episodes of livestock overgrazing.

Dams

The construction of large federal dams on the Rio Grande at

Figure 4-28. Map of 1987 sample point locations in the Frijoles watershed displaying evidence of accelerated erosion. The black line outlines the entire sample area. The colored patches are from the 1981 ecosystem patch map, with juniper woodlands/savannas = yellow, piñon-juniper woodlands = green, and grassland and grass/shrubland = gray. Severe ongoing erosion = red points, moderate ongoing erosion = black points, and past severe/moderate erosion = blue points.

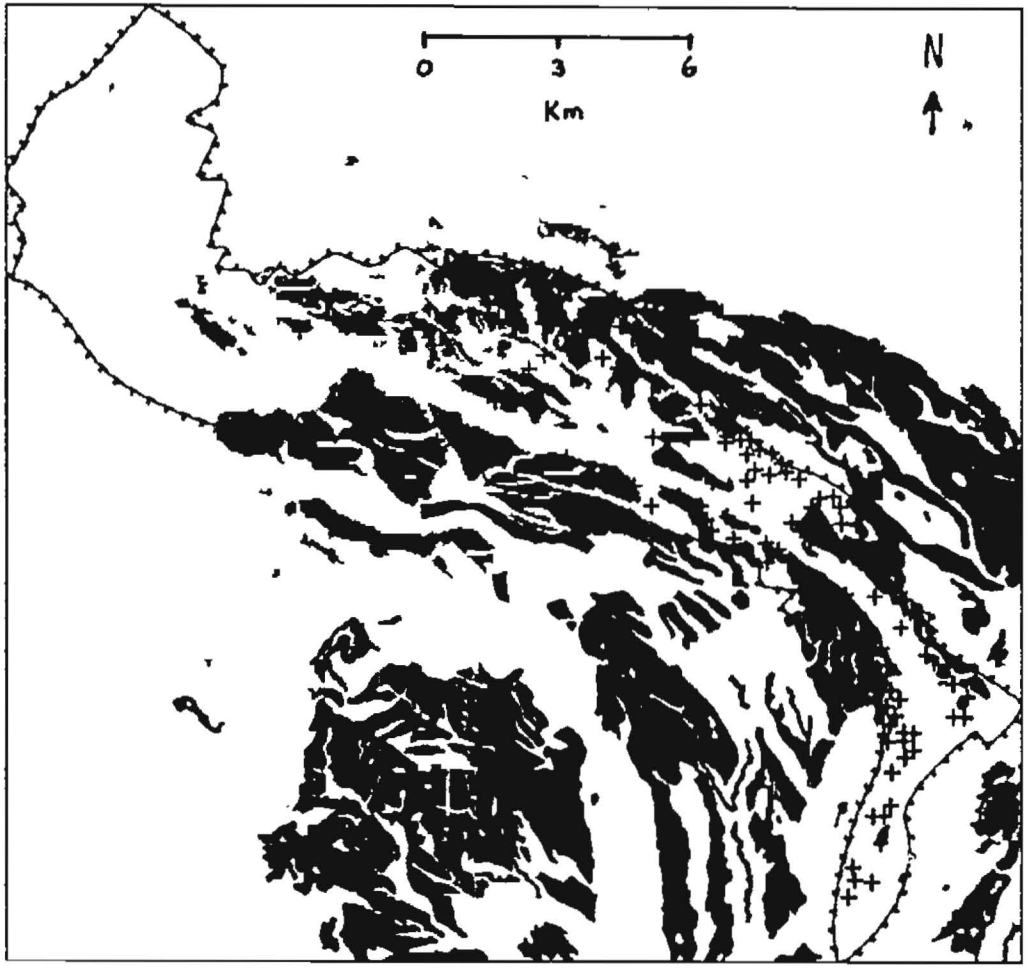


Figure 4-28

Cochiti and the Rio Chama at Abiquiu has had major impacts upon the Jemez Mountains landscape. Plans existed for a dam in White Rock Canyon at least as far back as 1942 (C. Thomas - 5 March, 1942, memo on file at BNM). In 1975 the Cochiti Dam, an earth-fill structure 8.8 km long, was completed for flood control purposes by the U.S. Army Corps of Engineers on the Rio Grande just downstream from Bandelier. Cochiti Dam has greatly altered the natural flooding regime of the Rio Grande; the reservoir backs up into White Rock Canyon, drowning up to 350 acres of Bandelier's eastern boundary along the Rio Grande and tributary canyon bottoms at its maximum flood pool level of 1667 m. The peak flood pool elevation reached to date was 1657.5 m on June 24, 1987. Long-term carryover of flood waters occurred from 1985 through early 1988, inundating portions of the flood pool for most of this period. Notable impacts of this flooding included the killing of basically all pre-flood plant life, the introduction of tamarisk and other weedy alien species to the flooded zone, the deposition of drift litter along miles of shoreline, wave erosion, massive slumping along the western cliffbank, and the choking of the former floodplain with many feet of sediment (BNM 1989). Sediment burial of the floristically diverse spring at the mouth of Frijoles Canyon has apparently caused the direct extirpation of a half dozen plant species from the park (B. Jacobs-personal communication). Water seepage from this reservoir has also forced the abandonment of many downstream farm fields at Cochiti Pueblo.

The ecological character of the affected portions of White Rock Canyon will continue to change as the Cochiti Reservoir gradually fills with sediment. The authorized permanent pool elevation will continue

to rise from its 1987 level of 1626 m, and the growing delta that is developing at its head has the potential to become a valuable wetland habitat for wildlife (Johnson 1987). The development of Cochiti Reservoir has already attracted larger numbers of overwintering bald eagles to this area (Johnson 1988). It is estimated that this dam will be completely filled with sediment within about 500 years (D. Kreiner, U.S. Army Corps of Engineers - personal communication).

Similar landscape impacts have resulted from the operation of the Abiquiu Dam on the Rio Chama, completed in 1963 along the northern boundary of the Jemez Mountains. This flood control reservoir is on a major upstream tributary of the Rio Grande and has contributed to altered stream flows and sediment loads in both atreams. High water levels also occurred in this reservoir from 1985-1987, causing public controversy over the drowning of grazing land and popular whitewater rapids, in addition to lingering concern over the potential for dam failure with catastrophic impacts on downstream communities (Lieu 1986).

Smaller dams have been built throughout the Jemez Mountains. A 1934 report states that: "About 40 wooden dams have recently been placed in Rito de los Frijoles above the Monument boundary. About one-fourth of them have washed out, but each of the others is providing a good hole above and another below the dam (for trout)" (A. Borell-Nov., 1934, report on file at BNM). This section of Frijoles Creek is now a little-used portion of the park, and these dams are no longer apparent. Mapping reveals that 12 stock ponds, covering 16.8 ha, were built across a 85,745 ha portion of this landscape around Bandelier between 1935 and 1981. The impacts of such impoundments upon local

streams and wildlife populations remain undocumented.

Pollution

Various forms of anthropogenic pollution increasingly affect the Jemez Mountains landscape. Local air quality is currently being degraded from several sources. Some long-time residents suspect that the regional haze that has been expanding over much of the Four Corners region has begun to be detectable around the Jemez Mountains. Recent increases in human activities associated with the rapid buildup of local human populations, especially the use of wood fuel for heating, have caused apparent declines in air quality and visibility in the Jemez, Española, and Rio Grande valleys. Views toward the Sandia Mountains are increasingly obscured by burgeoning Albuquerque's infamous "brown cloud".

Contamination of surface and ground water is another escalating local problem, especially in the adjacent Española Valley (Peterson 1988). Untreated sewage, agricultural chemicals, and solid waste (garbage) are polluting the Rio Grande from human activities (BNM 1989). Widespread logging and livestock grazing activities also affect water quality in the Jemez Mountains (USDA Forest Service 1987-b). As one example, repeated sampling of the East Fork of the Jemez River at Las Conchas finds state and federal water quality standards exceeded for allowable temperature and ammonia concentrations, presumably due to past and ongoing livestock operations in the Baca Location that allow cattle free access to the whole streamcourse (J. Piatt, New Mexico Environmental Improvement Division - personal communication).

While largely contained within the boundaries of Los Alamos

National Laboratory, research and development activities since 1943 have resulted in some contamination of this landscape by radioactive and toxic materials. About 300 contaminated sites have been identified at LANL (R. Ferenbaugh, LANL ecologist - personal communication). Environmental monitoring indicates that radioactive contamination of local air and water occurs at only fractional amounts above background levels (Environmental Surveillance Group 1987). Still, cleanup of onsite environmental contamination at LANL has recently been estimated by the Department of Energy to require as much as 2.1 billion dollars between now and the year 2010 (Spice 1989).

Climate Change

It has become increasingly clear that human activities are rapidly altering the composition of the global atmosphere (McElroy and Salawitch 1989) and thus global climate (Schneider 1989). A general trend of warmer and wetter weather is expected for the American Southwest and the Jemez Mountains in coming decades, but despite ongoing research efforts the local expression and impacts of anthropogenic climate change remain speculative (W. Moir, USFS research ecologist - personal communication). Still, the potential clearly exists for major alterations in disturbance regimes and biotic distributions in the Jemez Mountains landscape due to anthropogenic climatic change in the coming century (Roberts 1988).

INTERACTIONS AMONG DISTURBANCES

It is important to recognize that many of the disturbances described above do not function in isolation from each other in the

Jemez Mountains. In particular most of the natural disturbances are tied to weather events or climatic fluctuations. For example, windthrow, lightning, hail, snow, and flooding disturbances are directly related to weather events, while fires and bark beetle outbreaks are associated with periods of low precipitation. Insect outbreaks alter the volume and/or continuity of potential fuels, and thus affect fire regimes. Fire suppression and logging practices have likely altered patterns of spruce budworm outbreaks (Swetnam and Lynch - in review). Livestock grazing has affected fire regimes and accelerated soil erosion, as have logging activities and road development. Rapid climate change would stress existing forests, leading to increased incidence of insect outbreaks and possibly intense fires from the resultant fuel loadings. Disturbances are often interactive in this landscape.

CHAPTER V. RESULTS AND DISCUSSION: LANDSCAPE PATTERNS THROUGH TIME

TREE AND SHRUB SPECIES IN THE RITO DE LOS FRIJOLES WATERSHED

The elevational distributions of 25 tree and 10 prominent shrub species in the Frijoles Watershed, based upon presence at 969 field sampling points, are listed in Table 5-1. Associated topographic potential moisture index (TPMI) values for these same species are given in Table 5-2. Median values are used to order the data in these tables as this measure of central tendency is less biased by outliers or skewed distributions.

Trees and shrubs grow throughout the elevational range of the Frijoles Watershed (Table 5-1). Figure 5-1 shows that the central points of woody species distribution are spread across most of the elevational gradient of the watershed, consistent with an individualistic species distribution view of community organization in the Jemez Mountains (Whittaker 1975, USDA Forest Service 1986).

Woody species TPMI central points are focused around the median value for the watershed (25), except for a number of riparian specialists with median TPMI values at or near 50 (Table 5-2, Figure 5-1). Most of these species are capable of growing across a range of site moisture conditions (Table 5-2), often from xeric sites at high elevations to more mesic sites at lower elevations (e.g. Figure 5-2). Thus median TPMI values near 25 might be expected for many species, especially since the use of presence data may compress the median TPMI distribution by failing to account for differences in species abundance at TPMI's away from the median value.

Integration of these elevation and TPMI data with other site-specific information (e.g. percent slope, aspect) and map locations in

TABLE 5-1. 1987 elevational distribution of tree and shrub species in the Frijoles Watershed, arrayed from highest to lowest median elevation. All elevations in meters. Species abbreviations from Table 3-1. Species labelled *.PAST reflect the presence of the species in the recent past, based on standing and downed woody material as well as live individuals. Several species display both tree and shrub growth forms. N = 969 sample points.

Tree Species	n	Range	Median	Mean	Standard Deviation
ABLA	10	2922 - 3059	2991	2992.8	45.3
PIEN	38	2586 - 3076	2962	2937.9	104.2
ACGL	83	1950 - 3053	2684	2603.3	275.1
POTR	326	1896 - 3059	2678	2635.8	211.9
RONE	20	2109 - 2840	2660	2598.0	207.2
PIPU	62	1995 - 2937	2634	2547.8	233.9
PIFL	281	1877 - 3041	2631	2571.6	230.9
ABCO	400	1777 - 3035	2606	2554.8	248.1
ABCO.PAST	415	1777 - 3035	2596	2546.9	248.3
PSME	516	1777 - 3076	2519	2504.5	287.3
PSME.PAST	590	1777 - 3076	2472	2475.6	284.5
QUGA	209	1653 - 3000	2449	2402.4	299.6
AMBA	5	2243 - 2438	2330	2335.1	72.8
SASC	10	2010 - 2745	2312	2326.4	268.4
PIPO	727	1668 - 3076	2312	2376.4	288.6
PIPO.PAST	790	1650 - 3076	2284	2353.8	290.4
CRER	3	2109 - 2330	2199	2212.8	111.2
PRsp	42	1696 - 2907	2126	2149.8	284.1
JUDE	30	1868 - 2254	2117	2085.7	89.5
BEOC	32	1796 - 2312	2053	2050.5	158.4
QUUN	23	1653 - 2660	2030	2027.8	189.0
JUSC	68	1786 - 2503	2028	2037.0	140.2
JUSC.PAST	94	1786 - 2503	2077	2097.4	157.6
ALTE	42	1745 - 2438	2025	2047.7	191.0
PIED	249	1685 - 2678	2008	2021.6	119.3
PIED.PAST	288	1650 - 2678	2011	2026.9	124.0
JUMO	307	1650 - 2403	1984	1985.1	132.5
JUMO.PAST	357	1650 - 2495	2008	2015.8	149.7
ACNE	58	1653 - 2559	1982	2020.3	205.2
POAN	30	1668 - 2243	1896	1942.3	164.3
PTTR	7	1668 - 2025	1857	1858.3	113.?
Shrub Species					
JUCO	86	2077 - 2946	2684	2680.2	164.4
ACGL	70	1938 - 3053	2666	2589.6	291.5
QUGA	402	1777 - 3029	2330	2391.1	249.7
RONE	352	1668 - 2869	2278	2339.5	228.9
AMBA	6	2010 - 2538	2275	2293.6	196.8
CEFE	97	2138 - 2562	2259	2284.3	98.5
PRsp	71	1696 - 3027	2182	2219.6	281.5
QUUN	359	1812 - 2669	2123	2118.6	140.6
CEMO	181	1860 - 2379	2100	2096.7	99.6
FAPA	78	1665 - 2262	2040	2013.3	134.4

TABLE 5-2. 1987 TPMI distribution of tree and shrub species in the Frijoles Watershed, arrayed from highest to lowest median elevation to match Table 5-1. Species abbreviations from Table 3-1. Species labelled *.PAST reflect the presence of the species in the recent past, based on standing and downed woody material as well as live individuals. Several species display both tree and shrub growth forms. N = 969 sample points.

Tree Species	n	Range	Median	Mean	Standard Deviation
ABLA	10	16 - 30	25	24.3	4.2
PIEN	38	10 - 47	24	24.2	8.2
ACGL	83	12 - 50	32	35.1	11.8
POTR	326	9 - 50	25	28.5	10.9
RONE	20	12 - 50	25	31.1	14.4
PIPU	62	12 - 50	44	39.6	10.8
PIFL	281	7 - 50	25	28.1	12.2
ABCO	400	7 - 50	26	29.1	12.1
ABCO.PAST	415	7 - 50	27	29.2	12.0
PSME	516	7 - 50	26	29.0	12.2
PSME.PAST	590	7 - 50	26	28.8	11.8
QUGA	209	7 - 50	30	31.7	14.4
AMBA	5	44 - 50	50	48.8	2.7
SASC	10	22 - 50	50	42.5	12.2
PIPO	727	5 - 50	25	28.1	11.7
PIPO.PAST	790	5 - 50	25	27.8	11.5
CRER	3	50 - 50	50	50.0	0
PRsp	42	14 - 50	50	46.6	8.7
JUDE	30	5 - 50	23	25.5	11.3
BEOC	32	50 - 50	50	50.0	0
QUUN	23	6 - 47	25	26.7	11.7
PIED	249	5 - 50	22	25.2	11.2
PIED.PAST	288	5 - 50	22	25.0	10.9
JUSC	68	6 - 50	44	34.8	14.8
JUSC.PAST	94	6 - 50	30	31.7	14.4
ALTE	42	50 - 50	50	50.0	0
JUMO	307	5 - 50	22	24.8	11.1
JUMO.PAST	357	5 - 50	21	24.6	10.9
ACNE	58	18 - 50	50	47.6	6.0
POAN	30	50 - 50	50	50.0	0
PTTR	7	30 - 50	50	44.7	9.1
Shrub Species					
JUCO	86	13 - 50	25	28.2	10.9
ACGL	70	12 - 50	33	35.5	11.9
QUGA	402	7 - 50	27	29.3	12.4
RONE	352	7 - 50	27	29.3	11.8
AMBA	6	20 - 50	50	44.0	12.0
CEFE	97	12 - 44	24	24.2	7.5
PRsp	71	8 - 50	47	43.3	11.0
QUUN	359	5 - 47	23	25.3	10.9
CEMO	181	5 - 47	20	23.7	10.4
FAPA	78	13 - 50	23	26.9	11.3

FIGURE 5-1. Central distribution point of each tree and shrub species in the Frijoles watershed.

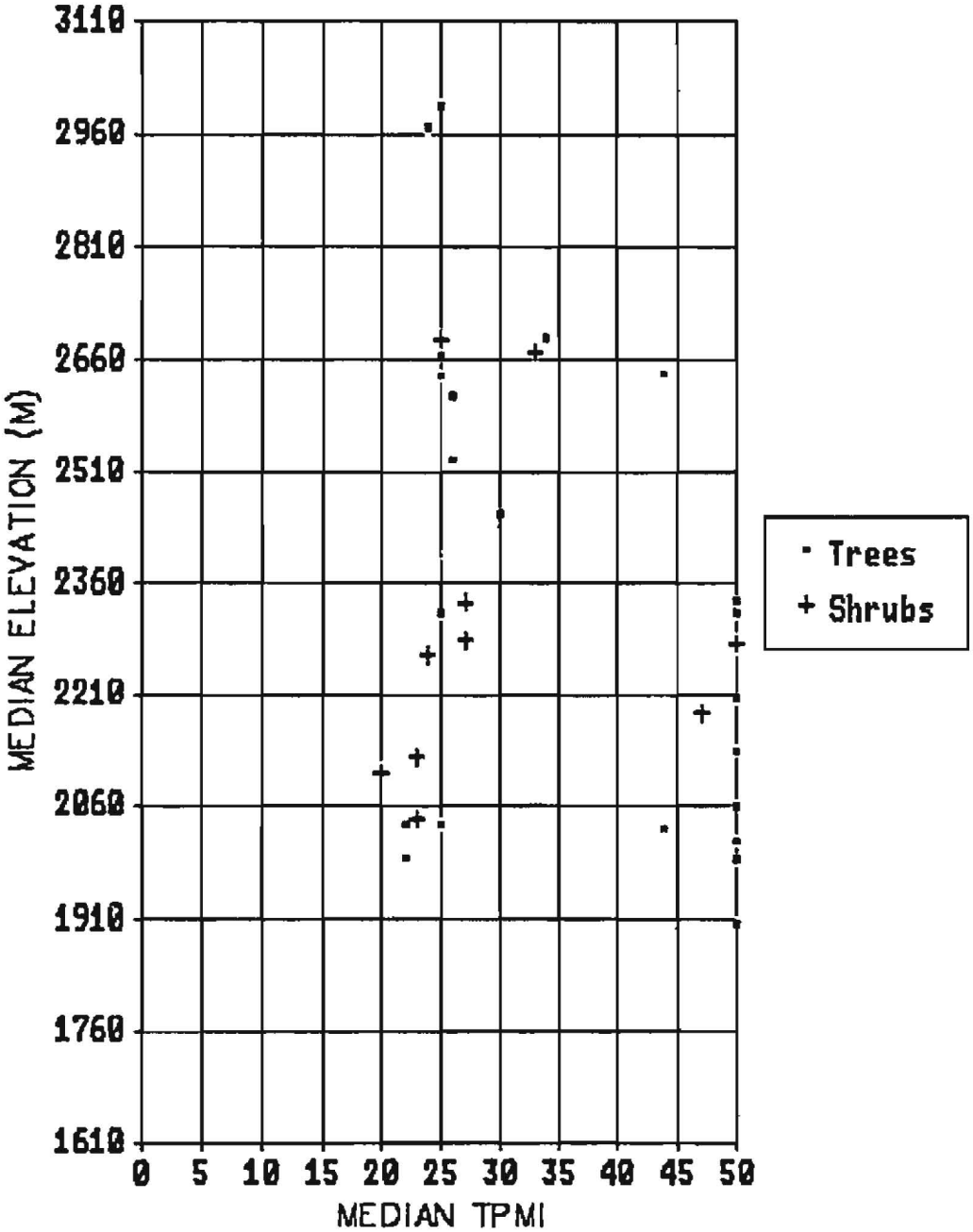
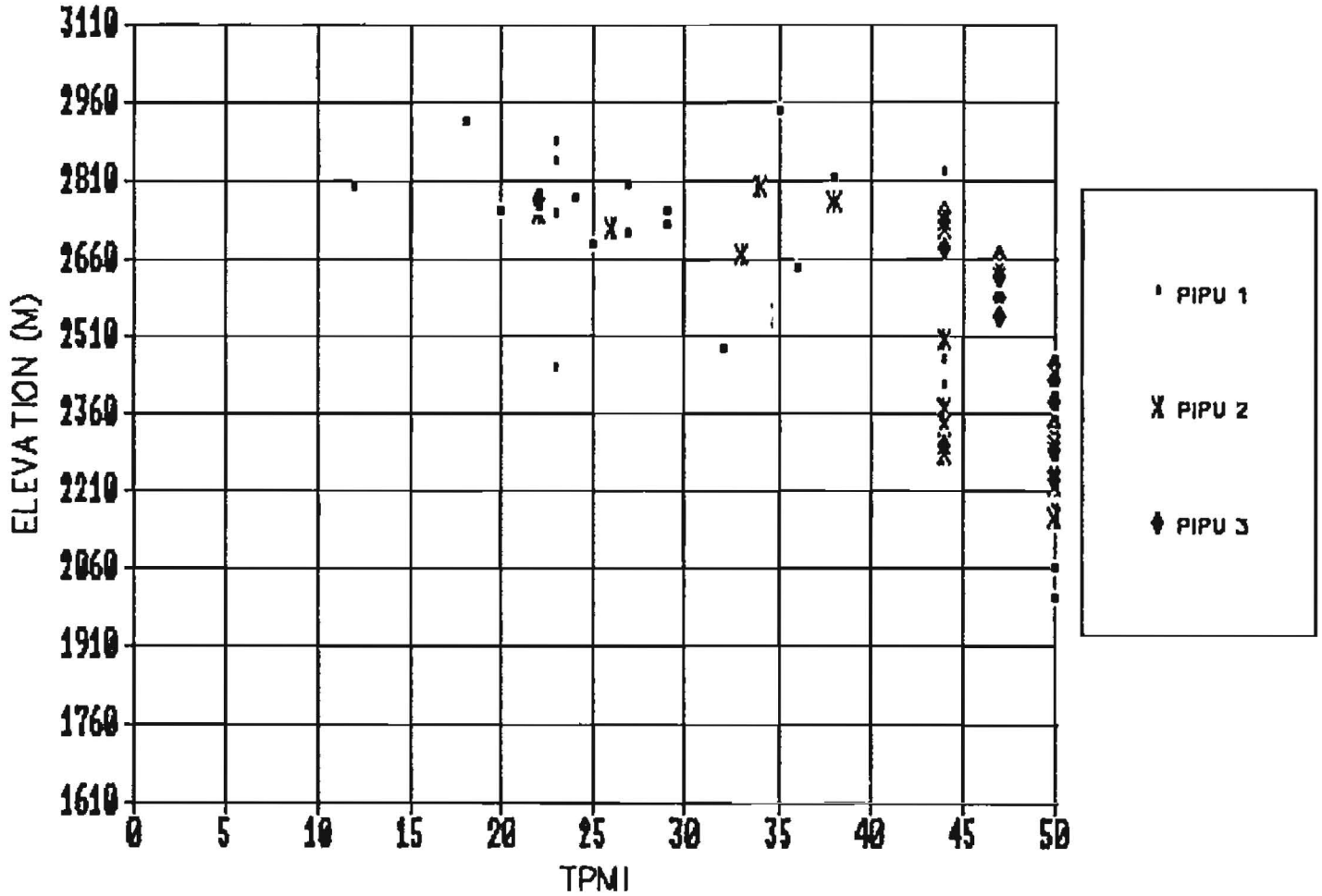


FIGURE 5-2. Blue spruce sample point distribution.



the GIS (e.g. Figures 5-3, 5-4) yields a high resolution picture of the 1987 distribution of most trees and shrubs in the Rito de los Frijoles Watershed. Analysis of these field data reveal changes in the distributions of certain woody species which are discussed individually below.

CORKBARK FIR (ABLA)

This species is typically found in low abundance on the north-facing slopes of the highest peaks in the Jemez Mountains, usually in association with Engelmann spruce. On Cerro Grande most mature fir trees died sometime in the last several decades, with beetle galleries evident under the bark of many dead trees. Perhaps the 1950's drought or a concurrent local spruce budworm outbreak (Swetnam 1989) killed these trees. Saplings and seedlings are common on most of these sites, and corkbark fir is apparently now increasing in abundance in most areas where it was formerly present and colonizing new sites.

BLUE SPRUCE (PIEN)

Blue spruce is a dominant in some mid to high elevation riparian areas, with individuals scattered in upper mixed conifer forests (Figures 5-2, 5-3). Blue spruce appears to be increasing in density and invading moist meadow sites in the headwater portions of the Frijoles drainage.

WHITE FIR (ABCO)

White fir is a common codominant of local mixed conifer forests. Fire suppression has allowed this species to increase in density on

Figure 5-3. Map of 1987 sample points in the Frijoles watershed where blue spruce was present. The gray patch is the entire sampled area, with the Frijoles drainage outlined with black dashes. Blue spruce was a dominant at the red points, common at the yellow points, and scarce at the blue points.

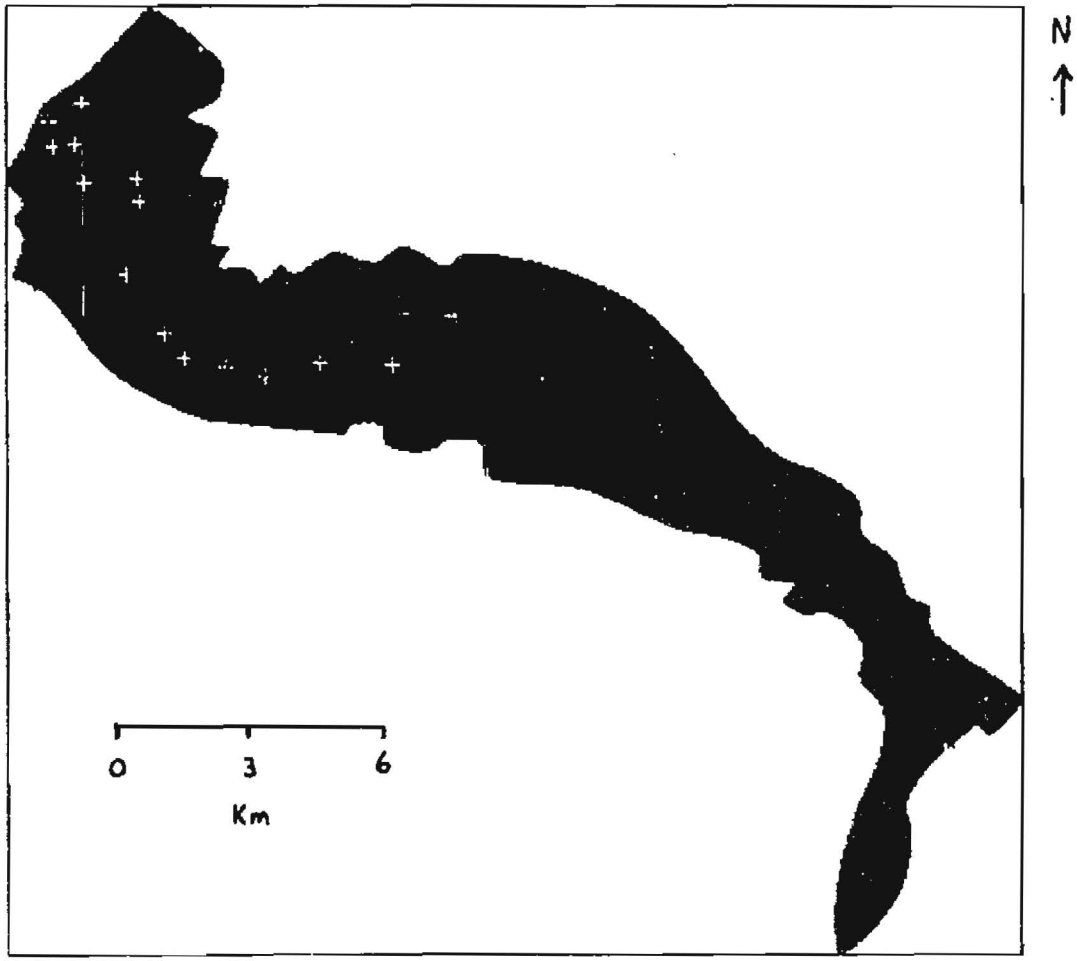


Figure 5-3.

Figure 5-4. Map of 1987 sample points in the Frijoles watershed where ponderosa pine was present. The gray patch is the entire sampled area, with the Frijoles drainage outlined in black. Ponderosa pine was a dominant at the red points, common at the yellow points, scarce at the blue points, and present only as dead specimens at black points.

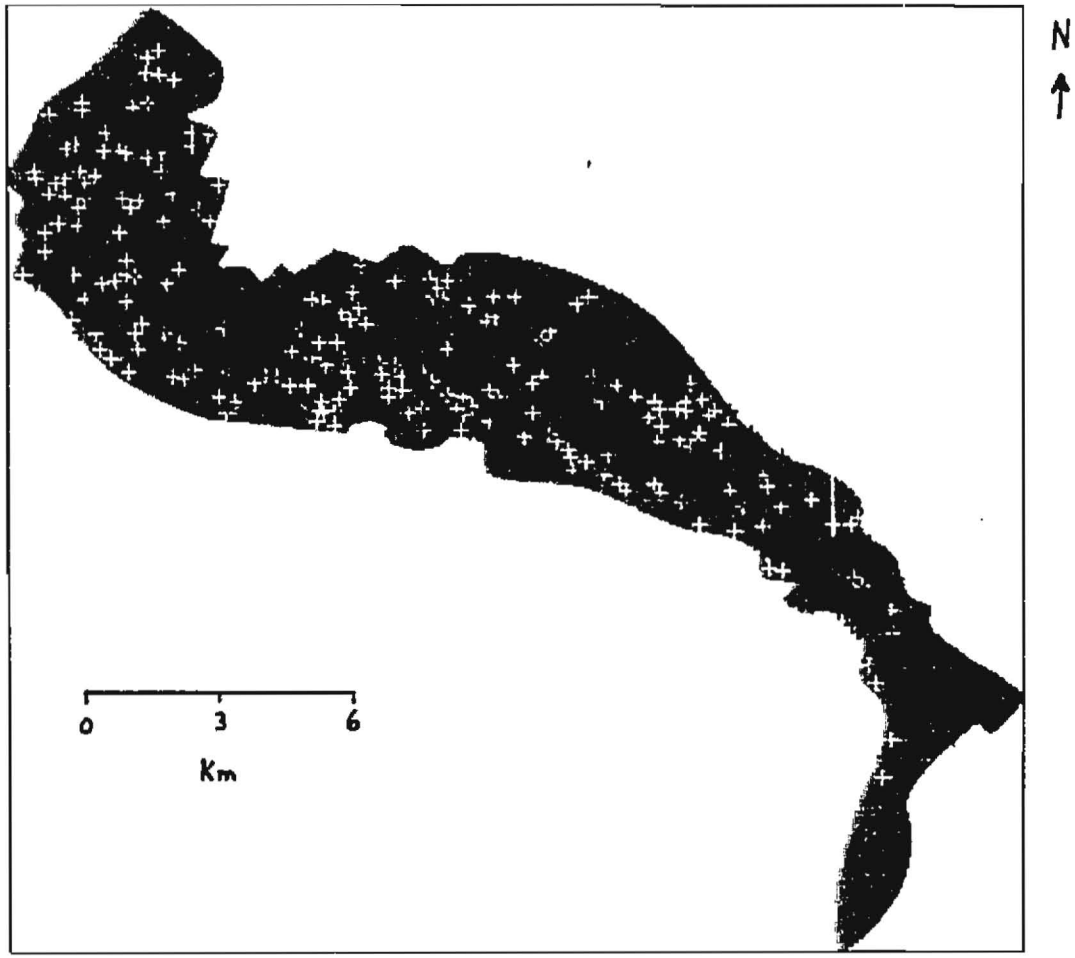


Figure 5-4.

many sites, although the spruce budworm outbreak of the 1980's caused obvious mortality and crown dieback in some thickets of white fir seedlings and saplings. The 1977 La Mesa Fire killed young white fir which was expanding downslope into the understories of dense ponderosa pine stands on Apache Mesa. This intense fire also eliminated old-growth white fir and other mixed conifer species from some of the side-valleys which drain Escobas and Burnt Mesas. By destroying all white fir from at least 15 sample points this fire caused the median elevation for this species to rise by 10 m (Table 5-1). The extent of low-elevation white fir prior to the La Mesa Fire may be underestimated by these data, as remnant woody material was classified as Douglas-fir in the field if its identity as white fir was uncertain, and remnants of many small trees were likely no longer visible 10 years after this fire.

DOUGLAS-FIR (PSME)

The distribution of Douglas-fir has changed in a similar but more obvious fashion in response to post-1900 fire suppression and the resultant La Mesa Fire. In 1977 Douglas-fir poles and saplings were common and even dominant in many ponderosa pine understories on Escobas Mesa and the similar mesa across Frijoles Canyon, despite the absence of Douglas-fir in the overstories of these stands. The crown fires that swept these mesas and dissecting valleys eliminated Douglas-fir from 74 sample points, reversing this species' downslope expansion and forcing the median Douglas-fir elevation upslope 47 m from its 1977 position (Table 5-1). The 1980's spruce budworm outbreak had fewer obvious impacts on young Douglas-fir than co-occurring white fir in the

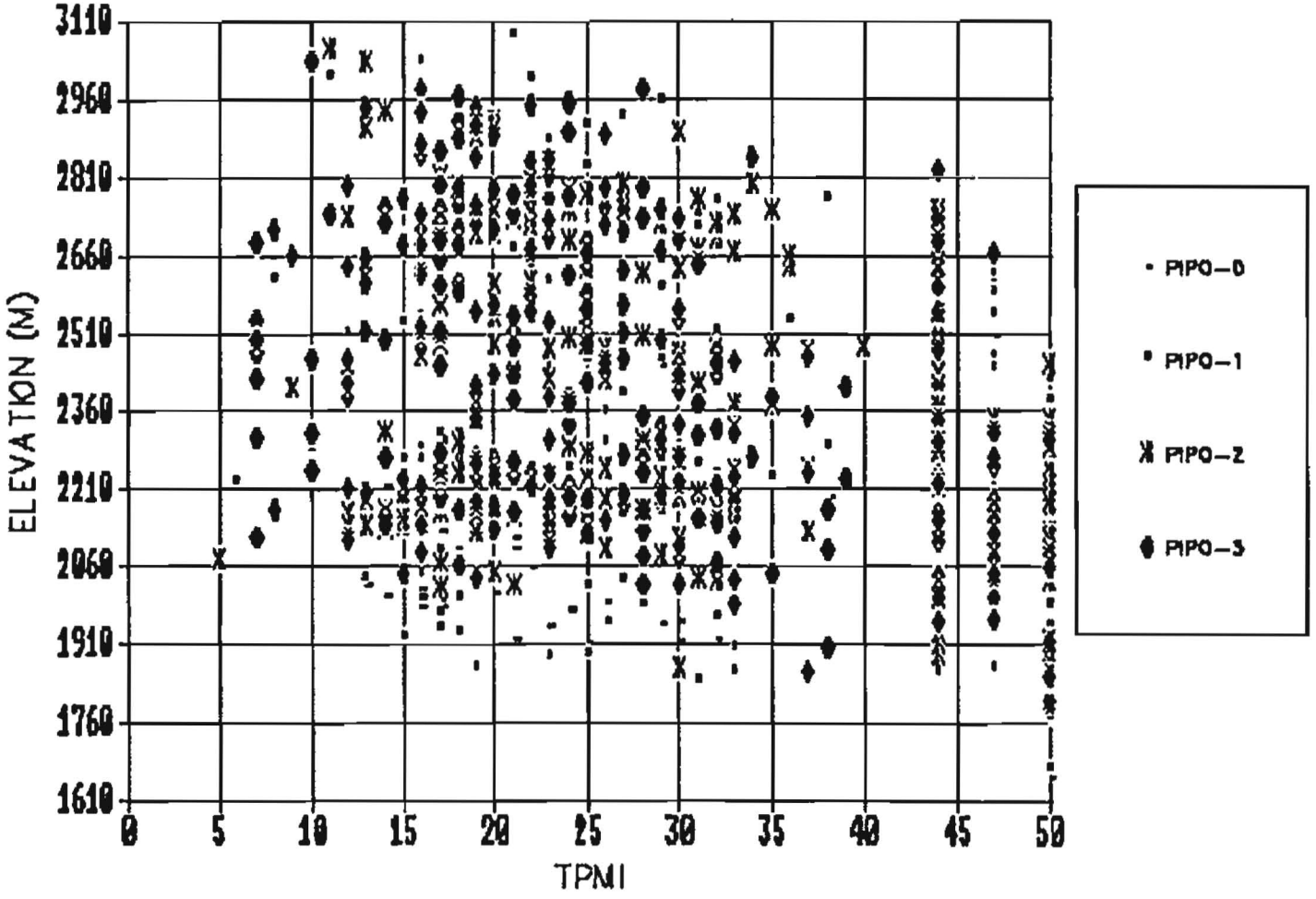
mixed conifer forests above the La Mesa Fire zone. Douglas-fir is a prominent species in the invasion of ancient montane grasslands, like those on Cerro Grande, which is described below under LANDSCAPE ELEMENTS: MONTANE GRASSLANDS.

PONDEROSA PINE (PIPO)

Ponderosa pine is the most characteristic tree of the Frijoles watershed, recently ranging from 1632 m at its mouth on the Rio Grande floodplain to 3100+ m near the summit of Cerro Grande (Figures 5-4, 5-5). Several changes have occurred in the local distribution of ponderosa pine during the 20th Century. Fire suppression and past grazing practices triggered massive encroachment of ponderosa pine into ancient montane grasslands, which is detailed below under LANDSCAPE ELEMENTS: MONTANE GRASSLANDS. The previously described 1950's drought and bark beetle outbreak killed mesatop ponderosa pines across a several-kilometer wide band along its lower range limits (Figures 4-22, 5-5). This altered lower ecotone is the primary reason for the recent 28 m increase in the median elevation for this species (Table 5-1). The 1977 La Mesa Fire eliminated or reduced the abundance of the formerly dominant ponderosa pine from large patches in mid-portions of the Frijoles Watershed (Figures 4-16, 5-4). Finally, the high water levels retained by Cochiti Reservoir from 1985-1987 inundated the mouth of Frijoles Canyon for extended periods, killing the lowest elevation ponderosa pines in the watershed.

Fire suppression, as well as past timber harvesting practices (Hessburg and Beatty 1986), may have contributed to increased abundance of dwarf mistletoe (Arceuthobium vaginatum ssp. cryptopodum) parasites

FIGURE 5-5. Ponderosa pine sample point distribution.



on ponderosa pine (cf. Wicker and Leaphart 1976, Zimmerman and Laven 1984). Parasitized ponderosa pine trees were observed at 124 of 727 field sample points, especially on portions of Apache and Escobas Mesas that had been high-graded early in this century.

ROCKY MOUNTAIN JUNIPER (JUSC)

This species has one of the broadest TPMI ranges of any tree (6-50). Its median TPMI value of 44 indicates that it is typically found today in swales, valleys, and canyon bottoms, although it achieves its greatest community dominance on rocky, xeric sites along the north rim of Frijoles Canyon. Many upland sites in the higher portions of this juniper's range were burned by the La Mesa Fire, eliminating this species from 26 sample points (27.7% of its pre-fire total). This range reduction caused the median elevation of Rocky Mountain juniper to drop by 49 m, and its TPMI to increase by 14 units, in the Frijoles Watershed (Table 5-1).

PIÑON (PIED)

Piñon is the familiar co-dominant of mid to lower portions of the Pajarito Plateau. Its seed is widely dispersed by piñon jays (Gymnorhinus cyanocephalus) and Clark's nutcrackers (Nucifraga columbiana), and I have seen piñon growing as a shrub as high as 3050 m on the slopes of Cerro Grande. Prior to this century frequent fires in the up-mesa ponderosa pine forests must have restricted the upper range limits of piñon. Fire suppression over the past century likely allowed piñon to expand its distribution somewhat into the ponderosa pine forests, as well as increase its density on former savanna and woodland

sites. The La Mesa Fire burned downslope to the approximate modern ecotone between ponderosa pine forests and piñon-juniper woodlands in this watershed, eliminating piñon from some of the higher elevation sites it had expanded into. Previously the 1950's drought and bark beetle outbreak described above had killed many piñon in the lower and drier portions of its local distribution (Figure 4-23), although subsequent recolonization has returned piñon to many of these sites. These changes eliminated piñon from 39 sample points, but the contrary impacts of upper and lower range restrictions had little net effect upon the median elevation for this species, despite a 35 m rise in the lower range limit (Table 5-1).

ONE-SEED JUNIPER (JUMO)

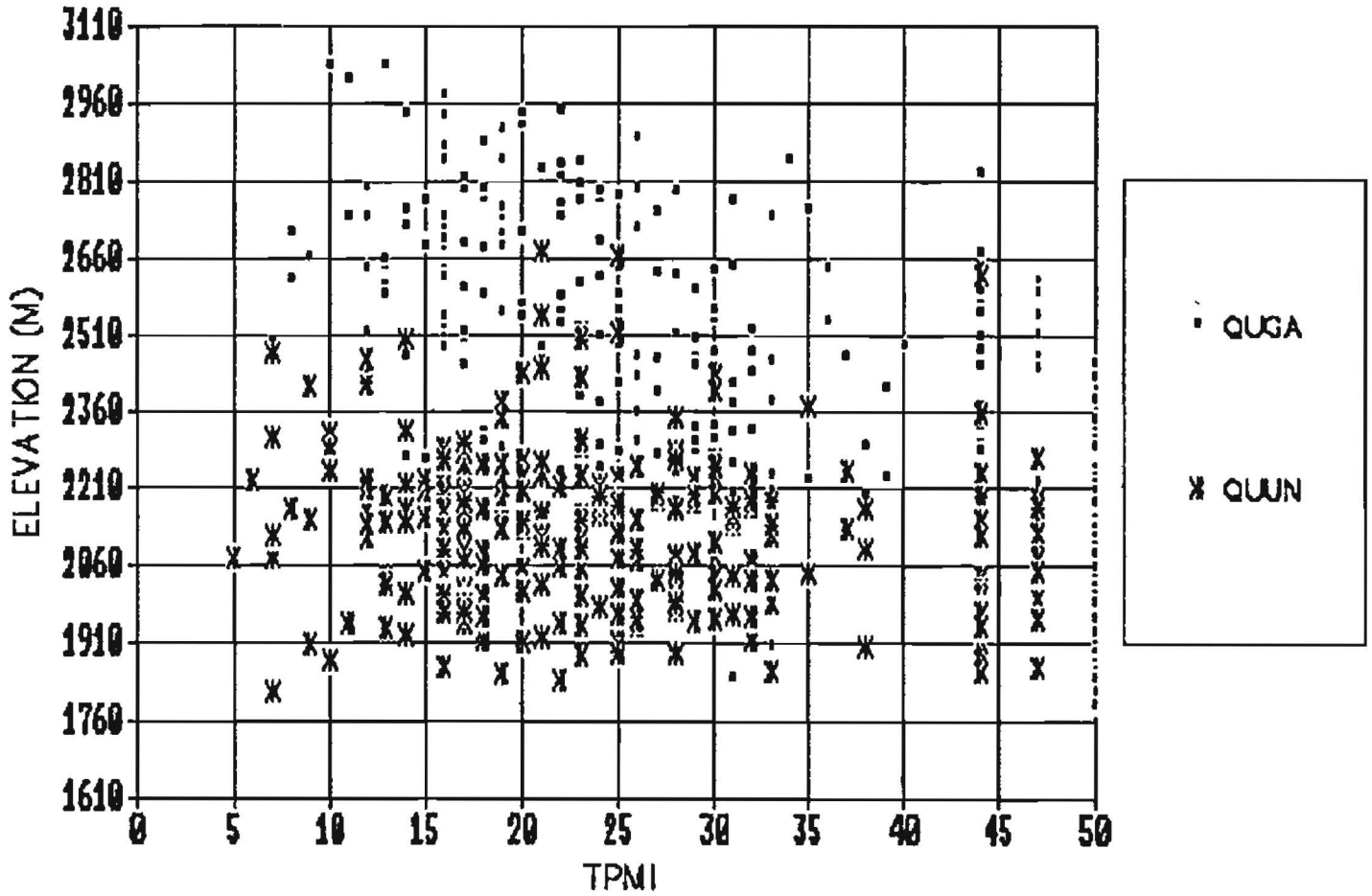
One-seed juniper is quite susceptible to fire damage when young (Johnsen 1962), which apparently was a major factor limiting the pre-1870 density and distribution of this species. Fire suppression and grazing-induced reductions in herbaceous ground cover over the past 120 years have apparently permitted one-seed juniper to expand its local range both upslope into ponderosa pine forests and downslope into former grasslands in the lowlands surrounding the Jemez Mountains, as only young junipers (<120 years old) are observed in these locations. In the Frijoles watershed low densities of old juniper are commonly found amidst relatively thick groves of young trees, indicating that increases in density have also occurred. One-seed juniper continues to increase across the ponderosa pine forest/piñon-juniper woodland ecotone where dense clumps of young juniper are commonly found surrounding individual pine trees, apparently due to bird dispersal of

juniper seeds. The La Mesa Fire truncated part of the upslope range expansion of one-seed juniper, causing the maximum sample elevation to drop by 92 m and the median elevation to decrease by 24 m (Table 5-1). The high incidence of true mistletoe (Phoradendron juniperinum ssp. juniperinum) infections on old junipers in the Frijoles watershed (128 of 307 sample points) may be related to the long absence of a potential regulator, fire, from these sites (cf. Wicker and Leaphart 1976, Zimmerman and Laven 1984). Most of these infections were observed in the upper portions of the sampled range of this species, where pre-1900 fire frequencies would have been highest.

SHRUBS

Most shrubs in the Frijoles Watershed will resprout after damage to surface stems, an adaptation that served these understory species well prior to 1900 when surface fires burned across this watershed at frequent intervals. Even shrubs from relatively mesic environments, such as Prunus sp. and Rocky Mountain Maple (Table 5-2), are vigorous sprouters. Roughly a century of fire suppression has probably caused shrub species to decline in many areas where dense forest canopies and understory tree thickets have developed. For example, in the dense mixed conifer forests of the Upper Frijoles Watershed Gambel oak is often found as small suppressed sprouts amidst the remants of large, dead oak stems which obviously once thrived in a more open environment. The La Mesa Fire benefitted many such shrub species by stimulating them to sprout in an environment in which competing trees had been eliminated or reduced. This fire burned 63.7% of 1987 sample points recording Gambel oak shrubs (256 points), 56.3% of wavyleaf shrub oak

FIGURE 5-6. Gambel oak and wavyleaf oak sample point distribution.



sites (202 points), 48.1% of mountain mahogany sites (81 points), and 74.4% of New Mexico locust shrub sites (262 points). This burn area corresponds to the transition zone between Gambel and wavyleaf oaks (Figure 5-6) and apparent hybrids are commonly observed here. Shrub thickets have developed in some portions of the La Mesa Fire burn area which will likely retard the return of trees to these sites.

Fendler ceanothus (CEFE) is a non-sprouting shrub; persistent seeds germinate readily following fire (Conard et al 1985). Fire suppression in the Frijoles watershed must have markedly reduced the abundance of this favored deer browse species. All 97 points with this shrub present were burned by the La Mesa Fire, with crown fire occurring on 67% of these sites (65 points). The nitrogen-fixing capacity of this ceanothus and New Mexico locust may be important to the long-term nutrient cycles of ponderosa pine forests.

In contrast to the fire-dependent shrubs just discussed, common juniper (JUCO) is a fire-sensitive non-sprouter. Frequent fires prior to 1900 may have largely restricted this species to rocky sites where it was protected from fire. No doubt fire suppression has allowed this juniper to expand its distribution within the mixed conifer forests of the Frijoles Watershed.

COVER-TYPES IN THE RITO DE LOS FRIJOLES WATERSHED

The elevational distributions of 27 cover-types in the Frijoles watershed, based upon field sampling at 969 points, are listed in Table 5-3. Associated TPMI values for these same cover-types are given in Table 5-4.

The distributions of forest cover-types reflects the underlying

TABLE 5-3. 1987 elevational distribution of cover-types in the Frijoles Watershed, arrayed from highest to lowest median elevation. All elevations in meters. Cover-type abbreviations from Tables 3-2 and 3-3. Cover-types labelled *.PAST reflect the dominance of the cover-type in the recent past, based on standing and downed woody material as well as live individuals. N = 969 sample points.

Cover-Type	n	Range	Median	Mean	Standard Deviation
ES	5	2989 - 3076	3044	3040.5	31.9
MG	4	2794 - 3052	3029	2954.7	116.7
QG	1	2913	2913	2913.0	-
TA	24	2409 - 3027	2840	2806.2	157.0
FELS	2	2603 - 2995	-	2799.1	277.1
MEAD	5	2455 - 2795	2742	2703.2	141.2
MC	211	2085 - 2971	2678	2610.7	197.3
PPMC	58	2159 - 2931	2547	2548.4	196.7
PPMC.PAST	82	2159 - 2931	2580	2553.6	187.9
DF	4	2037 - 3056	2303	2371.0	471.0
RI-E	21	1950 - 2634	2300	2298.9	215.7
GRSH	101	1856 - 2669	2251	2278.1	155.1
GRSH.PAST	4	1856 - 1918	1860	1873.9	29.8
PP	154	1853 - 3026	2230	2318.0	245.6
PP.PAST	233	1853 - 3026	2234	2298.4	219.3
SH-O	7	2143 - 2611	2219	2310.4	189.9
GRAS	42	1943 - 2611	2211	2239.2	147.4
GRAS.PAST	2	1943 - 2111	-	2026.7	118.6
SHRU	20	1903 - 2580	2182	2214.4	179.7
PPDF	28	1982 - 2940	2162	2234.9	230.0
PPDF.PAST	60	1982 - 2940	2210	2251.3	188.9
PPPJ	36	1908 - 2230	2047	2071.0	65.2
PPPJ.PAST	60	1850 - 2230	2080	2065.8	99.9
PJ	124	1778 - 2220	2002	2006.3	80.8
PJ.PAST	146	1778 - 2220	1975	1996.2	87.8
RI-M	24	1777 - 2420	1982	2006.6	176.5
J	50	1665 - 1999	1912	1899.2	77.0
RI-D	9	1668 - 1973	1853	1817.5	96.9
CWCX	2	1830 - 1832	-	1830.9	1.1
JSAV	14	1650 - 1940	1760	1781.6	78.2
ROCK	11	1662 - 1851	1726	1735.9	59.0
TALU	1	1694	1694	1694	-
WRSW	1	1653	1653	1653.0	-
SAND	1	1650	1650	1650.0	-

TABLE 5-4. 1987 TPMI distribution of cover-types in the Frijoles Watershed, arrayed from highest to lowest median elevation to match Table 5-3. Cover-type abbreviations from Tables 3-2 and 3-3. Cover-types labelled *.PAST reflect the presence of the species in the recent past, based on standing and downed woody material as well as live individuals. N = 969 sample points.

Cover-Type	n	Range	Median	Mean	Standard Deviation
ES	5	21 - 26	24	23.4	2.3
MG	4	11 - 34	16	18.5	10.5
QG	1	20	20	20	0
TA	24	11 - 44	23	23.1	8.9
FELS	2	20 - 22	-	21	1.4
MEAD	5	18 - 44	27	31.4	11.9
MC	211	11 - 50	26	28.6	10.1
PPMC	58	7 - 44	19	21.6	10.7
PPMC.PAST	82	7 - 47	20	23.0	11.2
DF	4	7 - 35	26	21.3	12.0
RI-E	21	47 - 50	50	49.3	1.3
GRSH	101	12 - 47	24	26.0	9.3
GRSH.PAST	4	15 - 47	47	31.5	17.9
PP	154	7 - 47	24	25.9	10.6
PP.PAST	233	7 - 47	23	24.9	9.6
SH-O	7	8 - 47	21	27.6	17.7
GRAS	42	5 - 44	24	23.5	7.9
GRAS.PAST	2	20 - 30	30	25.0	7.1
SHRU	20	9 - 47	28	26.1	11.3
PPDF	28	10 - 47	28	29.2	11.5
PPDF.PAST	60	10 - 47	30	31.4	11.1
PPPJ	36	5 - 47	32	31.6	12.3
PPPJ.PAST	60	5 - 44	25	26.0	9.8
PJ	124	6 - 44	22	23.3	8.6
PJ.PAST	146	6 - 44	19	22.1	8.2
RI-M	24	50 - 50	50	50.0	0
J	50	10 - 47	23	24.1	9.4
RI-D	9	50 - 50	50	50.0	0
CWCX	2	16 - 22	22	19.0	4.2
JSAV	14	7 - 25	20	18.5	5.9
ROCK	11	7 - 27	16	16.8	6.2
TALU	1	20	20	20	0
WRSW	1	27	27	27	0
SAND	1	50	50	50	0

distributions of the individual species which were used to classify the overstory association dominant at each sample point. For example, juniper woodlands are obviously found at lower elevations than Engelmann spruce forests, and evergreen conifers increase in dominance with increasing elevation in the riparian zone of Frijoles Canyon (Table 5-3). Non-forest cover-type distributions typically result from the locations of past forest-removing disturbances (e.g. the La Mesa Fire's creation of grasslands and shrub-fields) or specific geomorphic situations (e.g. steep, unstable slopes for the talus cover-type).

The La Mesa Fire caused many changes in cover-type distributions in this watershed. This crown fire converted 153 forested sample points to non-forested points (15.8% of all sample points). The number of sample points with grass-shrub, grass, and shrub cover-types increased dramatically, while the number of ponderosa pine, ponderosa pine/douglas-fir, and ponderosa pine/mixed conifer points declined (Table 5-3). Ponderosa pine/Douglas-fir cover-types declined by 53.3% overall, with the 52 m rise in median elevation indicating that most of the loss occurred in the lower part of this cover-type's former range, where it had been expanding down into ponderosa pine cover-types. Ponderosa pine/mixed conifer cover-types were similarly selectively eliminated at lower elevations by this fire. Undisplayed GIS maps confirm these cover-type range changes.

The bark beetle outbreaks of the 1950's also changed the distribution of several cover-types. Beetle-caused mortality eliminated ponderosa pine as a co-dominant at 40% of past ponderosa pine/piñon-juniper points (24 of 60 points), but the mean elevation of this type was little changed due to the confounding affects of the

conversion of 19 other PPPJ points to other types by the La Mesa Fire and the conversion of other types to PPPJ. The impact of the beetle outbreak on piñon-juniper cover-types is more obvious, as 35 of the 42 converted sample points lost piñon as a co-dominant due to drought/beetle-induced mortality (Table 5-3). The selective loss of piñon along the lower portions of its former cover-type range caused the median elevation of this type to rise by 27 m.

MAPPED LANDSCAPE CHANGES: 1935-1981

LANDSCAPE COVER-TYPES

Figures 5-7 and 5-8 display the outlines of the 1935 and 1981 landscape cover-type maps, while Figures 5-9 and 5-10 are the maps in color. Each patch in Figures 5-7 and 5-8 is known as a polygon in GIS terminology. Tables 5-5 and 5-6 summarize the cover-type information contained in these two polygon maps.

These msps cover portions of 11 USGS 7.5 minute topographic quadrangles; the quadrangle boundaries are visible in Figures 5-7 and 5-8. These maps were digitized one quadrangle at a time, with each quad defined as a separate "grounit" - this procedure unknowingly imbedded the artificial quadrangle boundaries into the GIS maps in a tenacious fashion that has resisted all efforts to remove them to date. In the GIS this results in overestimates of total polygon number and thus underestimates mean patch areas, overestimates perimeters for hundreds of individual polygons and the entire map, overestimates perimeter:area ratios, and overestimates patch dissection indices. The artificial southern map boundary introduces similar but less significant errors. Total map area of each cover-type is unaffected.

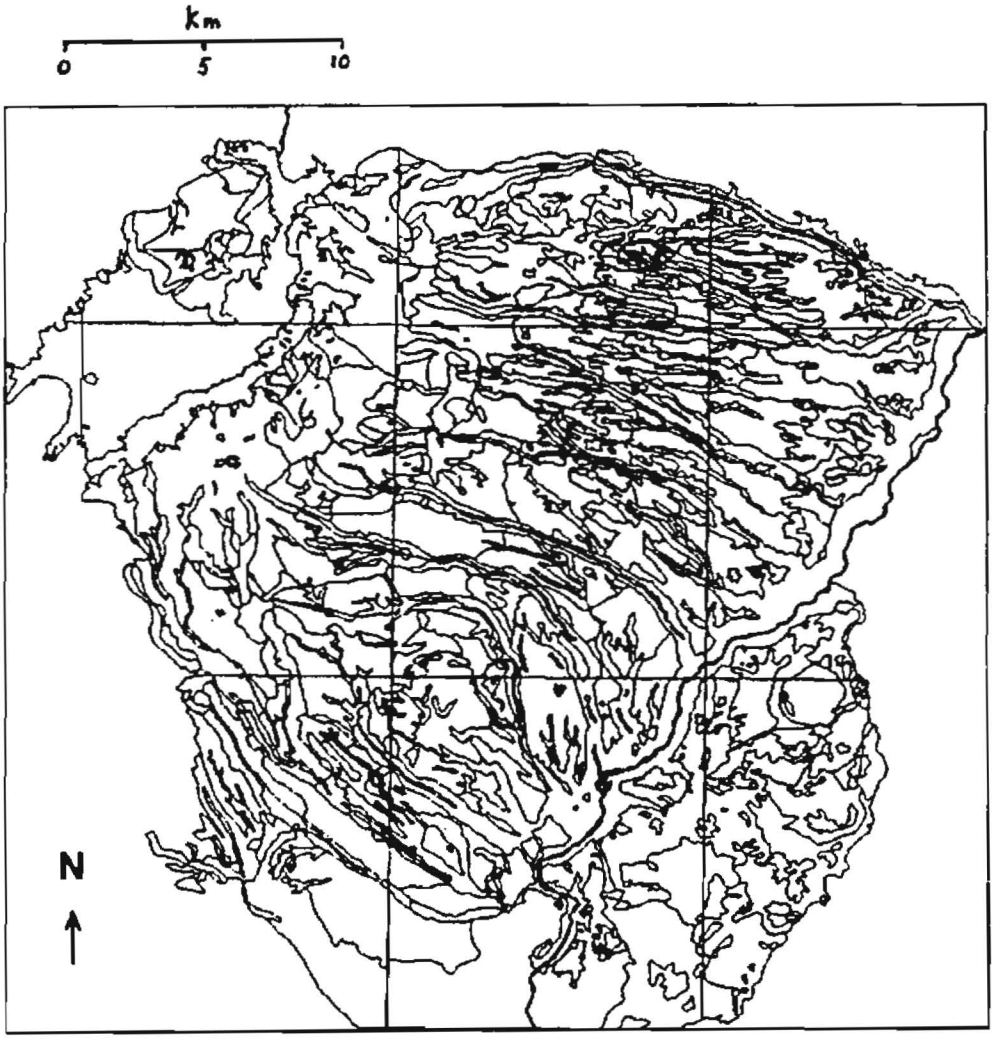


Figure 5-7. Outline map of 1935 landscape cover-types.

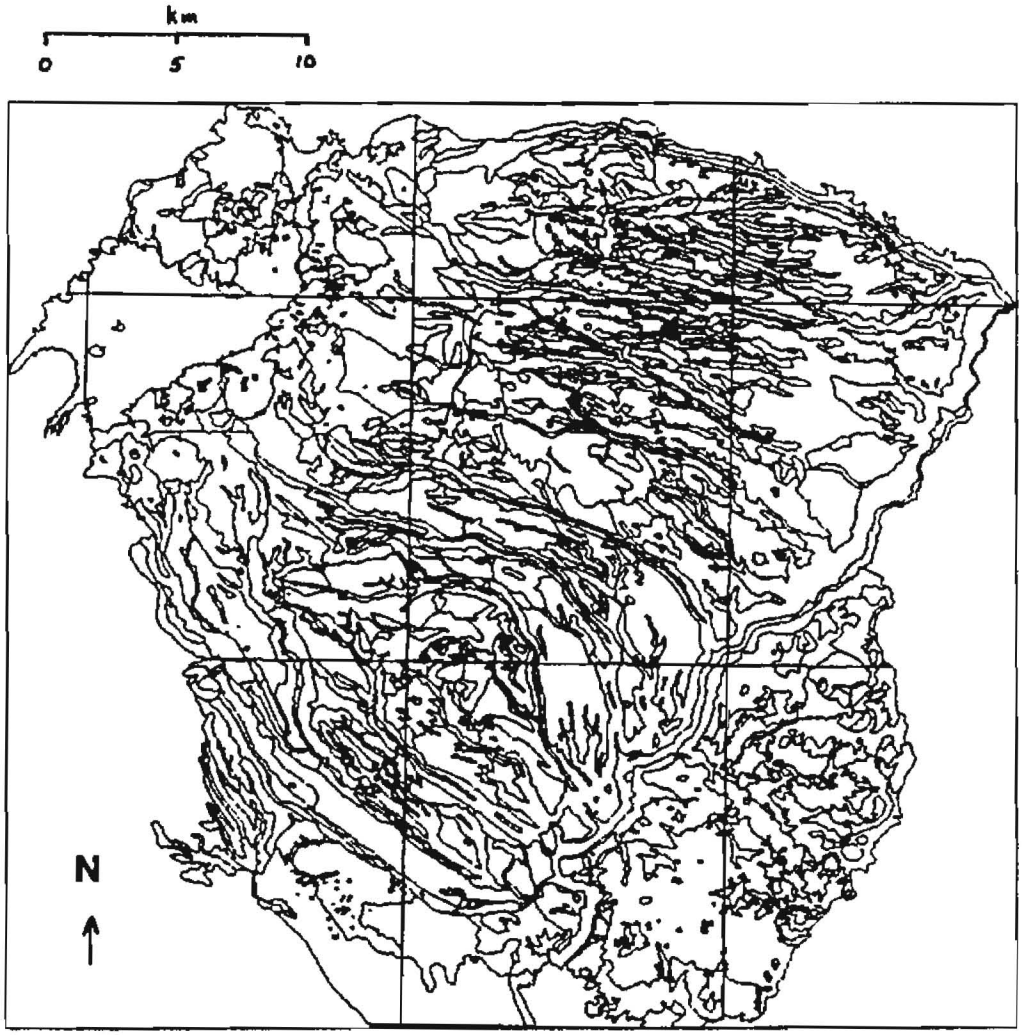


Figure 5-8. Outline map of 1981 landscape cover-types.

Figure 5-9. Color map of 1935 landscape cover-types. Forest = green, woodland = yellow, meadow = light blue, montane grassland = magenta, grassland and shrubland = gray patterns, cultural features (primarily agricultural fields) = red, canyon walls = black, and water = dark blue.

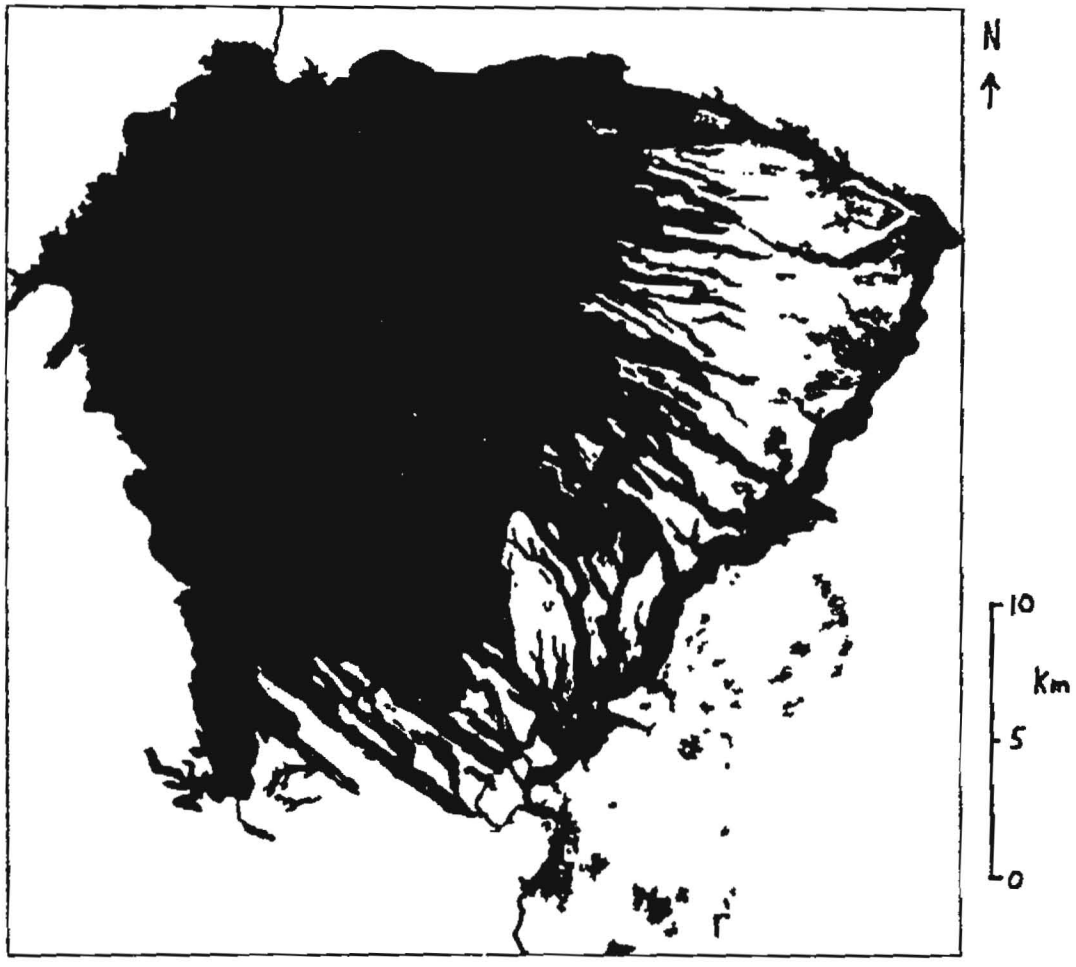


Figure 5-9.

Figure 5-10. Color map of 1981 landscape cover-types. Forest = green, woodland = yellow, meadow = light blue, montane grassland = magenta, grassland and shrubland = gray patterns, most cultural features (primarily industrial, residential, commercial, and mine areas) = red, ski area and golf courses = white, canyon walls = black, and water = dark blue.

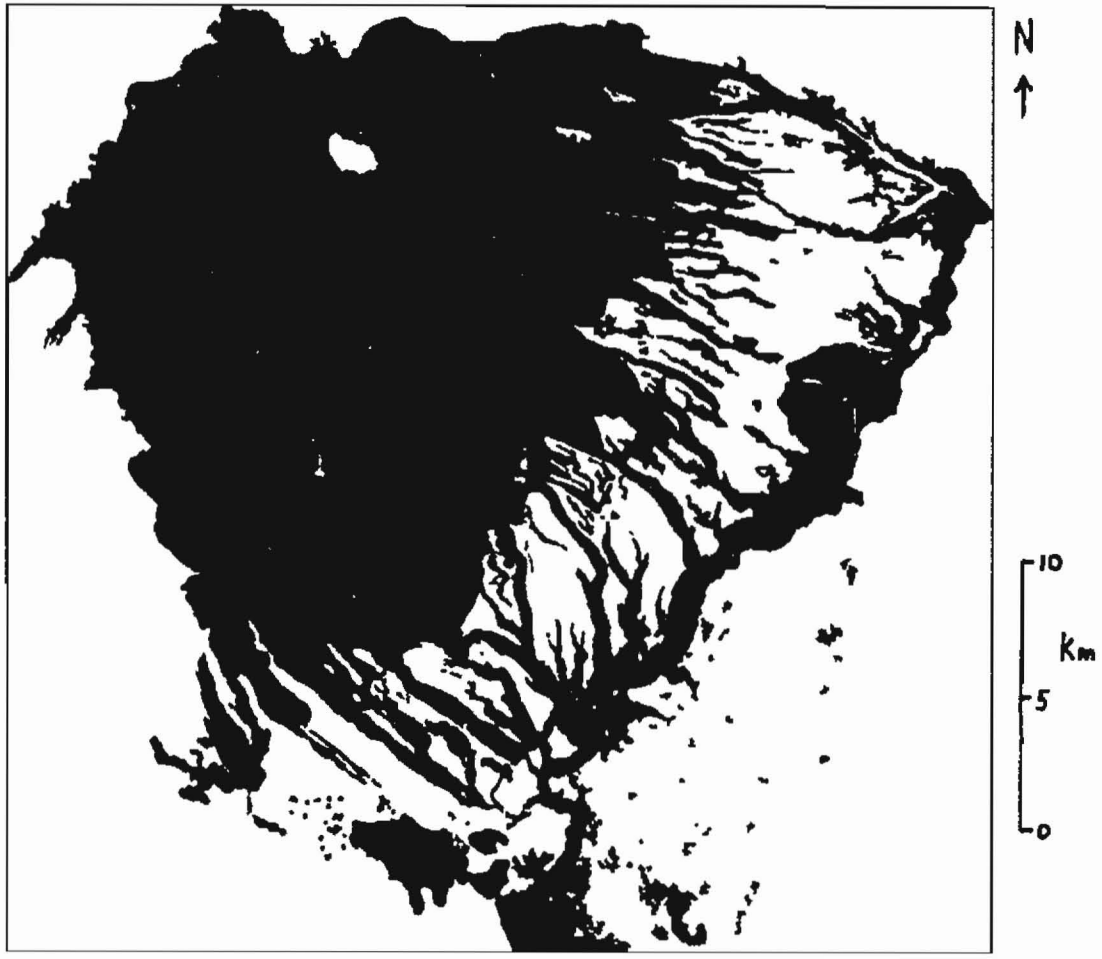


Figure 5-10.

TABLE 5-5. Summary cover-type information for the 1935 landscape cover-type map. Cover-type abbreviations are described in Tables 3-2 and 3-3, N is the corrected number of polygons, % area is the percentage of the total map area, mean area is the corrected mean polygon area (ha), % P is the percentage of the uncorrected total perimeter, mean P is the uncorrected mean polygon perimeter (km), mean P:A is the mean perimeter:area ratio (km/km^2), and mean PDI is the mean patch dissection index (unitless). Total map area is 85,726.0 ha, uncorrected total map perimeter is 5390.1 km, and corrected total map perimeter is 5131.4 km. Total (U) uses corrected total area but uncorrected perimeter data, while Total (C) uses corrected area and perimeter data.

Cover Type	N	% Area	Mean Area	% P	Mean P	Mean P:A	Mean PDI
COMM	1	~0	0.9	~0	0.4	44.4	1.18
CWCX	24	12.85	459.2	14.53	32.6	7.1	4.29
FARM	37	0.88	20.5	1.60	2.4	11.7	1.49
FELS	27	0.04	1.2	0.26	0.5	41.7	1.28
FORE	373	43.60	100.2	46.25	6.8	6.8	1.91
GRAS	128	1.84	12.3	3.91	1.7	13.8	1.36
MEAD	30	5.60	160.1	3.30	6.0	3.7	1.33
MG	21	0.65	26.4	1.09	2.8	10.6	1.53
POND	1	~0	0.7	~0	0.3	42.9	1.01
ROCK	3	0.01	3.1	0.06	1.1	35.5	1.76
SAND	11	0.03	2.0	0.14	0.7	35.0	1.39
SHRU	10	0.13	11.2	0.30	1.6	14.3	1.34
STRE	1	0.25	211.5	1.43	78.2	37.0	15.16
WOOD	157	34.09	186.3	25.80	9.0	4.8	1.86
Total (U)	823	100.00	104.16	100.00	6.55	6.28	1.81
Total (C)	823	100.00	104.16	100.00	6.23	5.97	1.72

Insertion of the correct number of polygons (obtained from a separate dBase III Plus database) into the tables (5-5, 5-6) results in accurate mean patch area for each cover-type and correct summary values for the whole map, leaving just the inaccurate perimeter-associated values for each cover-type. Accurate fractal dimensions (O'Neill et al 1988) cannot be calculated until this problem is resolved, which should occur

TABLE 5-6. Summary cover-type information for the 1981 landscape cover-type map. Cover-type abbreviations are described in Tables 3-2 and 3-3. N is the corrected number of polygons, % area is the percentage of the total map area, mean area is the corrected mean polygon area (ha), % P is the percentage of the uncorrected total map perimeter, mean P is the uncorrected mean polygon perimeter (km), mean P:A is the mean perimeter:area ratio (km/km^2), and mean PDI is the mean patch dissection index (unitless). Total map area is 85,631.9 ha, uncorrected total map perimeter is 6037.0 km, and corrected total map perimeter is 5776.9 km. Total (U) uses corrected total area but uncorrected perimeter data, while Total (C) uses corrected area and perimeter data.

Cover Type	N	% Area	Mean Area	% P	Mean P	Mean P:A	Mean PDI
COMM	6	0.26	37.0	0.29	2.9	8.0	1.36
CWCX	34	1.80	297.1	13.57	24.1	8.1	3.94
FARM	7	0.05	5.5	0.15	1.3	23.9	1.53
FELS	29	0.03	1.0	0.24	0.5	52.1	1.44
FORE	416	37.44	77.1	42.68	6.2	8.0	1.99
GOLF	3	0.10	28.9	0.19	3.8	13.2	1.99
GRAS	116	3.92	29.0	4.37	2.3	7.8	1.19
IND	51	0.99	16.7	1.58	1.9	11.2	1.29
LAKE	1	1.89	1622.4	1.81	109.4	6.7	7.66
MEAD	32	5.22	139.6	3.02	5.7	4.1	1.36
MG	24	0.29	10.4	0.65	1.6	15.7	1.43
MINE	53	0.10	1.7	0.50	0.6	33.5	1.23
POND	13	0.02	1.3	0.10	0.5	36.9	1.19
RESI	15	2.64	150.7	1.56	6.3	4.2	1.44
ROCK	3	0.01	3.3	0.05	1.0	31.3	1.60
SHRU	10	0.14	11.8	0.36	2.2	18.3	1.77
SKI	1	0.23	194.7	0.11	6.6	3.4	1.33
STRE	1	0.02	19.4	0.13	8.0	41.1	5.10
WOOD	228	34.84	130.9	28.63	7.6	5.8	1.87
Total (U)	1043	100.0	82.10	100.0	5.79	7.05	1.80
Total (C)	1043	100.0	82.10	100.0	5.54	6.75	1.72

by the end of calendar year 1989 when these maps are running on a different GIS system. Rasterizing these polygon maps in MOSS to create a cell map could merge the divided map units but would lead to a loss of multiple attribute data and the precision obtained by developing the polygon maps in the first place.

Still, this added source of error probably has little overall effect on the landscape change results presented here. Both 1935 and 1981 maps have the same problem, and it seems reasonable to assume that the imbedded quadrangle boundaries are approximately randomly distributed with respect to cover-type location. Therefore correction of these data will alter the absolute values of calculated results for individual cover-types, but not their relative values within and between each map. Even the absolute value changes will not be large. Measurement of the imbedded quadrangle boundary lengths shows that total perimeter is only overestimated by 5.0% and 4.5%, mean perimeter by 5.1% and 4.3%, mean perimeter:area by 5.2% and 4.3%, and mean patch dissection indox by 4.1% and 3.5% for the 1935 and 1981 maps, respectively (Tables 5-5 and 5-6). Thus these landscape cover-type data must be considered preliminary until corrected values can be obtained, but the overall patterns of landscape change discussed here should remain valid after this high-tech headache is resolved. This difficulty must also be present although undiscussed in published data which use individual USGS land use digital maps as a source of information (e.g. O'Neill et al 1988, Iverson 1988) because those data are also truncated at quadrangle boundaries, although the degree of the problem is reduced by using smaller scale (1:250,000) maps where the quadrangle boundary is a lesser proportion of the total map perimeter.

Number of Patches

Total polygon number increased 26.1% from 823 in 1935 to 1043 in 1981 (Tables 5-5, 5-6) for several reasons. First of all, numerous cultural patches were added to this landscape (Figure 5-11), especially pumice mines and industrial, commercial, and residential areas associated with the development of LANL and the townsites of Los Alamos, White Rock, and Cochiti Lake. The introduction of these cultural patches also fragmented pre-existing patches, creating more total polygons. Similarly, the La Mesa Fire created more polygons by fragmenting forest and woodland patches with grasslands, and logging in some forest areas has created new patches. In addition, the greater resolution of the recent air photos may have permitted more detailed interpretation of cover-types, resulting in more 1981 polygons, despite efforts by the interpreters to be consistent between dates.

Surface Area

Changes in the total area of cover-types (Figure 5-12) are dominated by decreases in forest, agricultural lands, meadows, and montane grasslands, and increases in woodland, grass, lake, mine, recreational, industrial, commercial, and residential lands. Total map area decreased by 0.1% (94 ha), as tree invasion shrunk the Valle Grande meadow at the northwestern map boundary. Forest area dropped, while total and mean grassland patch area more than doubled, because of the La Mesa Fire. Mean patch size decreased by 21.2% from 104.16 to 82.10 ha, led by large decreases in woodland, montane grassland, and agricultural polygon size (Tables 5-5, 5-6). Because forest and woodland polygons are numerous and large they cover the most area at

FIGURE 5-11. Changes in number of landscape cover-type polygons, 1935 - 1981.

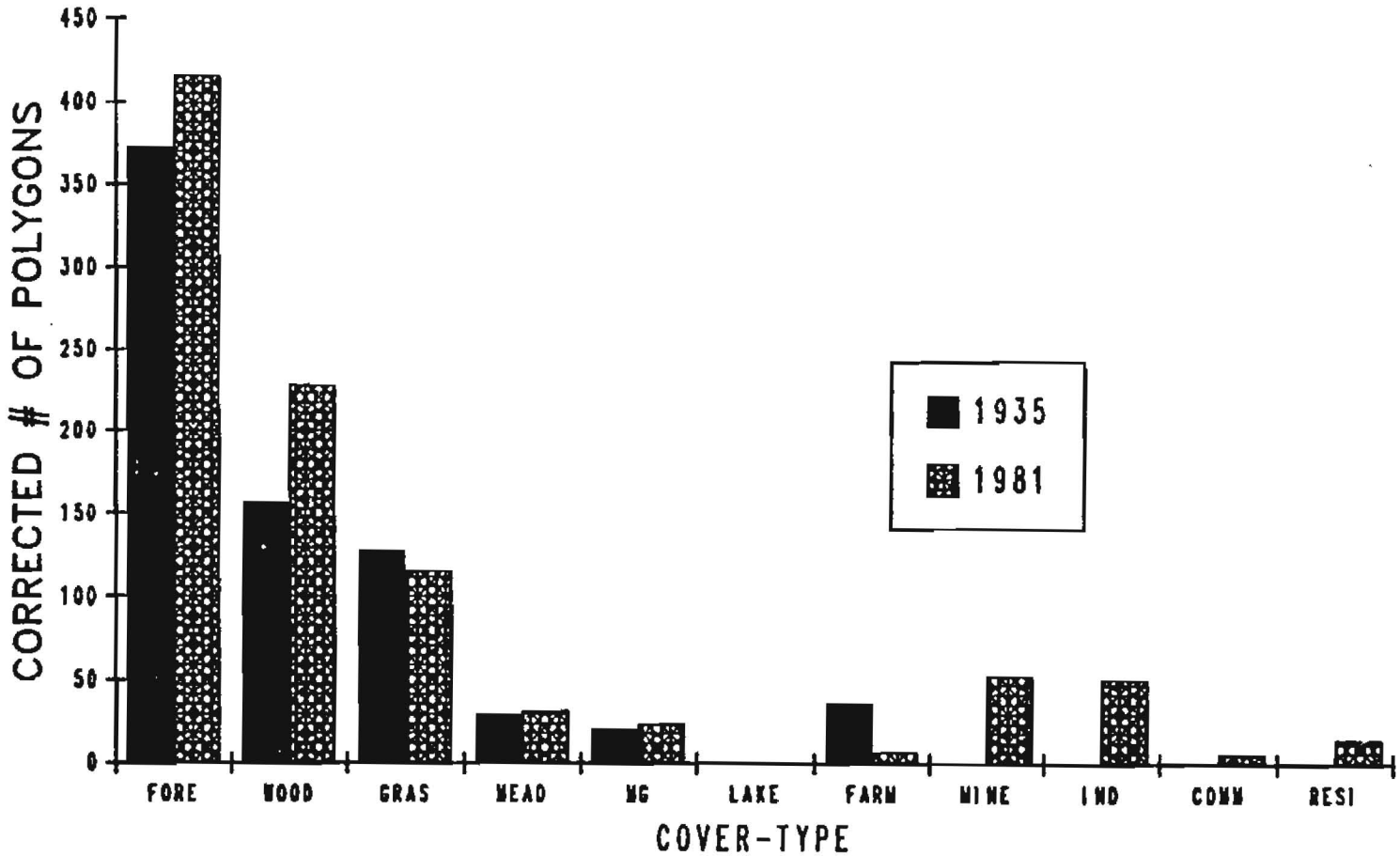
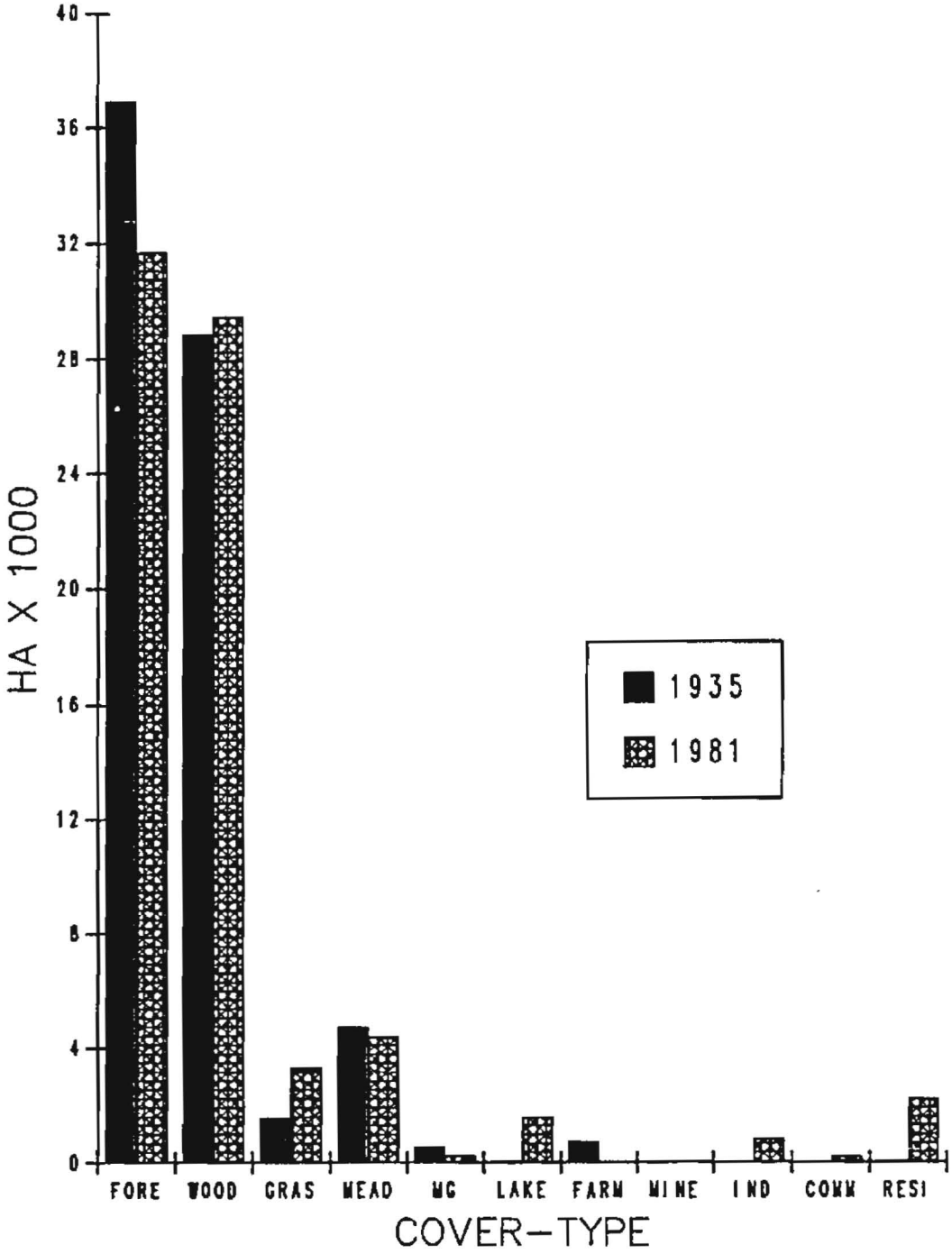


FIGURE 5-12. Changes in area of landscape cover-types, 1935-1981.



both dates, although their joint coverage declines from 77.7% to 72.3% of this landscape by 1981. Mean meadow areas are large because of the huge caldera patches.

In 1935 cultural features accounted for only 0.9% of this mapped landscape, primarily as dry-farmed bean fields on the mesas north of BNM (Figure 5-13). By 1981 cultural patches occupied 6.3% of this landscape (Figure 5-14). The bean fields were replaced by the technical (industrial) areas of LANL, and the townsites of Los Alamos, White Rock, and Cochiti Lake developed. Other new cultural features include pumice mines, stock ponds, Cochiti Reservoir, golf courses, and a ski area. Cochiti Reservoir alone floods 1.9% of the map area when about full, as in the summer of 1987.

Perimeter and Perimeter:Area

Corrected total map perimeter increased 12.6% (645.5 km) between 1935 and 1981 (Tables 5-5, 5-6), yet mean patch perimeter dropped 11.1% (from 6.23 to 5.54 km) due to the greater decrease in mean patch area. Mean perimeters are lowest for small patches with smooth, circular shapes. Perimeter:area (P:A) ratios measure the relative importance of edge effects on patch interiors (Forman and Godron 1986) - large ratios indicate relatively great edge effects. P:A ratios are a function of both patch size and shape. As patch area increases P:A values decrease, and as patch shape becomes more elongate and dissected P:A values increase. Overall corrected landscape P:A increased from 5.97 to 6.75 km/km^2 between 1935 and 1981, mirroring the increased mean P:A values of the dominant forest and woodland cover-type patches (Tables 5-5, 5-6). A major exception to this trend is the drop in mean P:A for

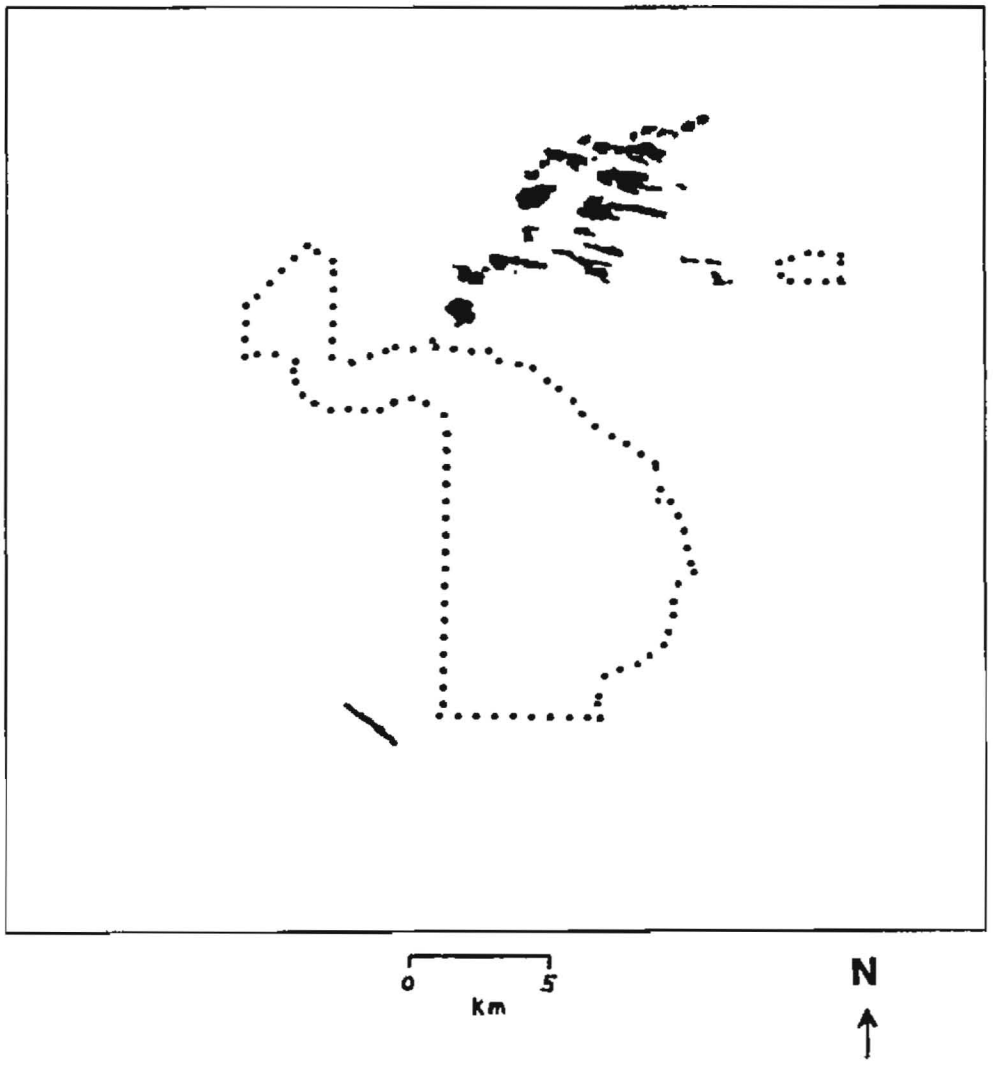


Figure 5-13. Map of 1935 cultural cover-types in the Bandelier area. Bandelier's current boundary is shown with the dotted line, and the shaded cultural cover-type patches are almost entirely agricultural fields.

Figure 5-14. Map of 1981 landscape cover-type patches of cultural origin in the Bandelier National Monument area. Los Alamos National Laboratory technical areas and Cochiti Dam = magenta, residential = yellow, commercial = dark blue, ponds and Cochiti Lake at maximum flood pool stage = blue, pumice mines = black, agricultural fields = green, golf courses = orange, and ski area = horizontal striping. The dotted line is the boundary of Bandelier National Monument.

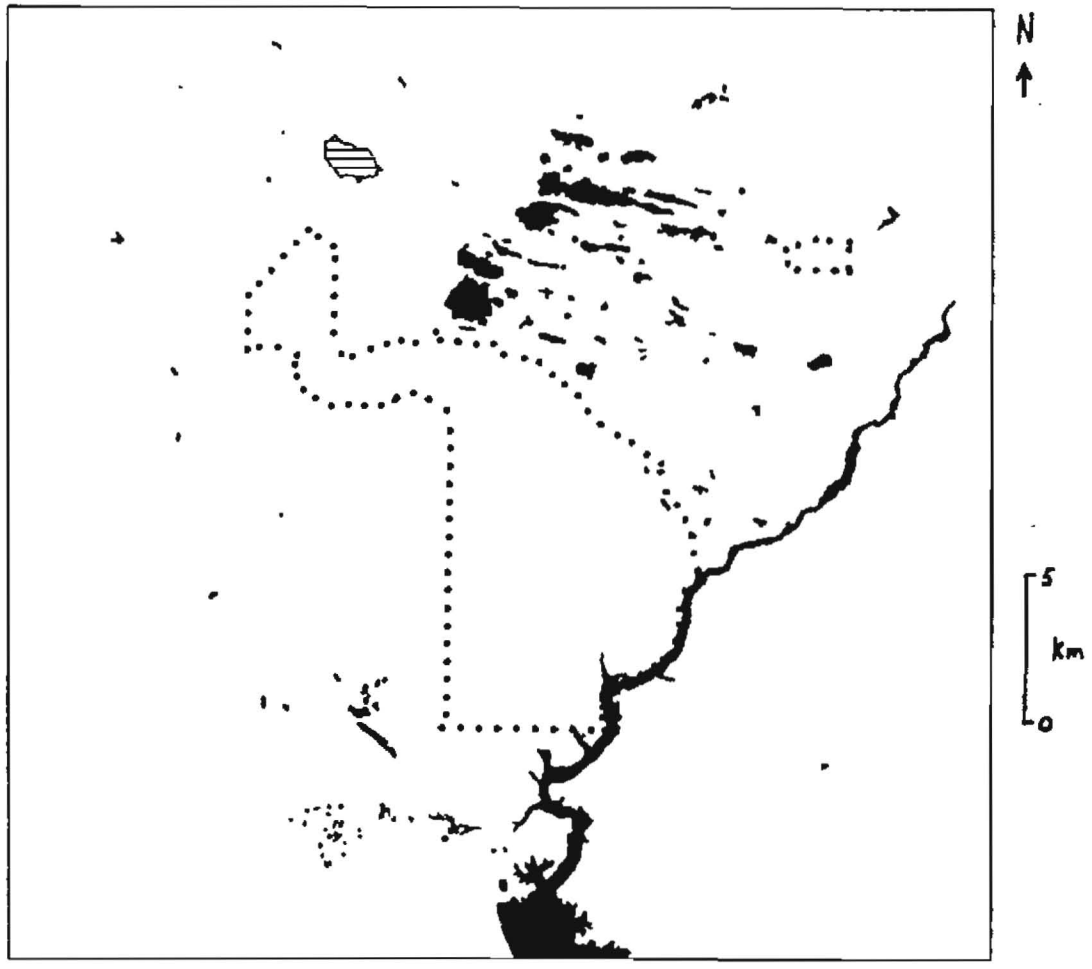


Figure 5-14.

grass patches, reflecting the large increase in mean grass patch size. Edge effects increased for most cover-types over the mapped time-frame, reflecting the fragmentation of this landscape into smaller patches. This relative fragmentation is still much less than that observed in such originally forested landscapes as the Pacific Northwest (Franklin and Forman 1987), the eastern United States (Sharpe et al 1981), or portions of the Amazon Basin (Ellis 1988).

Patch Dissection Index

Patch dissection index (PDI, Sharpe et al 1981) values are also presented in Tables 5-5 and 5-6. A perfectly circular patch has a PDI = 1.0, the lowest possible value. As patch shape becomes more elongated and dissected the PDI increases, with a theoretical maximum approaching infinity. The PDI is a better measure of patch shape than the P:A ratio, as the PDI is not confounded by the influence of patch size. For example, 1935 and 1981 ponds display a low mean PDI as expected from their typically rounded shapes, but because of their tiny sizes their mean P:A value is large. The mean landscape PDI remained essentially constant from 1935 to 1981 (Tables 5-5, 5-6), indicating that although patches became smaller they did not become more elongate or complex in shape, on average. Individual cover-type PDI's reveal differences between patch types and between dates. Most of the new 1981 cultural cover-types have PDI's below the uncorrected mean landscape value of 1.80, indicating relatively compact, simple shapes. The La Mesa Fire apparently created relatively simple grassland patch shapes, resulting in a decrease in mean PDI from 1.26 to 1.19. Both forest and woodland mean PDI's increased, perhaps reflecting increased

internal edge left by the imbedded mosaic of numerous smaller La Mesa Fire and cultural patches. Thus despite a basically constant mean landscape PDI, most carryover patches in this landscape became slightly more complex in shape between 1935 and 1981.

Landscape Diversity, Dominance, and Evenness

Related indices of landscape diversity, dominance, and evenness provide other measures of landscape change (Table 5-7). The landscape diversity index (H) (Romme 1982) is highly sensitive to changes in the number of cover-types observed (m); H increases as m becomes larger and/or the proportion of each cover-type becomes more equal in the landscape. The formulation of the dominance index ($D = H_{max} - H$) reduces the variability of D across different values of m (O'Neill et al 1988). Values of D may range between 0 and H_{max}, with large D values characteristic of a landscape dominated by a small proportion of the cover-types present, and small D values present if cover-types are present in relatively even proportions. Relative evenness ($E = H/H_{max}$ [Romme 1982]) ranges from 1.0 for a landscape in which all m cover-types are equally represented to near 0 for highly uneven proportionate areas.

With m = 7 all 3 indices support the intuitive conclusion that increased numbers of patch types and decreased area covered by the two dominant cover-types (forest and woodland) since 1935 have led to a more diverse landscape, as measured at this coarse level of resolution. While the increase in m_{max} between 1935 and 1981 raised the H value of this landscape, the increased H with m held constant at seven indicates that the cover-type categories covered more equal proportions of the

TABLE 5-7. Landscape diversity (H), dominance (D), and evenness (E) indices for the 1935 and 1981 landscape cover-type maps, calculated using natural logarithms. The number of cover-type categories = m. The 1935/1981 comparison with m = 7 was achieved by combining cover-types into the following categories: forest, woodland, canyon walls, grass/shrub lands, water, urban, and agricultural.

Year	m	Hmax	H	D	E
1935	14	2.639	1.333	1.306	0.505
1981	19	2.944	1.564	1.380	0.531
1935	7	1.946	1.255	0.691	0.645
1981	7	1.946	1.436	0.510	0.738

landscape in 1981. Decreased dominance and increased evenness indices confirm that cover-types are distributed in more equal proportions in the 1981 landscape.

The data in Table 5-7 suggest two reasons why the relative evenness index (E) used by Romme (1982) may be a better measure of relative evenness/dominance than the dominance index (D) advocated by O'Neill et al (1988). First of all, absolute values of D are harder to interpret, since D tends to increase as m increases, while E values are restricted to a known maximum of 1.0. Secondly, this sensitivity of D to m confounds comparisons between maps with different m values, whereas E is insensitive to changes in m. For example, with m = 7, the 1981 map displays lower D and thus higher E values than the 1935 map. When m_{max} is used for both maps the E value continues to indicate that the 1981 map has a closer to equal distribution of cover-types, but a higher 1981 D value suggests the opposite. However, in this case the D value is greater for the 1981 map only because its m_{max} of 19 is larger, incorrectly suggesting that the relative dominance of cover-

types increased since 1935.

O'Neill et al (1988) report dominance values between 0.19 and 1.59 for 94 similar-sized landscapes (m = 7) in the eastern United States. The comparable m = 7 dominance values for the 1935 and 1981 Jemez Mountains landscapes (Table 5-7) fall into the low end of this range. The Jemez Mountains blend of co-dominant forest and woodland categories with significant proportions of canyon walls/rock and grass/shrub lands apparently has more even proportions of cover-type categories than most of the eastern landscapes surveyed by O'Neill et al (1988).

ROAD NETWORKS

Road network mapping across a 187,858 ha portion of the Jemez Mountains reveals a nearly 12-fold increase in total road length from 719 km in 1935 (Figure 5-15) to 8433 km 1981 (Figures 5-16, 5-17, 5-18). These road maps cover the same area as the 1980 land ownership maps (Figure 2-2, Table 2-1), allowing analysis of road development as a function of both road type and land ownership type. Three small ownership categories (State of New Mexico, General Services Administration, and Los Alamos County) are ommitted from discussion here as they comprise < 0.0068 of the map area, although their data are included in totalled values. The ownership type abbreviations used in graphs and tables in this section are: SFNF (Santa Fe National Forest), PRIV (private), PUEB (Native American Pueblos), BNM (Bandelier National Monument), LANL (Los Alamos National Laboratory), BLM (Bureau of Land Management), and ALL (all types combined). Several significant land ownership changes have occurred in the map area since 1935 (cf. Chapter II), so caution must be exercised when attributing observed

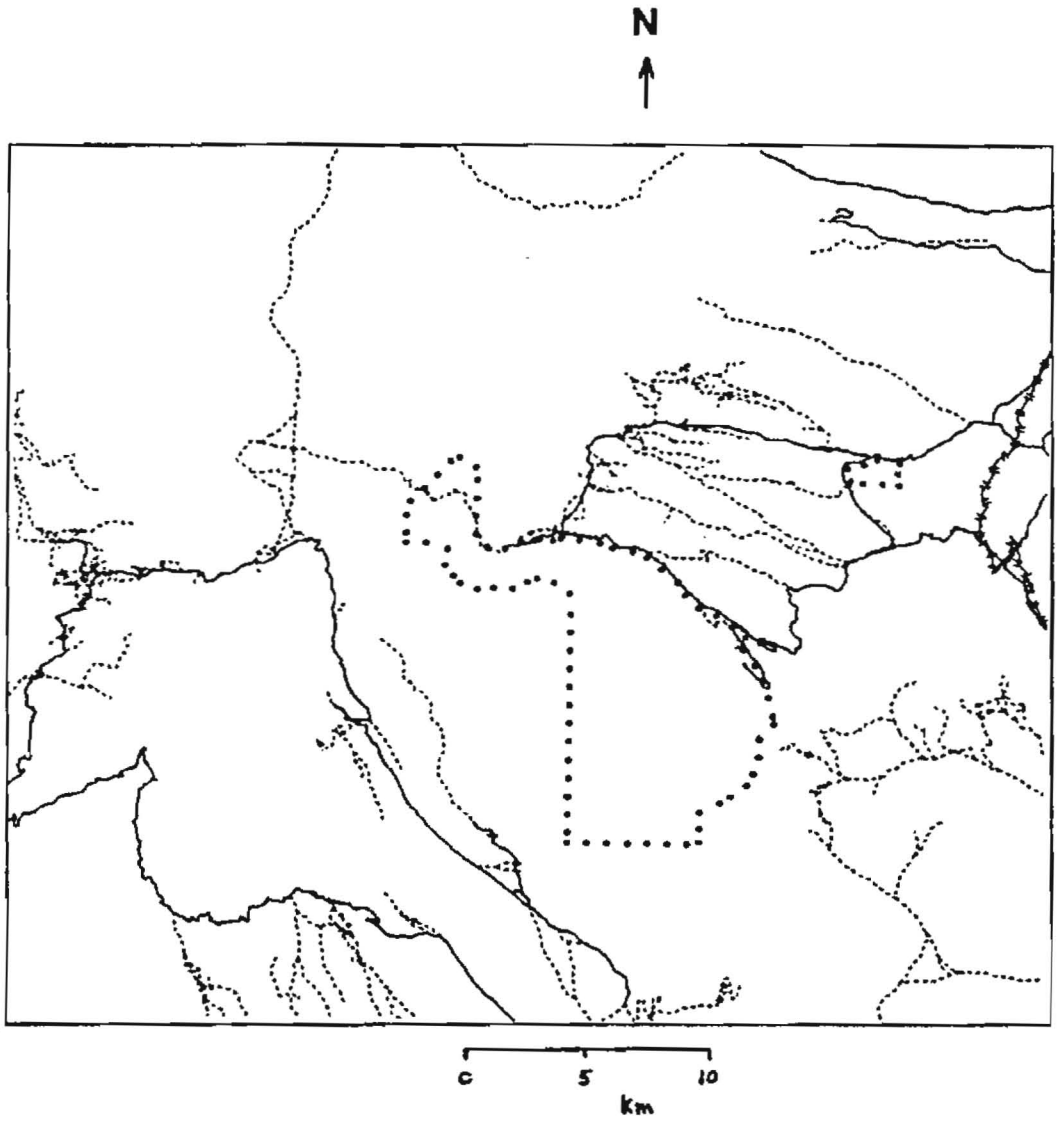


Figure 5-15. Map of all 1935 roads. The crosshatched line is a railroad, the solid lines are dirt roads, the thin dashed lines are primitive roads, and the dotted lines mark the current boundaries of Bandelier National Monument.

Figure 5-16. Map of 1981 paved, improved, and dirt roads across 187,858 ha of the Jemez Mountains. Paved = black, improved = yellow/green speckles, and dirt = magenta. The dotted line is the boundary of Bandelier National Monument.

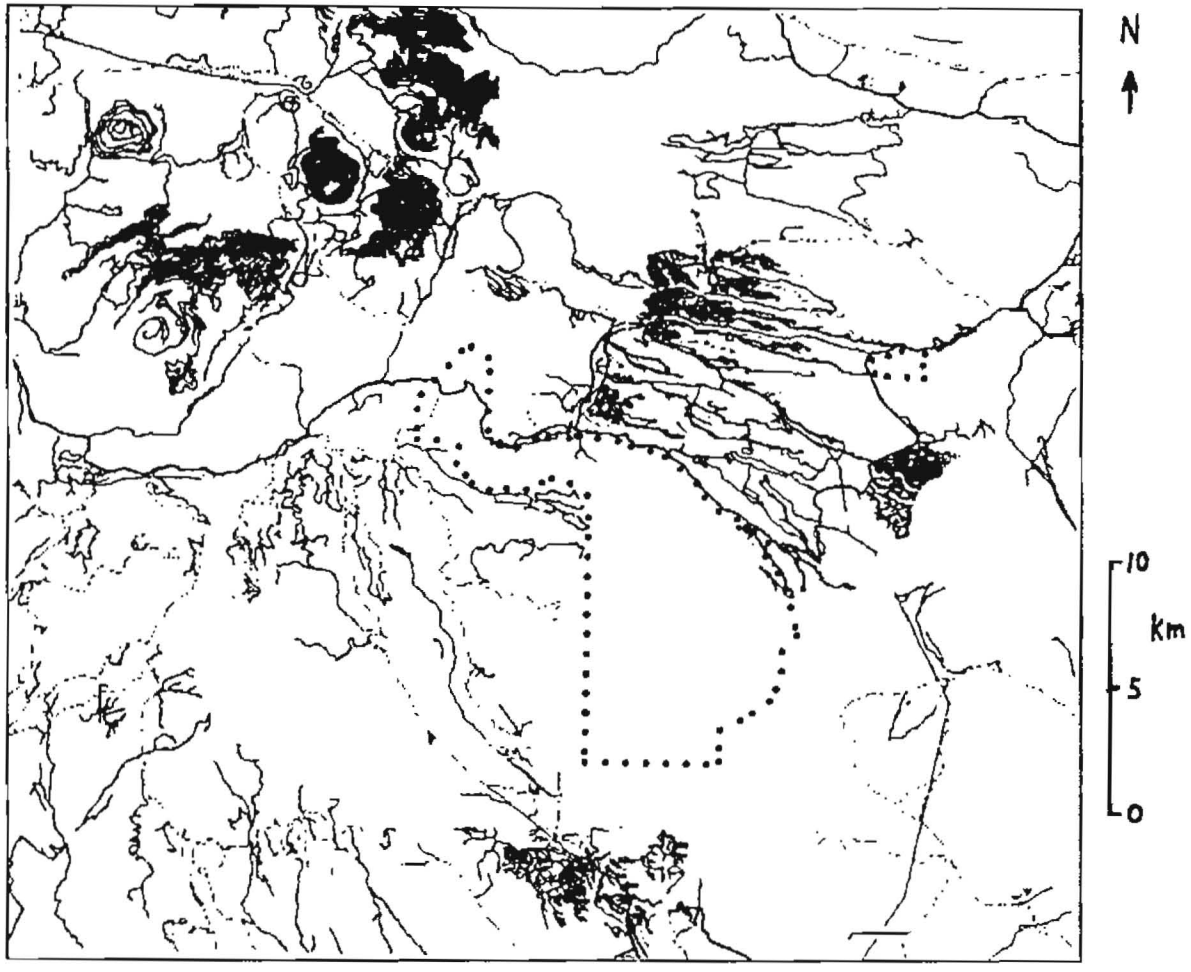


Figure 5-16.

Figure 5-17. Map of 1981 primitive roads (blue) across 187,858 ha of the Jemez Mountains. The dotted line is the boundary of Bandelier National Monument.

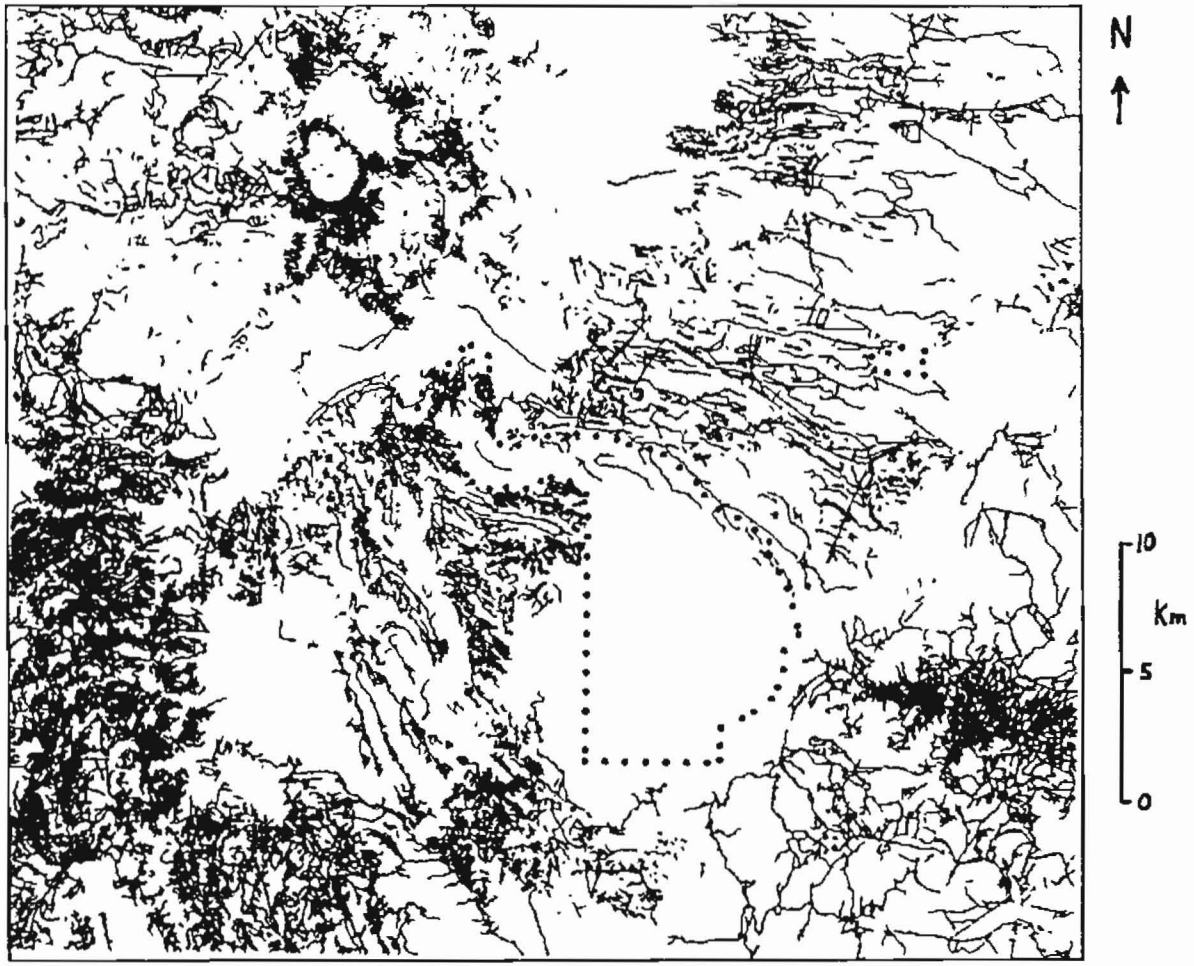


Figure 5-17.

Figure 5-18. Map of all 1981 roads across 187,858 ha of the Jemez Mountains. Paved = black, improved = yellow/green speckles, dirt = magenta, and primitive = blue. The dotted line is the boundary of Bandelier National Monument.

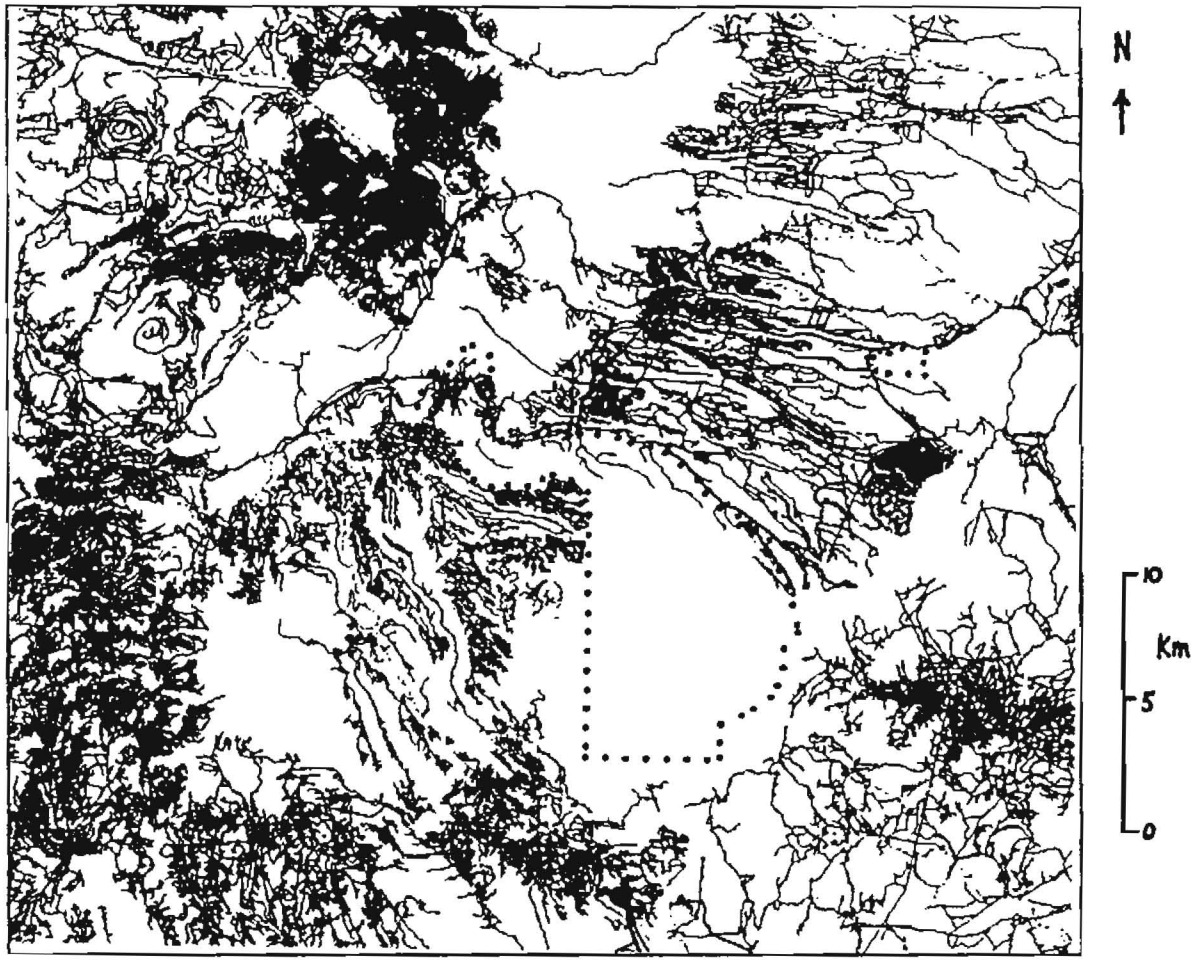


Figure 5-18.

changes in road patterns to the contemporary land owners. These estimates of road surface areas do not include shoulders, cut and fill slopes, or ditches, and thus are conservative estimates of landscape area directly altered by roads.

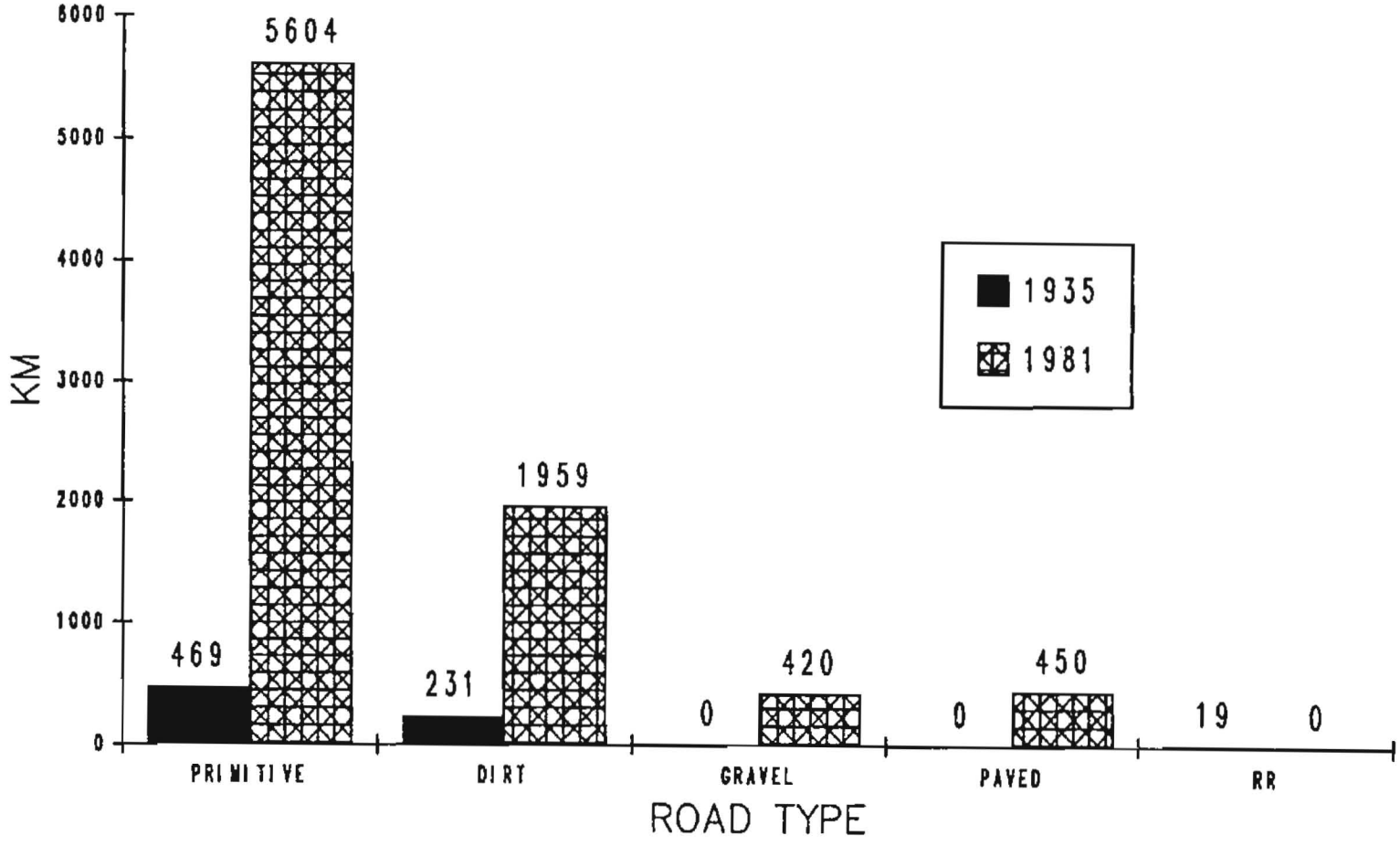
Railroad

In 1935 the Denver and Rio Grande Railroad (the "Chili Line") was still in operation through the edge of the map area, with 18.92 km of track (Figures 5-15, 5-19). The railroad ran through land that is now owned by the BLM, San Ildefonso Pueblo, the SFNF, and private individuals (Table 5-8). This stretch of track was completed between 1880 and 1886 and was an important early link between the Jemez Mountains and the outside world (Rothman 1989); it precipitated some of the landscape changes described here. The railroad was abandoned in 1941.

Paved Roads

Paved roads had not yet reached the Jemez Mountains in 1935, but by 1981 449.61 km of paved roads had been built (Figure 5-19). The paved road network reflects the intensive development activities associated with LANL and the townsites of Los Alamos and White Rock to the north of BNM, and the Town of Cochiti Lake to the south (Figures 5-16, 5-20, 5-21). Total paved road density was 0.19 km/km^2 in 1981, with a high value of 1.60 km/km^2 in LANL (Figure 5-20). Paved road densities were higher in the townsites than on LANL lands, but the near-absence of paved roads in the large Baca Location reduced the PRIV average density. LANL and PRIV contained 77.9% of the total paved road

FIGURE 5-19. Changes in total road length by road type, 1935 - 1981.



1

FIGURE 5-20. 1981 paved road density by land ownership.

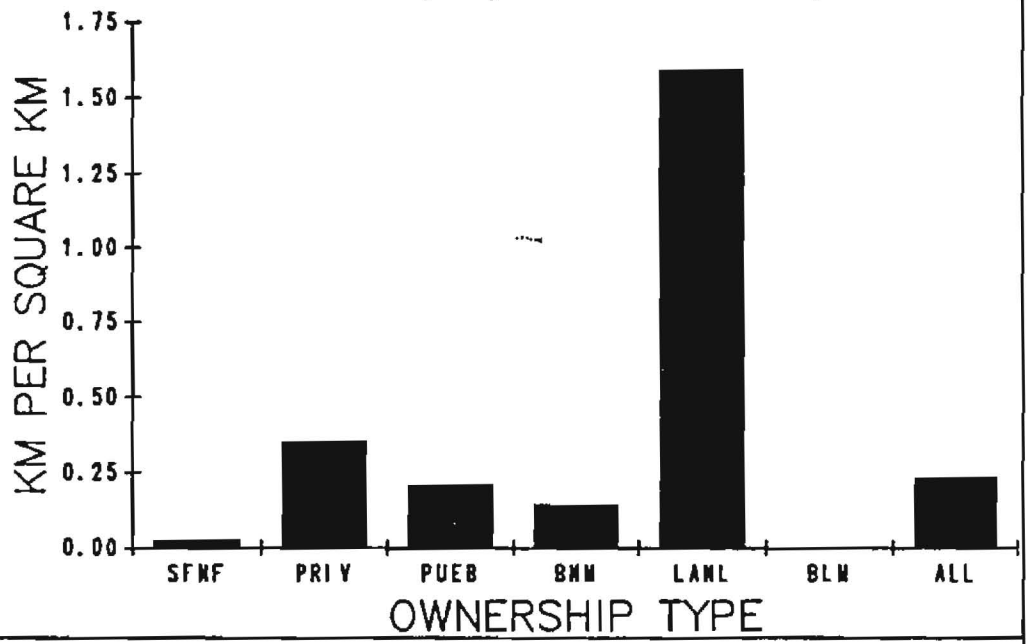


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FIGURE 5-21. 1981 paved road length by land ownership.

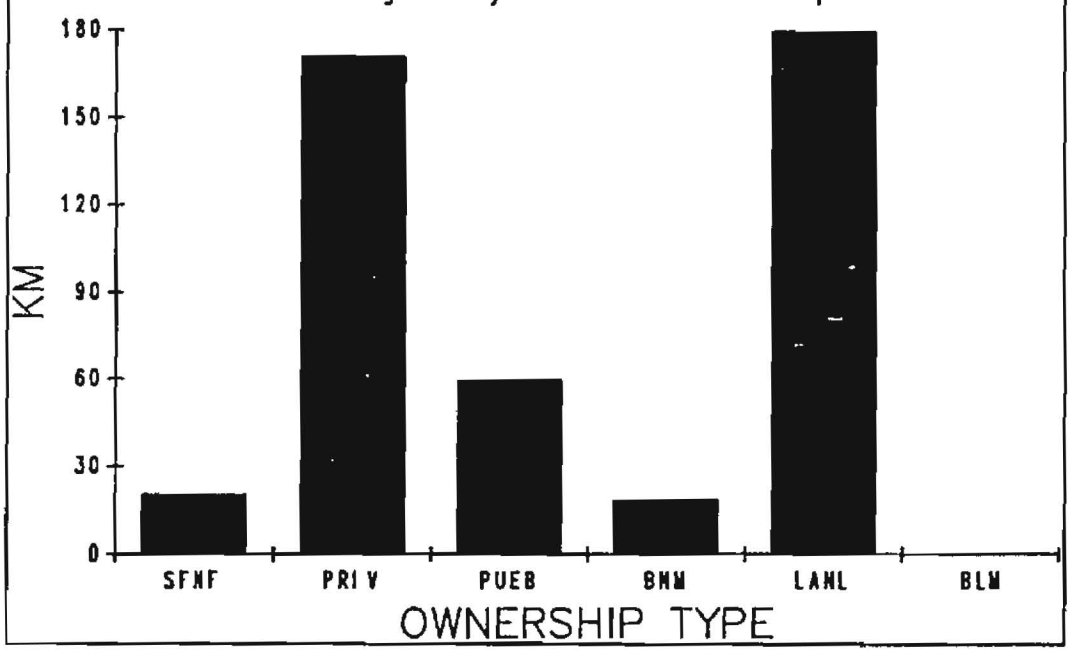

TABLE 5-8. Estimated surface area of roads in 1935 and 1981 by road type and 1980 land ownership. The first row beneath each road type heading lists road surface areas (hectares), while the second row lists the percentage of the modern ownership type covered by the road type.

1935 — Ownership Type

	SFNF	PRIV	PUEB	BNM	LANL	BLM	ALL
Primitive							
	55.1 ha	40.8	15.6	4.0	21.0	4.9	142.8
	0.070 %	0.085	0.056	0.03	0.186	0.070	0.076
Dirt							
	28.7 ha	13.3	28.2	3.6	18.2	5.7	98.3
	0.036 %	0.027	0.101	0.027	0.162	0.082	0.052
Railroad							
	0.9 ha	0.5	2.5	-	-	1.8	-
	0.0012 %	0.0010	0.0091	-	-	0.026	-
Total							
	84.7 ha	54.5	46.3	7.6	39.2	12.4	246.9
	0.107 %	0.113	0.165	0.057	0.348	0.179	0.131

1981 — Ownership Type

	SFNF	PRIV	PUEB	BNM	LANL	BLM	ALL
Primitive							
	899.3 ha	367.3	243.7	28.8	94.8	61.4	1710.1
	1.14 %	0.76	0.87	0.22	0.84	0.89	0.91
Dirt							
	169.5 ha	503.0	92.7	4.0	42.5	21.3	835.4
	0.21 %	1.04	0.33	0.03	0.38	0.31	0.44
Improved							
	129.9 ha	50.3	27.2	1.7	11.8	7.1	230.3
	0.16 %	0.10	0.10	0.01	0.10	0.10	0.12
Paved							
	16.3 ha	135.4	47.3	14.8	141.8	0.2	356.0
	0.02 %	0.28	0.17	0.11	1.26	0.002	0.19
Total							
	1215.0 ha	1056.3	410.9	49.3	290.9	89.9	3131.8
	1.54 %	2.19	1.47	0.37	2.58	1.30	1.667

length (Figure 5-21). Most of the paved road length in BNM was due to the presence of State Highway 4 along and through the park boundary. In 1981 paved roads covered 0.19% (356.0 ha) of the map area (Table 5-8), with 1.26% (141.8 ha) of LANL lands beneath paved roads.

Improved Roads

Most roads in this category have gravelled surfaces, and all have culverts and other improvements. The absence of gravelled surfaces in 1935 was interpreted as a complete absence of improved roads, although the main travelways certainly contained improvements that were not visible on the air photos. By 1981 419.66 km of improved roads were apparent (Figures 5-16, 5-19). Improved roads are the main travelways today for accessing much of the Jemez Mountains, thus their relatively high density and total length on the SFNF (Figures 5-22, 5-23). Improved road densities ranged from 0.025 km/km^2 in BNM to 0.286 km/km^2 in the SFNF, averaging 0.224 km/km^2 across the map area (Figure 5-22). Improved roads covered 0.12% (230.3 ha) of this area in 1981 (Table 5-8), and 0.16% (129.9 ha) of the SFNF.

Dirt Roads

Mapping showed 230.82 km of dirt road in 1935 (Figures 5-15, 5-19). Many of these roads persist in the modern landscape as improved or paved roads (Figure 5-16). Dirt road densities averaged 0.122 km/km^2 in 1935, with a higher density (0.380 km/km^2) around the homesteads that later became LANL land (Figure 5-24). Dirt roads covered 0.052% (98.3 ha) of the 1935 landscape, with a range from 0.027% of BNM to 0.162% of modern LANL lands.

1

FIGURE 5-22. 1981 improved road density by land ownership.

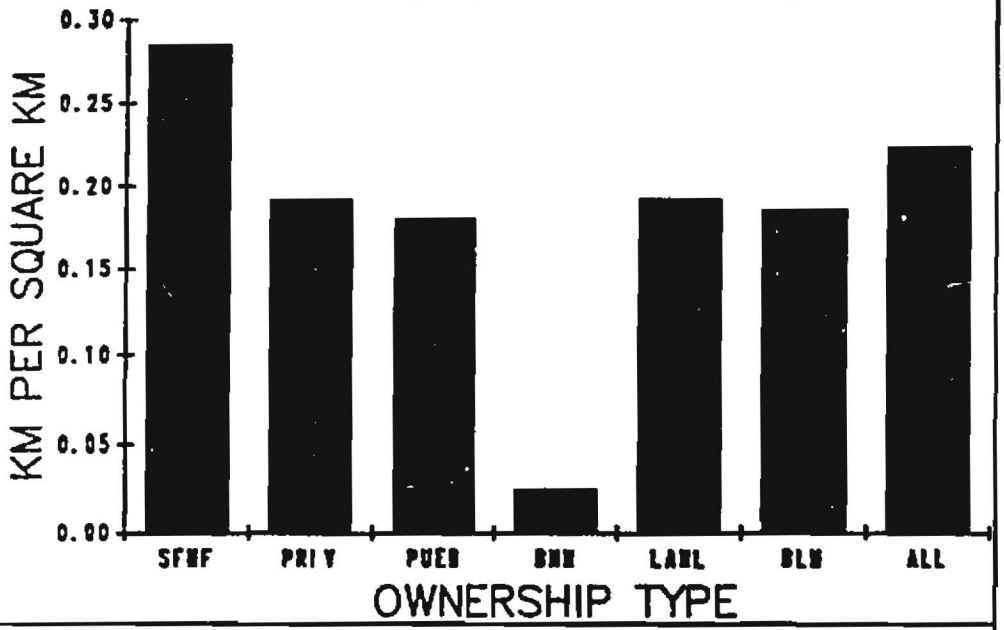

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FIGURE 5-23. 1981 improved road length by land ownership.

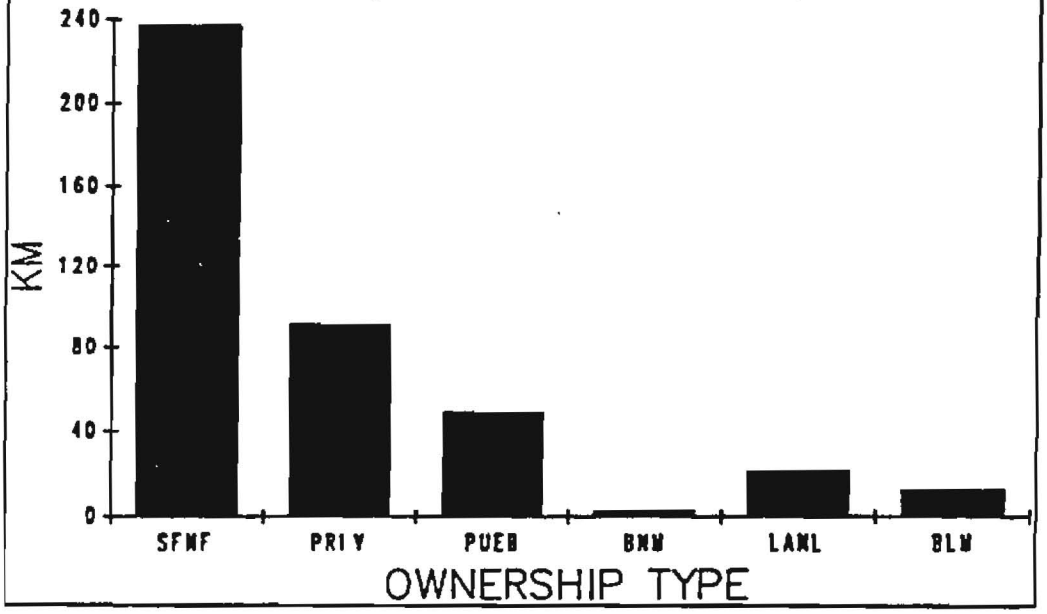

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FIGURE 5-24. Changes in dirt road density by land ownership. Solid bars = 1935, pattern = 1981.

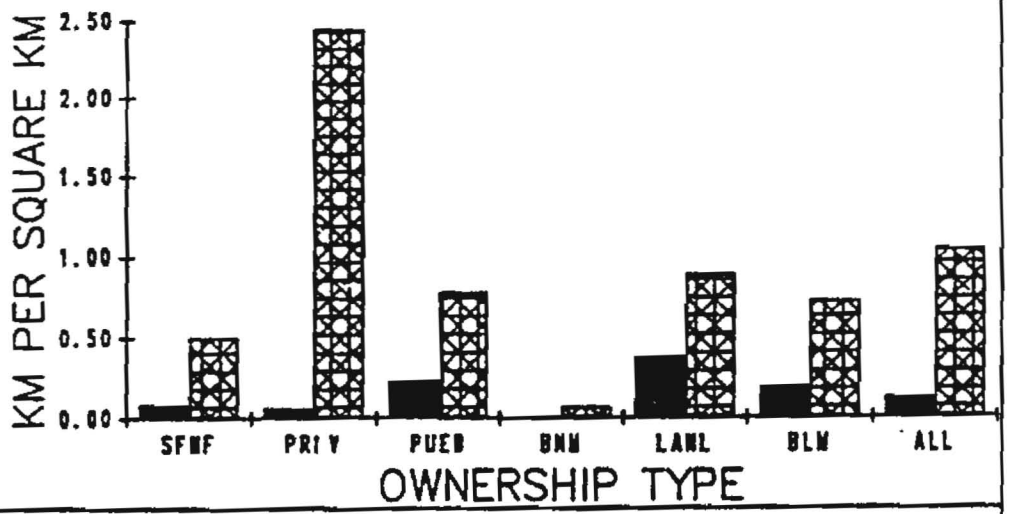

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FIGURE 5-25. Changes in dirt road length by land ownership. Solid bars = 1935, pattern = 1981.

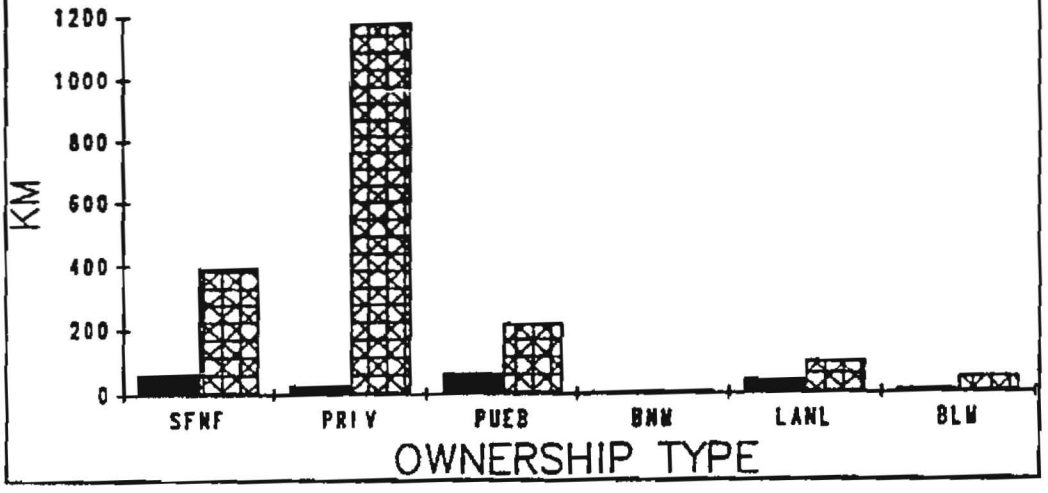

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Figure 5-26. Map of all 1981 roads across the 15,655 ha of the Valle Toledo quadrangle, located along the northern edge of Figure 5-18. Improved = yellow/green speckles, dirt = magenta, and primitive = blue. This road network largely reflects past logging activities.

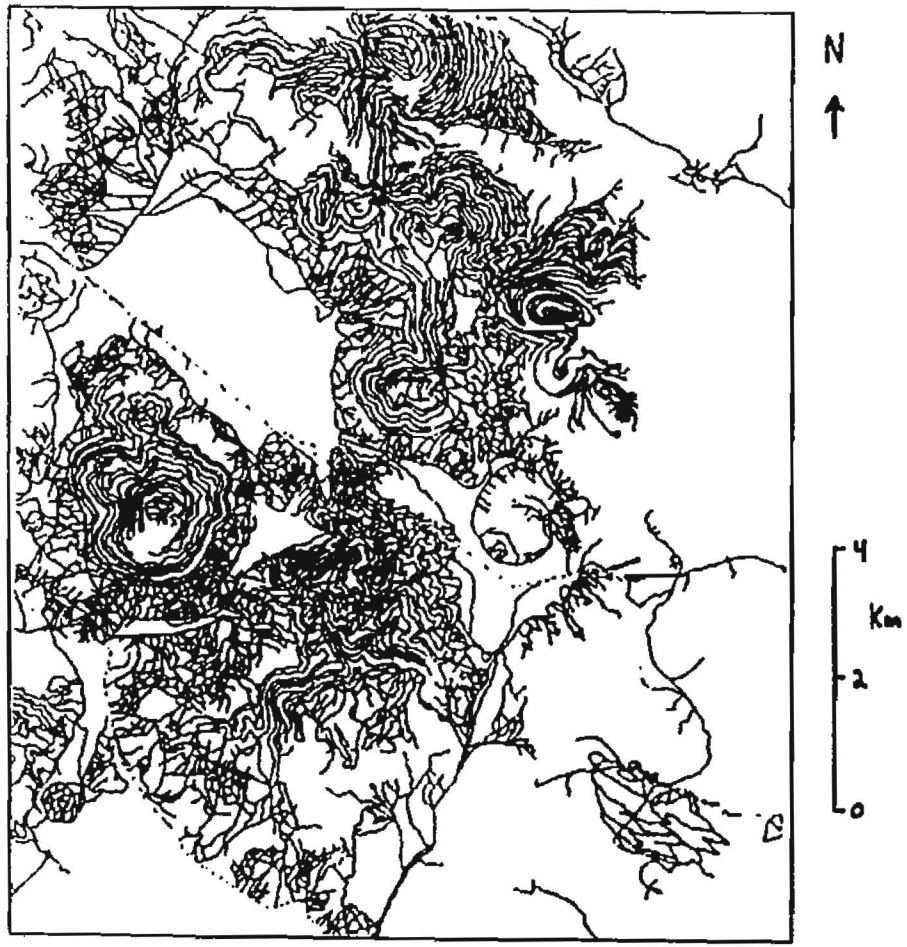

Figure 5-26.

around Graduation Flats and the southeast flank of Cerro Grande in the central map area. I know of 2 sections of narrow rock-bordered road in the Upper Frijoles area, likely part of an old route into the caldera, which invisible on both 1935 and 1981 air photos beneath forest canopies. In addition, an early report on Bandelier's trail system (Attwell 1933) describes "a thousand wood roads", used by wagons for hauling wood, crisscrossing the detached Otowi Section of the Monument (which is now largely LANL land); only a fraction of these roads were likely to have been mapped. Thus my road maps yield conservative estimates of 1935 road networks. Again, homesteading activities on the pre-LANL Pajarito Plateau led to the relatively high density of 0.620 km/km^2 of primitive roads in this area, above the overall density of 0.249 km/km^2 (Figure 5-27). Primitive roads covered 0.076% (142.8 ha) of the map area, and 0.186% of the pre-LANL area.

The 1981 road map shows 5604.27 km of primitive road in the map area (Figure 5-17, 5-19). This is the most variable 1981 road category, including logging skid roads, informal woodcutting tracks, powerline corridors, and off-road vehicle paths. Many of these "roads" represent traces of past human activity that are no longer used. For example, there are old logging roads in the Cerro Grande area that remain almost completely free of tree re-establishment after more than 50 years of non-use (based upon the ages of the few trees growing in the roads). Some of the mapped roads were at the limits of air photo resolution, and since it was impossible to ground truth this whole area some mapped "roads" may prove to be other sorts of ground features, such as interconnected openings in the piñon-juniper woodlands of the Caja del Rio area. Still, other primitive roads certainly existed in

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FIGURE 5-27. Changes in primitive road density by land ownership. Solid bars = 1935, pattern = 1981.

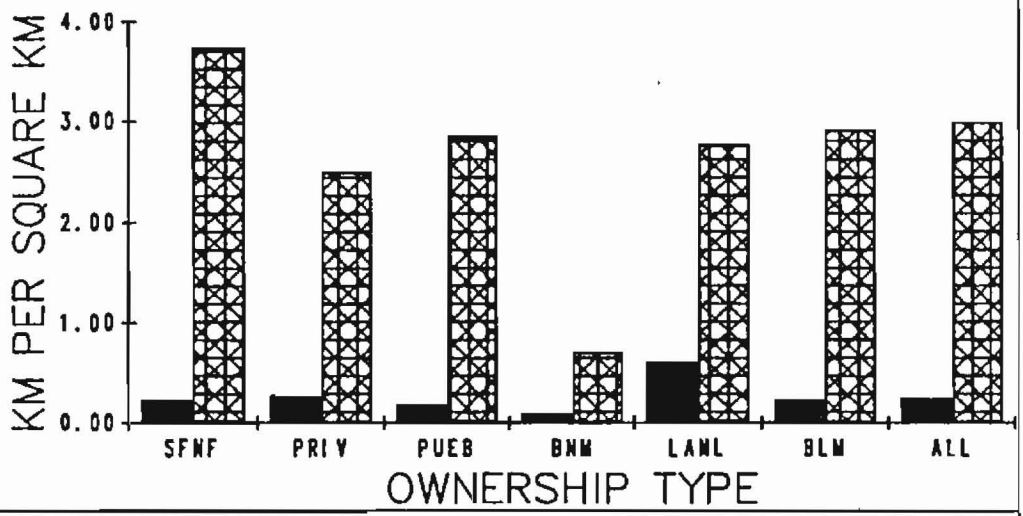

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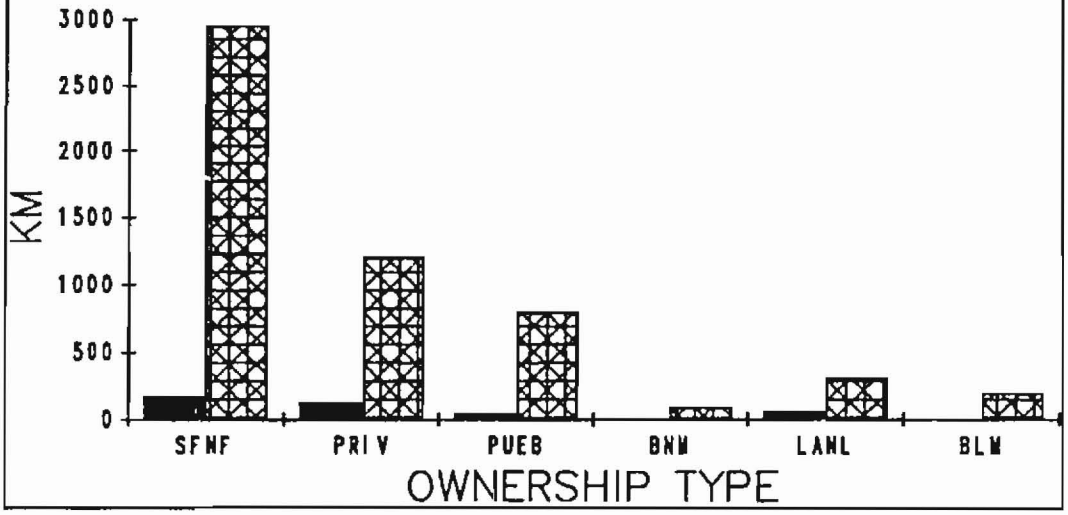
FIGURE 5-28. Changes in primitive road length by land ownership. Solid bars = 1935, pattern = 1981.



1981 that were not mapped. Overall I believe that this road map does not overestimate primitive road densities.

Primitive road densities in 1981 averaged 2.99 km/km^2, with a greater density of 3.74 km/km^2 on the SFNF reflecting extensive logging and woodcutting activities (Figure 5-27). Since 1935 primitive road lengths increased over 16 times on the SFNF to 2952.5 km (Figure 5-28), with large increases due to logging also observed on PRIV (especially the Baca Location) and PUEB (especially Jemez Pueblo) lands. An additional 21.14 km were mapped as "disked" by SFNF site preparation activities on Mesa del Rito in the central map area, leaving a distinctive signature like a plowed farm field on the landscape (not apparent at the resolution of Figure 5-17). The detail of the Redondo Peak quadrangle (Figure 5-29), from the west-central portion of the 12-quad map area, shows the pattern of intensive, primitive road development associated with past logging activities on the SFNF. Primitive roads covered 1.14% (899.3 ha) of the mapped portion of the SFNF in 1981, compared to the 0.91% (1710.1 ha) area average. The increase in primitive roads on BNM was due to the acquisition of lands in the upper Frijoles watershed that had been previously logged (see Forest Cutting above), yet primitive road lengths and densities remain markedly less than on surrounding lands.

Summary - Roads

Consideration of all road types combined from 1935-1981 indicates 5 to 16-fold increase in road density on each ownership type, with a landscape-wide increase of 11.75 times from 0.382 km/km^2 to 4.490 km/km^2 (Figure 5-30). Total road density in 1935 was greatest on the

Figure 5-29. Map of all 1981 roads across the 15,655 ha of the Redondo Peak quadrangle, located along the west central edge of Figure 5-18. Paved = black, improved = yellow/green speckles, dirt = magenta, and primitive = blue. This road network largely reflects past logging activities.

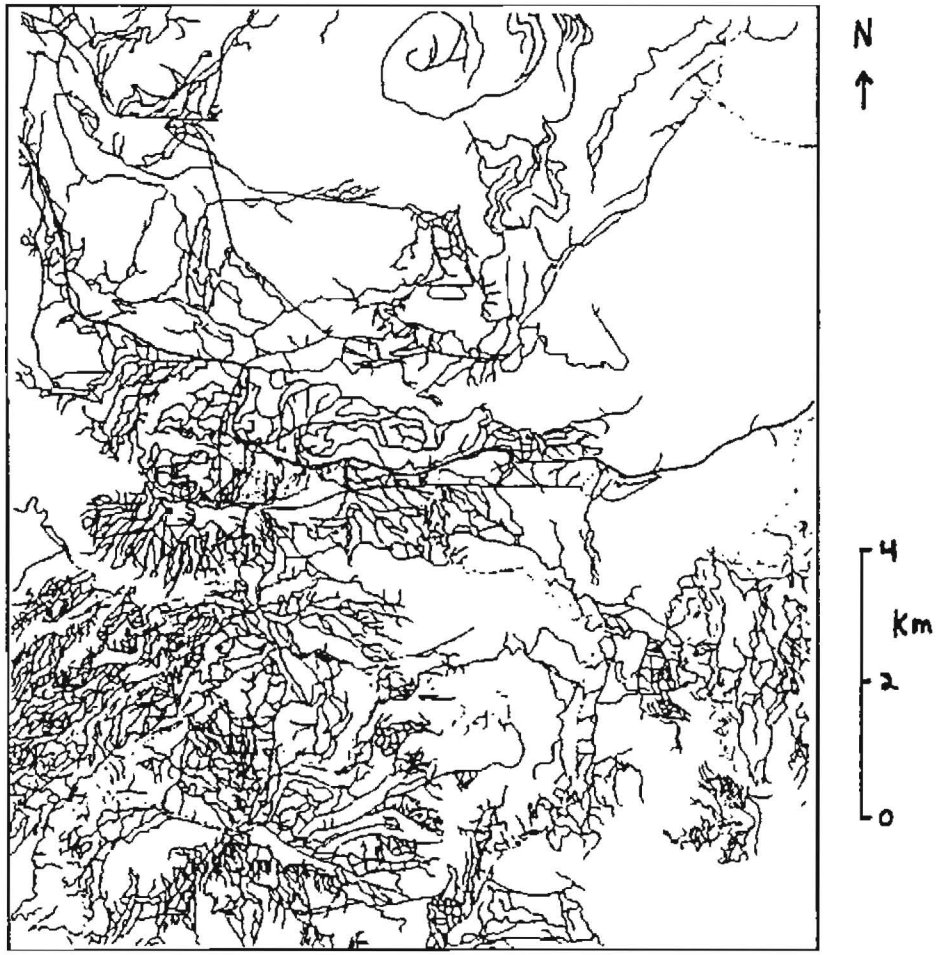


Figure 5-29.

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FIGURE 5-30. Changes in total road density by land ownership. Solid bars = 1935, pattern = 1981.

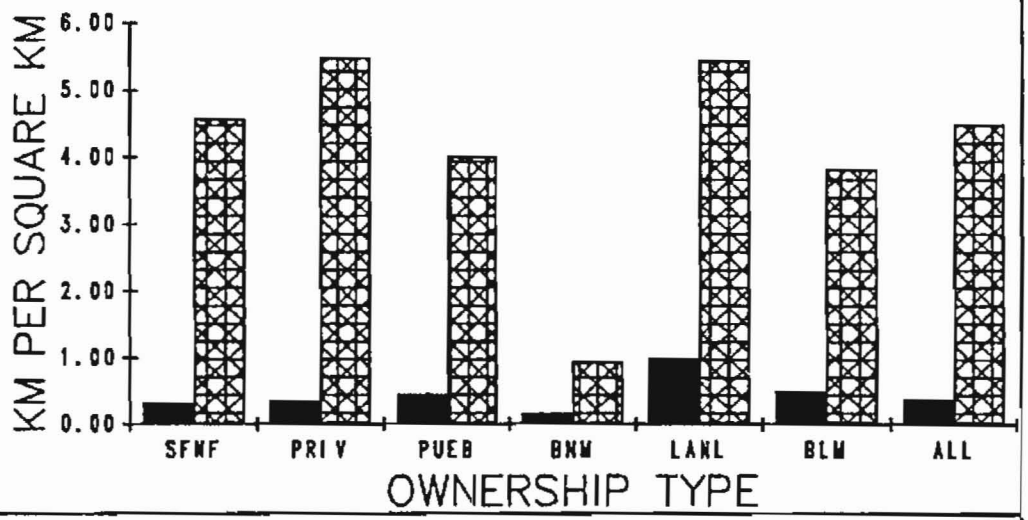

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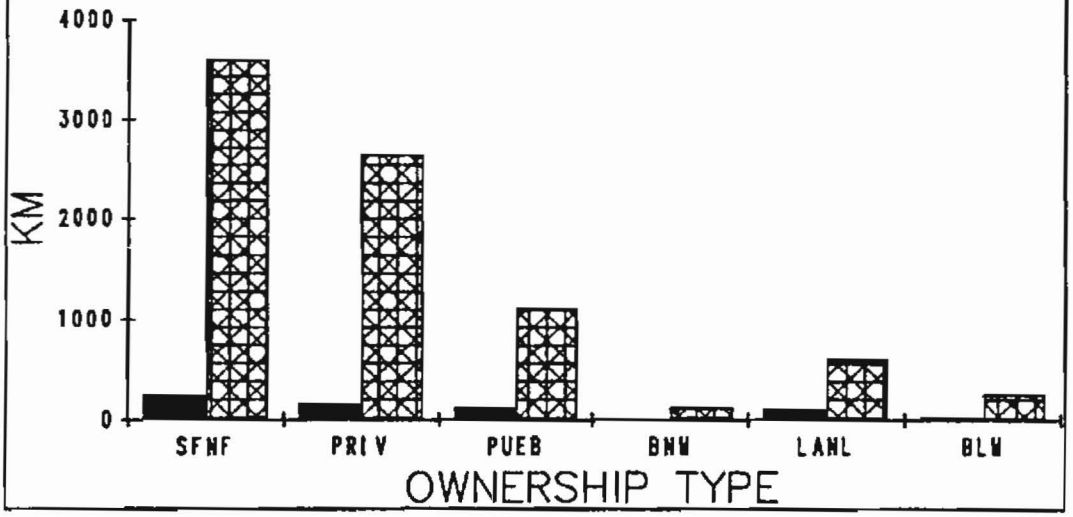
FIGURE 5-31. Changes in total road length by land ownership. Solid bars = 1935, pattern = 1981.



homesteaded lands of the pre-LANL area, with the subsequent industrial development of LANL on these lands keeping them tied with the urbanized and intensively logged private lands for the highest road densities in 1981. Of the total 1981 road length of 8442.9 km, 74.1% was mapped on the SFNF (3607.4 km) and private lands (2648.7 km) (Figure 5-31). Logging on Jemez Pueblo and subdivision roading on Cochiti Pueblo account for much of the 1126.7 km of road mapped on pueblo lands in 1981. Roads in the forested portions of the Jemez Mountains are usually a signature of past logging activity. The roadless area of the Bandelier Wilderness is the largest "hole" in the 1981 road map (Figure 5-18), showing in by far the lowest road densities in this landscspe for BNM. Other areas which had few roads in 1981 include a rugged area of the SFNF north of Ruiz Peak (in the southwest portion of Figure 5-18) the open meadows of the Valles Caldera (northwest quarter) the caldera rim peaks above Los Alamos (north-central region) and the small SFNF Dome Wilderness (2104.4 ha) adjoining the Bandelier wilderness to the west. Estimated total area of road surfaces grew from 0.131% (246.9 ha) of the map area in 1935 to 1.667% (3131.8 ha) in 1981 (Table 5-8). Only 0.37% of BNM was covered by roads in 1981, compared to 1.54% of the SFNF and 2.58% of LANL.

The ecological impacts of road development have been little-studied in the Jemez Mountains, but the order of magnitude increase in road networks observed since 1935 clearly allows the possibility of significant landscape-wide impacts (cf. DISTURBANCE REGIMES, ANTHROPOGENIC DISTURBANCES, Roads). Additional road-building has occurred since 1981. The Santa Fe National Forest Plan recognizes that obliteration and aggressive closure of roads are necessary to prevent

soil loss, improve wildlife habitat, and protect investments (USDA Forest Service 1987-b). The current Forest Plan calls for building 153 km of new road, reconstructing 1151 km of old roads, obliterating 1063 km of road, and closing 3357 km of road between 1987 and 1997, leaving 2898 km of recognized road (USDA Forest Service 1987-b). My data indicate that the road issue on the SFNF may be more difficult to resolve than is currently recognized.

In 1987 the SFNF recognized 1143 km of arterial and collector (improved/dirt) road, and 5941 km of local (primitive/dirt) road on the Forest, for a total road length of 7084 km and non-wilderness road density of 1.37 km/km^2 (USDA Forest Service 1987-b). My data cover 14.875% of the non-wilderness portion of the SFNF. Extrapolating my data to the entire SFNF yields estimated lengths of 1593 km of improved road, 2672 km of dirt road, and 19,848 km of primitive road for a total length of 24,113 km and overall non-wilderness road density of 4.67 km/km^2 for all 3 types combined. Thus my estimate of overall road density is 3.4 times greater than the SFNF estimate.

Several reasons can be suggested to account for this large discrepancy in figures. First of all, my data could have errors, although I believe that my mapping technique, use of professional digitizers, and utilization of a GIS to measure the roads has provided highly accurate results. Secondly, my map area may not be representative of the rest of the SFNF; it seems probable that past logging activity, and thus road densities, are above average here, but not by a factor of over 3. Much of the discrepancey in estimates may simply be due to different definitions of what constitutes a road. The SFNF probably considers most of the primitive roads I mapped as

"closed", and thus does not count them. An assessment of the degree of continued off-road vehicle use and ongoing erosion attributable to closed primitive roads might be desirable to determine which closed roads should be retained in Forest inventories. Finally, the SFNF may have underestimated the magnitude of road development on its lands. Detailed copies of these maps were provided to the SFNF in December, 1988, and will be field-checked as deemed appropriate by USFS personnel (R. Adams, SFNF engineer - personal communication).

LANDSCAPE ELEMENTS OF THE JEMEZ MOUNTAINS

Major vegetation/landform types can be identified in the Jemez Mountains. I will call these readily observable, broad, vegetation/landform types **landscape elements**, as they provide a coarse resolution, structural framework for this landscape. This section summarizes changes within individual landscape elements along an elevational gradient from the Rio Grande to the Valles Caldera of the Jemez Mountains. The prominent landscape elements discussed below are: the Rio Grande riparian zone; the riparian zones of the canyons which dissect the Jemez Mountains; canyon wall vegetation; piñon-juniper and juniper woodlands, juniper savannas, and grasslands of the Pajarito Plateau and Caja del Rio; ponderosa pine forests of the Pajarito Plateau and Sierra de los Valles; mixed conifer forests of the Sierra de los Valles; montane grasslands of the Sierra de los Valles; spruce-fir forests of the Sierra de los Valles; and meadows of the Valles Caldera and Sierra de los Valles.

RIO GRANDE RIPARIAN ZONE

Extensive ecological changes along the Rio Grande on the periphery of the Jemez Mountains have been documented by Dick-Peddie et al (1984), Hink and Ohmart (1984), and Potter (1981). Dick-Peddie et al (1984) review 5 sets of aerial photos taken between 1935 and 1983 of the 9-mile stretch of the Rio Grande immediately upstream from the Jemez Mountains, finding "considerable changes... in the river channel during this period, with a general trend toward the containment of the river in a single channel rather than the braided nature of multiple channels of earlier years." Riparian tree cover was mapped, revealing a gradual decline of 15.9% between 1935 and 1962, a dramatic 63.3% increase to a 1977 peak, and a 7.9% decline by 1983, leaving 26.4% more riparian tree cover than was present in 1935 on this stretch of the river. Dick-Peddie et al suggest that these changes in riparian tree cover reflect localized variation in rates of agricultural clearing and/or firewood and viga (roof beam) cutting over this time period. Hink and Ohmart (1984) observe that water regulation of the middle Rio Grande with dams, levees, and channelization has greatly altered the dynamics of this river system, with resultant diminished opportunities for native cottonwood and willow trees to regenerate on flood-bared or deposited sediments. They conclude that these changes in the fluvial system are also fostering the continuing spread and increasing abundance of alien plants, most notably Russian olive (Elaeagnus angustifolia) and tamarisk, which now commonly dominate riparian vegetation understories and even overstories in some areas.

Potter (1981) examined the impacts of the 1979 inundation of Bandelier's Rio Grande riparian zone by the Cochiti Reservoir, when

water levels peaked at 1643.2 m. This 2-month flood drowned many plants and killed others through burial in dense layers of silt. The more extreme flooding of the 1985-1987 period caused even greater vegetation kill and sediment deposition. The inundation of Frijoles Spring has apparently extirpated at least 6 species of plants from BNM (see **Human-Caused Extirpations** above). Potter expressed concern over the introduction of large numbers of tamarisk into this riparian area, considering their potential to dominate such sites. Many of the tamarisk plants which became established after 1979 were killed by the extended inundations from 1985-1987; as of the summer of 1988 only small numbers of re-established tamarisk seedlings were observed. Potter advocated aerial seeding of the area with native plants and establishment of cottonwood trees after major floods to minimize tamarisk invasion. Such revegetation efforts would also reduce the post-flood development of thickets of alien agricultural weeds, such as those which currently dominate much of the area flooded in 1987. Sediment deposition at the head of Cochiti Lake is creating a growing delta with potential as an increasingly important wetland habitat for many wildlife species (Johnson, 1987).

RIPARIAN ZONES OF CANYONS WHICH DISSECT THE JEMEZ MOUNTAINS

Less is known about the ecological patterns and changes of the riparian zones found in the numerous canyons which dissect the Jemez Mountains. The perennial or intermittent presence of surface water allows productive and floristically rich vegetation to develop in these canyon bottoms (Jacobs 1989). Surprisingly frequent surface fires occurred prior to 1900 in the mesic environment of Upper Frijoles

Canyon (see DISTURBANCE REGIMES, NATURAL DISTURBANCES, Fire). The subsequent cessation of the surface fire regime has allowed mixed conifer tree species to greatly increase in density, based upon visual estimation of tree age-classes. Analagous forest changes are discussed in greater detail below under **Mixed Conifer Forests of the Sierra de los Valles**. The 1977 La Mesa Fire devastated long stretches of old-growth, mixed conifer forest in the dendritic canyon network which cut into ponderosa pine-dominated Escobas and Burnt Mesas; the more xeric microenvironment and loss of seed sources associated with complete removal of tree cover along many sections of this canyon network indicate that recovery to mesic vegetation conditions will be slow. Canyon riparian zones are often a focus for human land uses such as the visitor center at BNM, roads and technical areas on LANL lands, and roads and livestock grazing on SFNF land.

CANYON WALL VEGETATION

Canyon walls are mentioned here primarily to bring up our ecological ignorance of these difficult-to-study areas, as noted by Cully (1986) who did some preliminary classification of canyon wall shrub communities. Undifferentiated canyon walls cover 12.9% and 11.8% of the 1935 and 1981 landscape cover-type maps (Tables 5-5, 5-6), and 13.4% of the 1981 detailed patch map of the immediate BNM area (see THE LOCAL LANDSCAPE OF BANDELIER NATIONAL MONUMENT below); the surface area of steep canyon walls is greater than indicated by my maps which are based upon vertical aerial perspectives.

Most canyon walls support vegetation, not just barren rockland. Grasslands, piñon-juniper savannas and woodlands, and shrub communities

predominate at low elevations and on south aspects, with forests cloaking higher elevation canyon walls. The slopes and benches of White Rock Canyon display more diverse and vigorous herbaceous plant communities than the piñon-juniper woodlands on adjacent mesas. While deer certainly use canyon walls, reduced browsing intensities may provide a refuge for favorite food plants like mountain mahogany which seldom displays its typical hedged mesa-top form on canyon wall sites. Exceptionally interesting and diverse plant communities are found at springs on the lower slopes of White Rock Canyon (B. Jacobs - personal communication, personal observation). The La Mesa Fire burned with variable intensity in the canyon wall forests of Upper Frijoles Canyon, leaving a mosaic of forest and non-forest vegetation. The presence of patches of shrub oak and aspen as interruptions within canyon wall forest vegetation indicates a past fire history that includes patchy crown fires on some sites. These variable and relatively unknown canyon wall systems deserve greater attention more in proportion to their extensive physical presence in this landscape.

PIÑON-JUNIPER AND JUNIPER WOODLANDS, JUNIPER SAVANNAS, AND GRASSLANDS OF THE PAJARITO PLATEAU AND CERROS DEL RIO

Definitive research on change in these landscape elements has not been conducted in the Jemez Mountains, but synthesizing my field observations with published research from other areas (e.g. Tausch et al 1981, Rogers 1982, West 1984, Van Pelt and West 1987, Evans 1988) and consultations with local resource managers and researchers leads me to propose the following general scenario of change in local piñon-juniper woodlands. A great deal of variability exists within and

between local piñon-juniper woodlands, juniper woodlands, juniper savannas, and low elevation grasslands which undoubtedly is reflected in somewhat different histories, but it seems likely that a pattern of change similar to that described below for piñon-juniper woodlands occurred in all of these landscape elements.

Prior to the 1860's surface fires swept through piñon-juniper woodland sites roughly every 15 to 40 years, favoring the maintenance of open woodlands, savannas, and grasslands by killing young piñon and juniper trees. Relatively dense grasses, herbs, and associated organic litter protected the soils from erosion and carried the fires which restricted the density of piñon and juniper. The introduction of large numbers of livestock onto the Pajarito Plateau by 1880 (Rothman 1989) apparently triggered a number of related changes on these sites. Overgrazing caused sharp reductions in the herbaceous ground cover and organic litter, suppressing the former fire regime. Reduced cover of herbaceous plants and litter led to decreased water infiltration and increased surface runoff from the typically intense local rainfall events - thresholds were reached which initiated accelerated erosion. Many young piñon and juniper trees established in the absence of thinning fires and competing herbaceous vegetation, with increases in tree density continuing to the present on mesic sites. 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. These changes apparently interacted as a positive feedback cycle in which decreased herbaceous ground cover promoted tree invasion

and erosion, which in turn fostered further decreases in ground cover.

Once initiated, this cycle is apparently difficult to break (Evans 1988). Soils which may have largely formed under more mesic climate and vegetation conditions during the Pleistocene are obviously eroding at unsustainable levels throughout piñon-juniper woodlands in the Jemez Mountains today (B. Simms, USFS hydrologist - personal communication), and herbaceous vegetation 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 BNM area in which erosion is currently occurring, although burro populations were building up south of Frijoles Canyon by this time which likely furthered subsequent erosion. North of Frijoles Canyon, where burros have never been a significant factor, removal or reduction of domestic livestock was accomplished by 1943, with absolute exclusion of livestock certain since 1959, yet here serious erosion continues and may even be worsening. Large areas of BNM are becoming piñon-juniper "rocklands" as their 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 erosion cycle began.

The re-establishment of herbaceous ground cover under these desertified conditions is difficult. Heavy utilization of the current herbaceous vegetation by animals ranging from harvester ants (e.g. Pogonomyrmex occidentalis) and mice (Peromyscus spp.) to elk may be limiting the availability of seed sources in many woodland areas (cf. Carlson 1987). Decreases in organic matter litter as a mulch, and reduced water infiltration, have caused the soil surface

microenvironment to become more xeric and experience more extreme temperatures, inhibiting seedling establishment. The relatively porous, organic matter-rich, surface soil horizons have eroded off of many sites, leaving relatively impermeable clay-enriched horizons exposed; these present more difficult nutritional and water-balance challenges for prospective seedlings. Winter freeze-thaw activity churns the top soil layer and creates polygonal cracking patterns in bare soils in the local piñon-juniper zone that must kill or damage the roots of seedlings that managed to establish successfully the previous summer.

The historic nature of accelerated soil erosion in local piñon and juniper woodlands may be indicated by the discontinuous pattern of entrenched Anasazi trails found on portions of the Pajarito Plateau, e.g. near Tsankawi. Local archeologist Rory Gauthier and I have traced a number of these trails, which are typically entrenched 30 cm or more into the tuff bedrock on geomorphic sites that likely always lacked soil cover, particularly steep slopes, rocky mesa-top knickpoints, and extremely narrow mesa surfaces. Multiple parallel trails exist at some sites, and these trails typically extend from 5 to 30 m at a site before gradually becoming invisible. These now-isolated sections of entrenched trail were clearly once part of connected trails that basically follow obvious linear paths of greatest ease of travel along the steplike topography of the long, narrow mesas; these trails may have been formalized routes between prehistoric settlements as late as the 1500's. Presumably rutted paths in a soil mantle were present between the rock-entrenched trail segments when in use during Anasazi times. Today the isolated segments of remnant trails are often

separated by stretchea of bedrock that display no trace of a trail, even immediately adjacent to entrenched trail remnants. We interpret this as evidence that post-Anasazi erosion has removed soil mantles from mesa surfaces on the Pajarito Plateau. While localized soil erosion must have occurred around Anasazi habitation and agricultural sites it seems that their activities did not trigger landscape-wide erosion, possibly due to the small size, dispersed locations, and short duration of use of their farm 'fields'.

To this scenario of current piñon-juniper and juniper woodlands as greatly altered artifacts of historic human activities must be added the impacts of the previously described drought and bark beetle outbreak of the 1950's. Field point sampling indicates that the piñon mortalitiy associated with this outbreak caused the ecotone between piñon-juniper and juniper woodlands to erratically shift about 1 km upslope in the Frijoles Watershed. Still, the most significant changes in these landscape elements involve diminished and altered herbaceous ground vegetation, fire suppression, increased tree densities, and accelerated soil erosion.

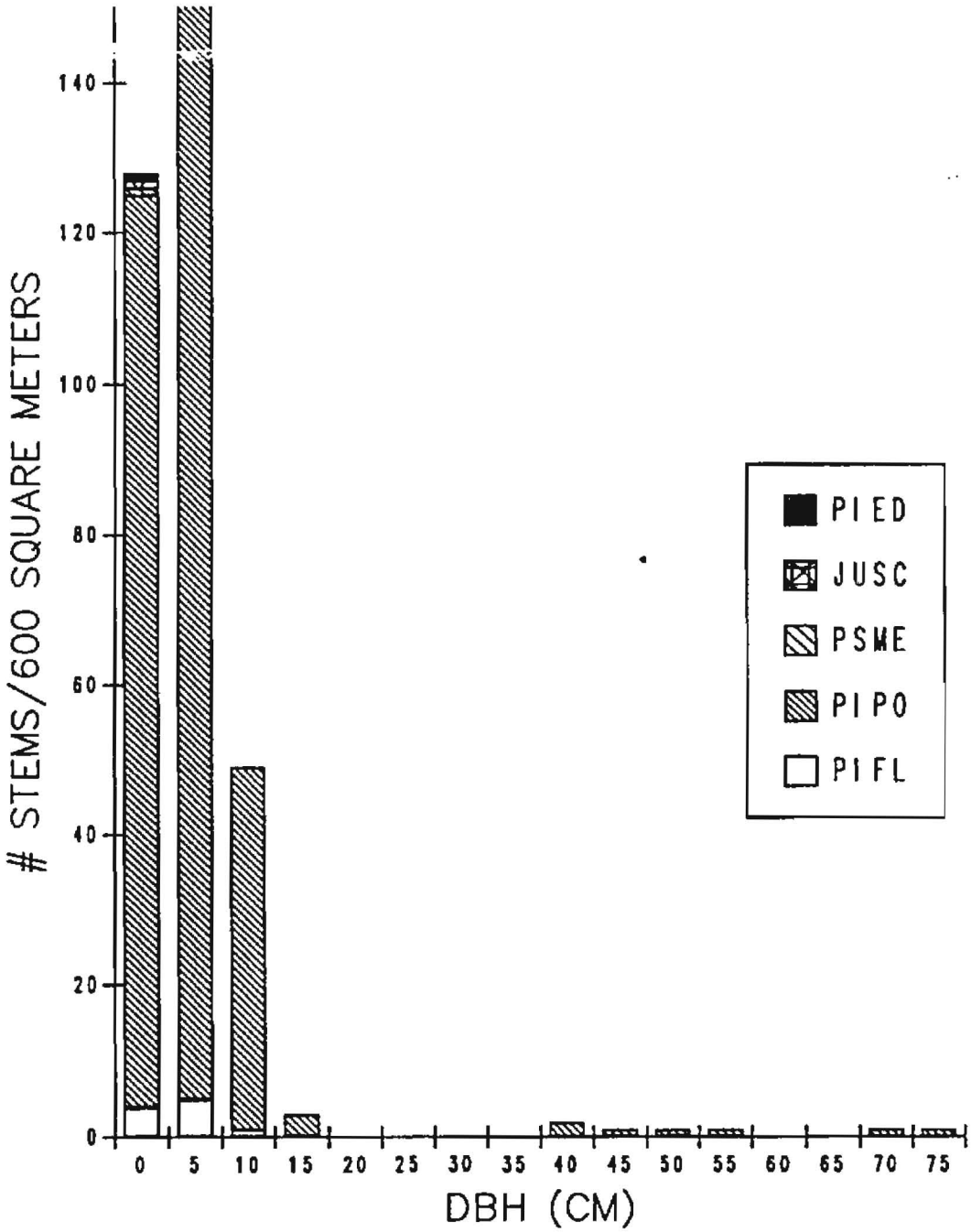
PONDEROSA PINE FORESTS OF THE PAJARITO PLATEAU AND SIERRA DE LOS VALLES

A century ago open ponderosa pine forests covered the mid and upper portions of the Pajarito Plateau, extending upward into the Sierra de los Valles on southerly aspects and extensive surfaces like Apache Mesa. A surface fire regime with mean fire intervals between 5 and 15 years favored a grassy understory and helped kept the pine density in check. By 1900 livestock reduction of the grassy surface fuels had instituted **de facto** fire suppression, which graded into the

institutionalized fire suppression of this century.

Effective fire suppression has had pervasive effects on local ponderosa pine forests. The most obvious result has been great increases in the density of local pine forests similar to those documented in Arizona by Cooper (1960). For example, Figure 5-32 displays the size-class structure typical of the 'pristine' Monument Canyon Research Natural Area (RNA), an unlogged 259 ha preserve located in the SFNF west of BNM in the Jemez Mountains. Established in 1932, this is the oldest RNA in New Mexico. This two-tiered stand displays an old-growth density of 100 stems/ha, with an understory thicket of stagnant saplings and poles that raises the total stand density to an incredible 21,617 stems/hectare. The sampled "doghair" thicket represents extreme density conditions, but similar conditions prevail throughout the RNA. Two dated understory saplings of typical size were established in 1933 and 1939, although these suppressed trees might have missing rings which would increase their true age. Twentieth Century suppression of the frequent surface fires that would have thinned this massive regeneration pulse has allowed a highly unnatural stand structure to develop. Similar stand structures and high fuel loads prevailed in the area burned by the 1977 La Mesa Fire (Forester 1976, Foxx and Potter 1978), resulting in an intense crown fire that converted the heart of Bandelier's ponderosa pine zone into grasslands (Figure 4-16) which will be slow to return to forest (Potter and Foxx 1986). Crown fires in ponderosa pine forests are notably absent in local and regional fire scar records (Swetnam - in press), which is another indication that widespread pine thickets are a historic aberration attributable to modern human fire suppression. By trying to

FIGURE 5-32. Ponderosa pine forest size-clas distribution, Monument Canyon Research Natural Area.



'protect' ponderosa pine forests from fire we have actually fostered conditions that can destroy these forests (Moir and Dieterich 1988). From this perspective the admonition emblazoned on the Monument Canyon RNA boundary sign acquires ironic overtones: "This area must be preserved in a natural state as near as possible".

Just inside the Monument Canyon RNA is a small Anasazi archeological site that has been damaged by the access road and past pothunting. This site is recorded as a 2-room field house, dated between 1330 to 1630, which may have been used while tending nearby farm fields. Today this site is shrouded by a ponderosa pine thicket that would make Anasazi farming impossible. Twenty meters away is an old ponderosa pine log with a catface that displays at least 10 individual fire scars. This forest has clearly changed a great deal since Anasazi farmers lived here, and the primary cause, fire suppression, is apparent.

Almost 100 years of fire suppression has also changed local ponderosa pine forests in less obvious ways. Recent research conducted in Jemez Mountains ponderosa pine forests by White (1986a, 1986b) shows that the buildup of monoterpenoid compounds associated with fire suppression may inhibit nitrification in ponderosa pine ecosystems, and the general importance of fire in controlling nutrient cycling in such forests is well known (Woodmansee and Wallach 1981).

Another significant change in local ponderosa pine forests occurred during the 1950's bark beetle outbreak described previously. Measurements taken along a 21 km-long stretch of the mapped forest/woodland boundary on the landscape cover-type maps show that the ecotone of the ponderosa pine zone shifted 1-3 kilometers upslope

between 1935 and 1981. Field sampling and historic records confirm that this ecotone shift was caused by the drought-triggered beetle outbreak. The local severity of this outbreak may have been exacerbated by the dominance of large, 300+ year old trees along the former ecotone (cf. Group 1 inside dates in Table 4-2) - these old trees were mostly killed by the outbreak, while young trees survived on some sites (personal observation).

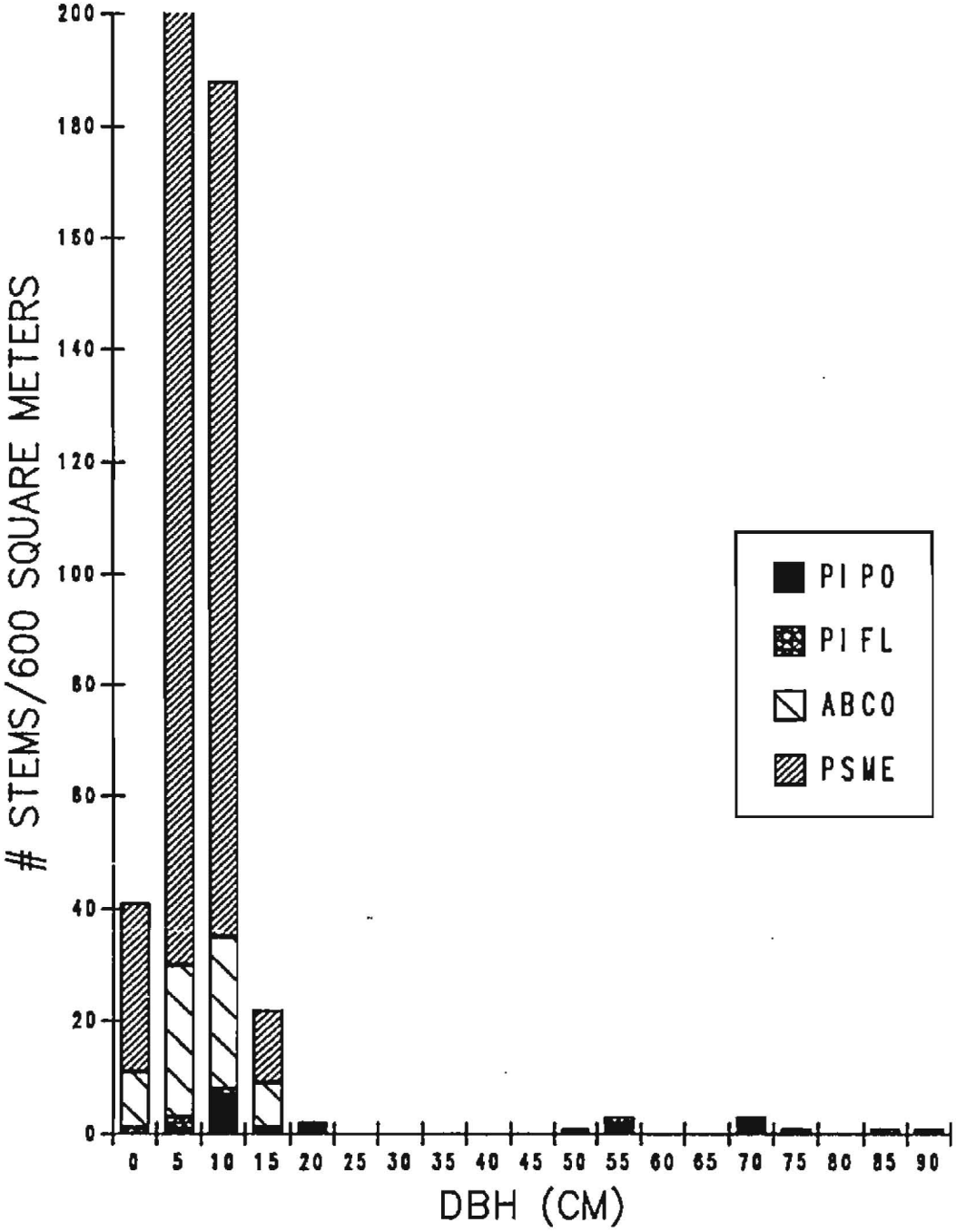
The 1935 air photos and field evidence show that prior to the outbreak ponderosa pine extended farther down-mesa around the Frijoles watershed than on adjacent portions of the Pajarito Plateau. This tongue of low-elevation ponderoso pine coincides with the main depositional field of the El Cajete pumice, which provides coarse-textured soils that allow high rates of deep water infiltration and mulch the sites, allowing ponderosa pine to grow at lower elevations than on comparable non-pumice sites (cf. Sala et al 1988). Field observations show that ponderosa pine currently grows at its lowest mesa-top elevations on pumice soils, and that it previously grew to even lower elevations on such pumice sites. Also, the presence of a thunderstorm track over this lobe of PIPO (U.S. DOE 1979) may increase locally available moisture in most years. However, neither the pumice soils or usual storm track adequately compensated for the extreme 1950's drought, perhaps the worst in over 1300 years (Dean and Robinson 1978). These low-elevation stands of ponderosa pine would have been subjected to greater drought stress than trees found at higher-elevation ecotones, and thus they were likely more susceptible to a severe bark beetle outbreak like the one that occurred. Periodic Dendroctonus outbreaks probably recur along the lower ponderosa pine

ecotone whenever old pine stands experience extreme droughts (c.f. Rykiel et al 1988).

MIXED CONIFER FORESTS OF THE SIERRA DE LOS VALLES

The mixed conifer forests of the Sierra de los Valles have undergone major changes in structure and species composition in the past century (cf. Allen 1984-a). On southerly aspects, as in much of the headwaters area of the Rito de los Frijoles, a mean fire interval of about 10 years prevailed before 1900. Widespread crown fires are unknown from these sites, although the presence of small aspen patches on steep slopes indicates that spotty crown fires did occur on relatively mesic sites (Figure 4-16). Ponderosa pine was a dominant to co-dominant component of these open forests, with Douglas-fir, white fire, aspen, and limber pine also common. Fire suppression has allowed the development of dense sapling understories in many mixed conifer forests, with the regeneration dominated by Douglas-fir and white fir. Figure 5-33 shows the size-class distribution of one such mixed conifer stand, located adjacent to sample point #471 and fire-scar Sample #9 at an elevation of 2721 m in the Frijoles Watershed. The low density (133 stems/ha), old-growth overstory is dominated by ponderosa pine, while the nearly impenetrable understory (12,466 stems/ha) is primarily composed of Douglas-fir, with some white fir. While few mixed conifer stands display such extreme, two-tiered stand structures or understory densities, the pattern of change observed at this site graphically illustrates the general trends which have occurred in local mixed conifer forests. Ten dated understory stems established between 1914 and 1934, with 80% established between 1926 and 1931. Both fire-scar

FIGURE 5-33. Mixed conifer forest size-class distribution near sample point #471, Cerro Grande slopes.



Sample #9 and another cored overstory tree germinated about 1780. The last fire recorded by fire-scar Sample #9 occurred in 1899, which was also the last major fire year for the whole Cerro Grande Accession sample area.

The absolute and relative increases in Douglas-fir and white fir abundance in local mixed conifer forests have apparently led to increasingly intense and synchronous spruce budworm outbreaks in the Jemez Mountains (Swetnam 1989). Similar forest changes and budworm histories are observed in the northern Rocky Mountains (Anderson et al 1987) and throughout the southern Rocky Mountains (Swetnam and Lynch-in review).

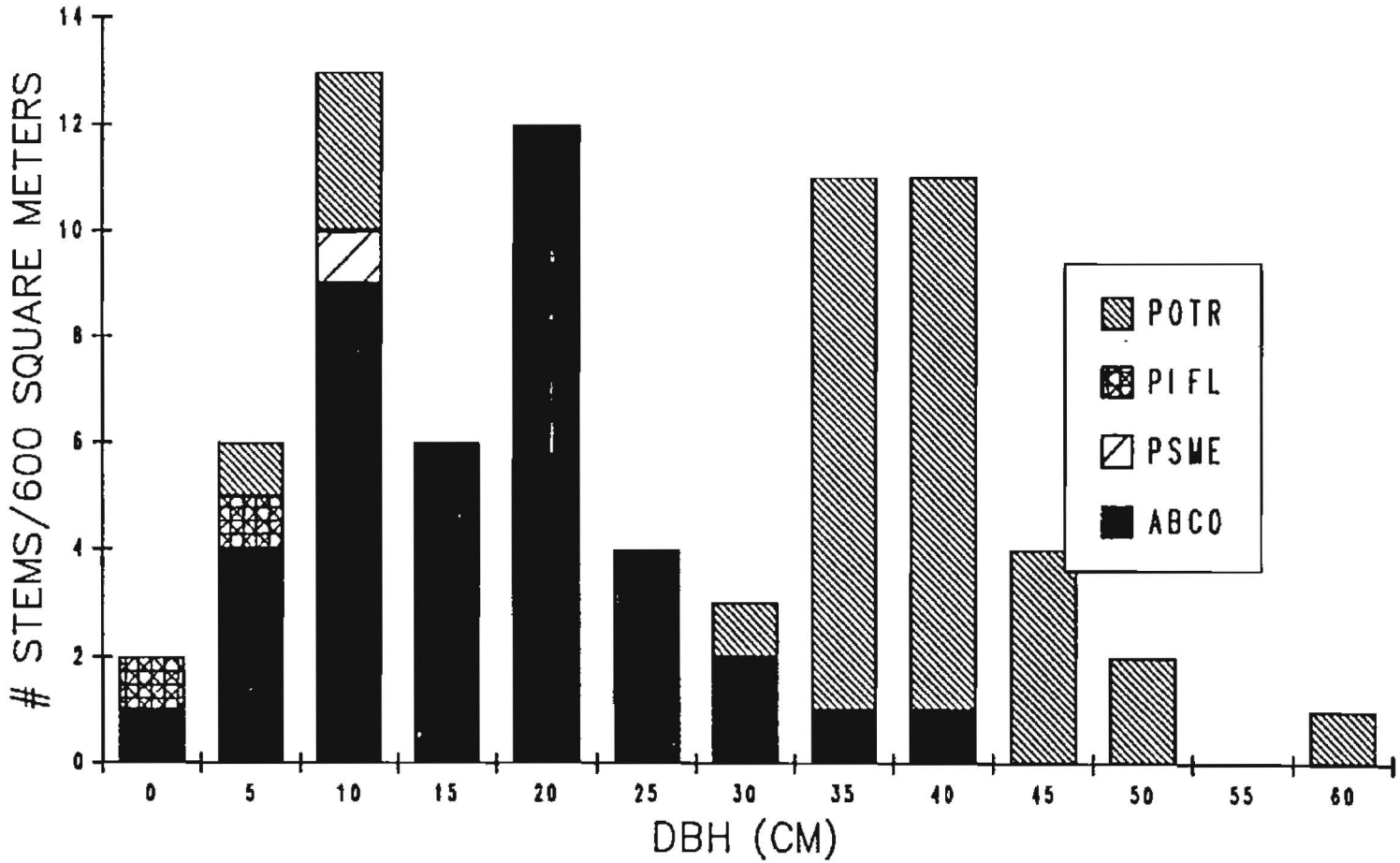
In 1984 I conducted an inventory of downed woody fuel loads across the mixed conifer forests of the Frijoles Canyon headwaters (Allen 1984-c). Unlogged mixed conifer forests on 4 relatively xeric sites displayed mean fuel loads of 21.4 tons/acre, while 4 relatively mesic mixed conifer sites displayed average fuel loads of 25.3 tons/acre. These values are near the mean "natural" fuel loading of 23.9 tons/acre found by Sackett (1979) for equivalent measurements in 16 southwestern mixed conifer stands. Sackett considered these values to be "heavy", with potential to support crown fire development where vertically continuous ladder fuels exist, especially in stands with a large component of ponderosa pine.

Similar changes in species composition, forest structure, and fuels have occurred due to fire suppression in other western mixed conifer forests (Dickman 1978, Parsons and DeBenedetti 1979, Ahlstrand 1980, McNeill and Zobel 1980, Dieterich 1983). The combination of high surface fuel loads and dense, laddered, vegetation structure indicates

that many Jemez Mountains mixed conifer forests are now susceptible to the development of crown fires if ignited during dry, windy conditions. Barring preventive management intervention, I believe that intense fires will inevitably occur within local mixed conifer forests in the coming decades .

On mesic sites (e.g. north-facing slopes) patchy crown fires were apparently one component of pre-1900 mixed conifer fire regimes, based upon the presence of variable-sized aspen stands interspersed within the matrix of mixed conifer forest. Most mixed conifer aspen ramets are now over 100 years old and subject to increased insect and disease problems as they decline in vigor. Modern fire suppression has prevented the regeneration of aspen stands. As current aspen overstories break up over the next twenty or so years conifer species which are now present in the aspen understories will establish control of these sites. Figure 5-34 displays such a typical situation, where white fir will become the dominant as the aspen ramets die. Until fire returns to these forests the aspen stands, so beloved by the public for their fall colors, will continue to decline. The Santa Fe National Forest recognizes this successional fact-of-life and is now regenerating aspen stands with small clearcuts. Fortunately, aspen clones are able to persist in a suppressed state in the understories of mixed conifer forests for extended periods of time. The high probability of intense fires in Jemez Mountains mixed conifer forests in the coming decades suggests that new aspen stands will develop again soon, although perhaps not under circumstances preferred by resource management agencies and the general public.

FIGURE 5-34. Aspen stand size-class distribution near sample point #716, Cerro Grande slopes.



MONTANE GRASSLANDS OF THE SIERRA DE LOS VALLES

Montane grasslands are found in a distinctive landscape pattern on the upper, south-facing slopes of nearly all of the larger summits and ridge crests in the Jemez Mountains. These montane grasslands have apparently existed as interruptions within mixed conifer forest life zones for millenia (Allen 1984-a), yet young ponderosa pine, Douglas-fir, and quaking aspen are common in and bordering these grasslands. This section reviews the tree invasion of this ancient landscape element in detail, updating the discussion in Allen (1984-a).

Conifer Invasion

Examination of the 1935 and 1981 air photos clearly reveals recent conifer tree invasion of Jemez montane grasslands. The GIS cover-type maps display an 85% decrease in open grassland area on Cerro Grande from 110 ha in 1935 to 17 ha in 1981 (Figure 5-35). Overall, in the southeast portion of the Jemez Mountains open montane grassland area decreased 55% from 554 ha in 1935 to 250 ha in 1981 (Tables 5-5, 5-6). Several small montane grasslands present in 1935 have disappeared, while the larger grasslands have been fragmented.

The age structure of the trees sampled along Transect VA (Figure 5-36) indicates that ponderosa pine invasion on Cerro Grande began in the early 1920's. Note the presence of two much older ponderosa pines; a number of such old ponderosa pines are visible in the 1935 air photos, scattered across this grassland. The age of the 1807 pine is underestimated, as my increment borer fell quite short of the center of this 1.01 m dbh tree. These "savanna relicts" apparently served as the seed source for the modern tree invasion. The lack of younger trees in

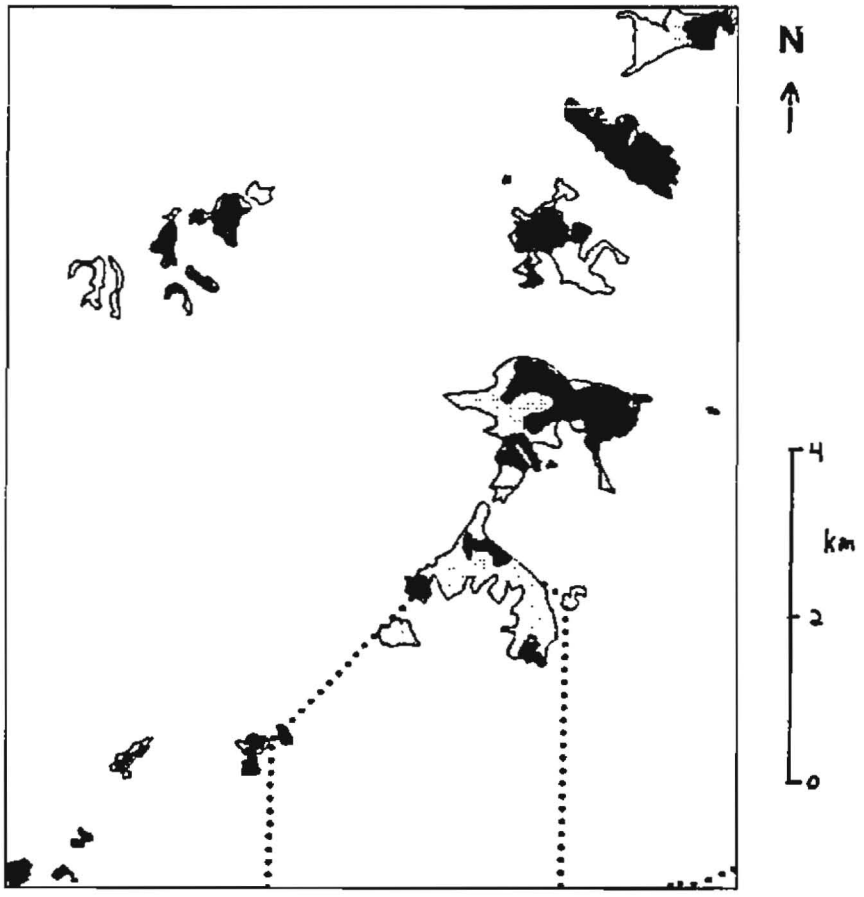


Figure 5-35. Map of reduction in montane grassland area between 1935 and 1981. Area of open grassland (with less than 10% tree canopy cover) is shown with a grey pattern for 1935 and as solid black patches for 1981. The dotted line is the current boundary of Bandelier National Monument, which reaches its northernmost extent at the summit of Cerro Grande.

FIGURE 5-36. Tree establishment dates, Transect VA, Cerro Grande, and reduction in sheep numbers.

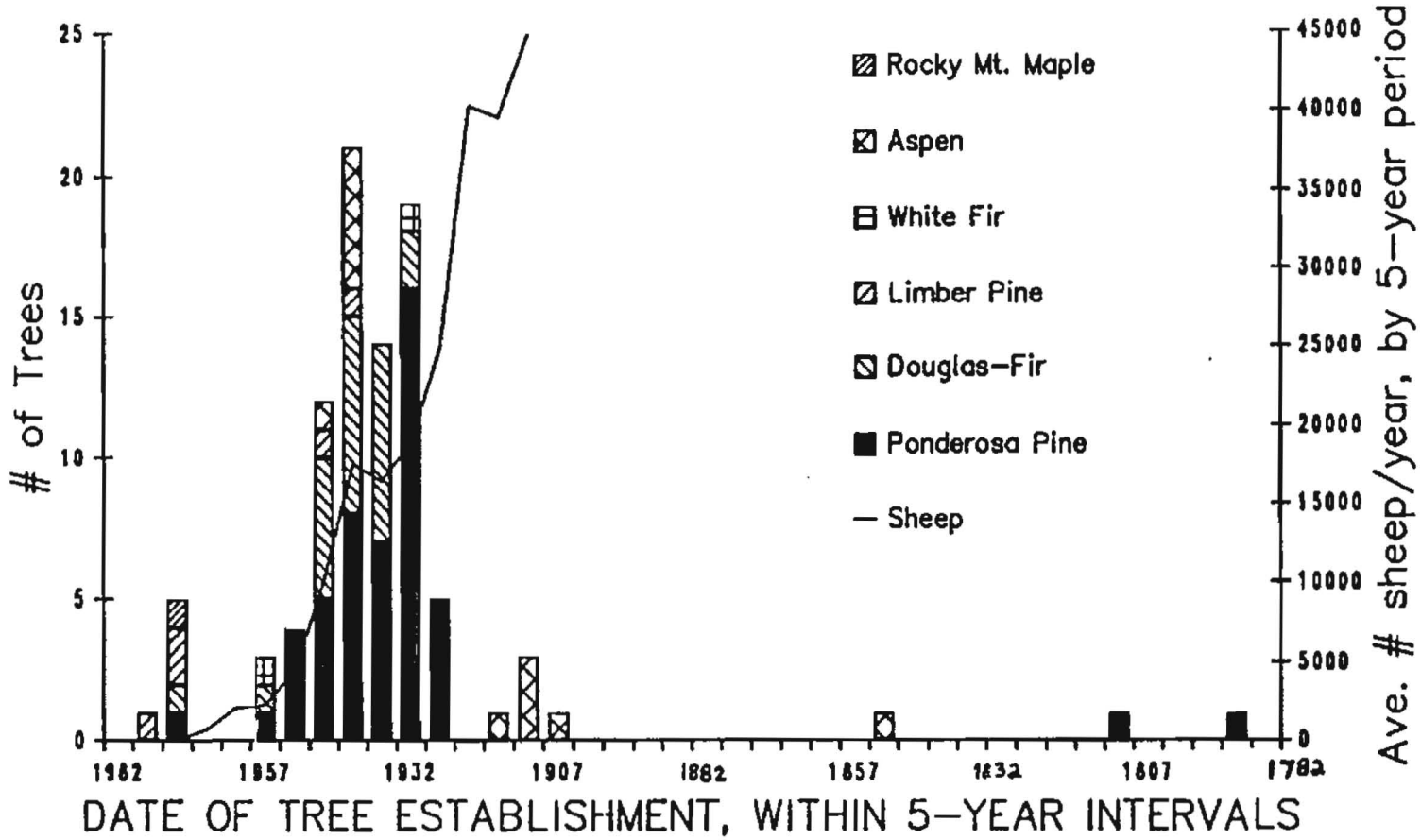


Figure 5-36 is an artifact of methodology, since only the largest two trees/species/quadrat were cored. Young ponderosa pine continue to invade the open grassland (VX and VB in Table 5-9) - even younger, tiny seedlings are observed elsewhere in this grassland. The only spruce growing in the open grassland are adjacent to the ridge-crest ecotone and are less than 60 years old.

The soils beneath montane grasslands are well-developed Mollisols, indicating the longterm presence of grassland vegetation on these sites (Allen 1984-a). Transect VA reveals no difference in mollic epipedon depth or color between sites invaded by trees ($x = 44.6$ cm, $n = 5$) and open grassland ($x = 44.8$ cm, $n = 4$) on Cerro Grande. Mollic epipedons tend to increase in depth with distance downslope, as expected (Montagne et al 1982) - on lower slopes below this transect mollic epipedon depths of 66 and 70 cm were observed. Shallow, lighter colored, ochric epipedons are observed beneath the old spruce forest on the north-facing slope.

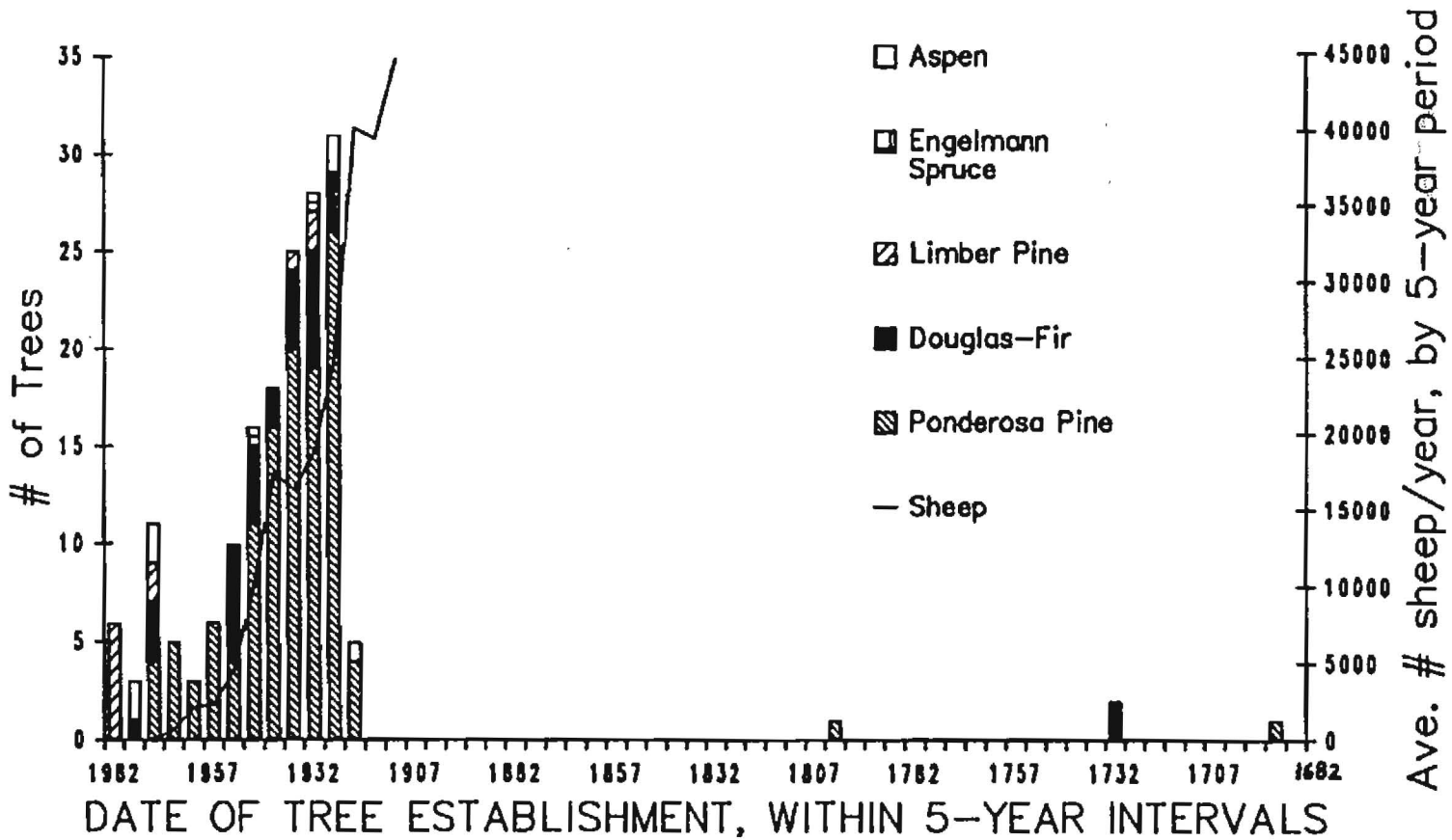
Conifer invasion of the Cañada Bonito grassland began in 1920, with ponderosa pine and Douglas-fir the primary invasive species (Figure 5-37). After an early peak in tree invasion the rate of tree establishment has declined continuously to a relatively low level. A few large, old ponderosa pine and Douglas-fir are present; the ages of the three oldest trees are underestimated due to the limited length of my increment borer.

In 1982 a U.S.Forest Service transect of 10 soil pits across the Cañada Bonito grassland revealed an average A horizon depth of 31 cm, with 7 sites classified as Pachic Cryoborolls and three sites as Pachic Paleborolls (W. Lucas and T.G., 1982, unpublished pedon descriptions on

TABLE 5-9. The number of invasive trees by age-class at 5 sites. Tree species abbreviations: PP = ponderosa pine, DF = Douglas-fir, ES = Engelmann spruce, QA = quaking aspen, WF = white fir, CF = corkbark fir. Transect VB and VX display ongoing ponderosa pine invasion on Cerro Grande, and non-invasive spruce forest on the north-facing slope (bold type). Transects IA, IIA and IIIC show aspen overstories with younger conifer understories developing since the 1920's. The 1967-1971 cohort is underlined - 22 of 26 trees in this cohort for Transect IA date to 1969.

		Transect													
		VB			VX		IA		IIA			IIIC			
	Age	Tree Species													
Year	Class	PP	DF	ES	PP	DF	QA	WF	QA	ES	WF	QA	ES	DF	CF
1982	0														
	1-5		1								1				
	6-10	5	1		8		1	7	3	1		3			
	-15	14	1		28			26	1			7			
	-20	4			7	1		2		1		1			
	-25			1	2		1				1			3	
	-30			2			1					4		1	
	-35							1		1		1			
	-40			1					1			1		1	
	-45	2							1				1		
1932	-50			2			1		2			2			
	-55						5		19	3					
	-60	1		1			2	2	12						
	-65						2		1						
	-70														
	-75								1						
	-80								1						
	-85			3											
	-90								1						
	-95						2								
1882	-100						1				1				
	-105			1											
	-110			1							1				
	-115			1					1		2				
	-120										2				
	-125			1							2				
	-130			3											
	-135			1											
	-140			2											
	-145														
1832	-150														
	-155		1	1											
1872	-210			1											

FIGURE 5–37. Tree establishment dates, Transect IB, Canada Bonito, and reduction in sheep numbers.



file at the Santa Fe National Forest). The modifying term "pachic" emphasizes the thick mollic epipedons of these soils, and the great group formative element "pale-" refers to the old development of these soils (Soil Survey Staff, 1975). Entic Cryandepts and Dystric Chryochrepts with shallow, lighter colored, ochric epipedons underlie the dense mixed conifer forests on adjacent north-facing slopes (Nyhan et al 1978).

Similar conifer tree invasion and soil patterns are observed in the 9 other Jemez Mountains grasslands where data were collected. Ponderosa pine and Douglas-fir are the primary conifer invaders of open grassland in the Jemez Mountains. Tree invasion has been greater where nearby seed sources of these relatively drought-tolerant species existed, usually in the form of scattered old savanna relicts on the fringes of the former grassland. The onset date of 20th Century conifer invasion varies between grasslands, ranging between 1920 and 1932; conifer establishment in the understories of aspen stands is also not observed before 1920 (IA, IIA and IIIC in Table 5-9). This initial period of massive tree invasion coincides with the precipitous drop in sheep and goat numbers on the Santa Fe National Forest in the decade after World War I (Figures 5-36, 5-37).

Aspen in Montane Grasslands

Aspen groves are found in or bordering every montane grassland I examined in the Jemez Mountains. Many of these aspen groves consist of clones of large extent, covering several hectares. The individual trees within these clones tend to be of uniform size, even to the edge of the stand, with an abrupt ecotone between the grove and the

adjoining grassland. The characteristic age structure of these stands is illustrated by Transect IIA (Table 5-9). This Chicoma Ridge site is dominated by a cohort of ramets that sprouted between 1924 and 1931. An older cohort of decadent and dying individuals is present in the central portions of the grove, 80 to 120 years old, presumably representing the progenitors of the present stand. Canopy gaps, due to the death of such older trees, are filled with younger trees. Otherwise the youngest trees represent a fringe of invasive root suckers in the grassland along the clones's edge. These sucker shoots display much damage from deer and elk browsing, are generally unhealthy, and many are dead. A few Engelmann spruce are present in the understory of this aspen stand. A thick Mollisol is found beneath this aspen grove and the adjacent grassland (epipedon depths of 55 cm and 77 cm).

A similar pattern is revealed by Transect IA (Table 5-9). This aspen grove displays a sharp boundary with the Cañada Bonito grassland. It is dominated by trees established between 1923 and 1929. A few older trees persist within the core of the stand from an earlier generation of clone ramets. Numerous dead, damaged, and unhealthy small aspen shoots were present in and especially adjacent to this stand which were not aged - most were less than 5 years old. White fir increasingly dominates the understory of this aspen stand. Mollic epipedon depths range from 18 cm to 40 cm along this transect.

Block IB intersects the corner of an even-sized aspen stand (Figure 5-37). The usual abrupt aspen/grassland boundary with a margin of unhealthy, small sucker shoots (not aged) is present. Sampled canopy dominants sprouted in 1920, 1923, and 1924. This aspen stand

grows in the deep Mollisols that blanket the Cañada Bonito grassland site.

The aspen clone of Transect IIIC on Polvadera Peak is old (Table 5-9), with only sporadic sucker shoots. Conifers dominate the understory. Mollic epipedon depth is 35 cm.

On Cerro Grande, Transect VA (Figure 5-36) shows three pulses of aspen regeneration within the single clone at about 1847, 1908, and 1941. A widespread fire occurred in the Cerro Grande Accession in 1847 (Figure 4-20). The 1847 ramets are visually apparent as large, mostly dead and down stems; most have rotten centers and are undatable. Fire scar samples were collected from 2 of these old aspen, but they were undatable (Caprio et al 1988). The 1906 to 1909 cohort dominates this stand. A 15m wide strip along the upslope margin of this grove was established between 1938 and 1943. Overall the clone boundaries have remained quite stable since 1935. A thick Mollisol (50 cm epipedon) underlies this stand.

Conifer Invasion: Climate Interactions

Climate change has been proposed to trigger tree invasion of mountain meadows, especially in subalpine zones near the upper limits of tree growth (Franklin et al 1971, Agee and Smith 1984) or in wet meadows where tree establishment may be limited by cold or excessive moisture (Wood 1975). Yet many studies of tree invasion in western North America have not found obvious climatic correlations (Dunwiddie 1977, Strang and Parminter 1980, Vale 1981-b).

Jemez montane grasslands do not appear to be climatically marginal sites for tree growth - invasive trees grow rapidly once established.

These grasslands interrupt productive montane forests and are found well below treeline. While climate data do not exist for these sites, this elevation zone is estimated to receive 90 cm of precipitation annually (Nyhan et al 1978). Some grassland forbs indicate a plentiful supply of soil moisture - e.g. Iris missouriensis is abundant and widespread in these grasslands. Weaver (1979, 1980) demonstrates that fescue grasslands from similar environments in the West are in coniferous forest mesoclimates, with frost-free periods, July temperatures, annual precipitation, and drought periods like those of the adjacent Douglas-fir and subalpine fir forest climates he examined. Weaver (1979) concludes that: "Environmental factors other than temperature and precipitation -perhaps wind, snow cover, soil characteristics, or fire - must allow fescue grasslands to dominate the sites they do". This conclusion appears valid for Jemez montane grasslands as well.

However, the warm, dry microclimate and direct sun exposure of the steep, south-facing slopes of these grasslands probably inhibits tree establishment by limiting germination and seedling survival, especially for such species as Engelmann spruce and white fir (Stahelin 1943, Alexander 1987). Wind-mediated snow transfer off these open slopes (Daubenmire 1981), sublimation, and suitable conditions for winter drying/sun scorch (Miller 1979) likely contribute to relatively xeric late winter and spring site conditions. Even at elevations up to 3300 m the primary invasive conifer species are ponderosa pine and Douglas-fir, both relatively tolerant of drought and direct sun. Seedlings of these two species may also compete better with grasses than the other available conifer species. The comparatively extreme invasion of Cerro

Grande (Figure 5-35) may reflect its atypically widespread and dense distribution of ponderosa pine savanna relicts, which served as seed sources. Yet even in the ameliorated microclimate of aspen stand understories conifer reproduction did not occur prior to 1920 (IA, IIA and IIIC in Table 5-9).

It might be hypothesized that a climate change to moister conditions promoted tree invasion. Annual and spring precipitation were unusually high from 1919 to 1921 at nearby Santa Fe (Rose et al 1981), coincident with the start of tree invasion. Such wetter conditions apparently contributed to the well-known 1919 establishment of dense ponderosa pine stands in the Southwest and likely helped successful conifer establishment in some Jemez montane grasslands, although similar wet periods have occurred before and since then (Dean and Robinson 1977, Rose et al 1981) without accompanying bursts of tree invasion (cf. Cooper 1960). Comparison of 20th Century weather records for northern New Mexico (Tuan et al 1973, Rose et al 1981, Bandelier National Monument - weather records on file, Los Alamos National Laboratory - weather records on file) with the pattern of tree invasion suggests that dry years from the 1950's through mid-1960's may have reduced tree invasion rates, and that the wet spring of 1969 (May/June precipitation was 311% of the mean at Bandelier) may have helped a mini-pulse of trees to invade some grasslands (Figures 5-36, 5-37, Table 5-9).

Overall, the evidence from the Jemez Mountains offers only weak support for the idea that climatic variability is responsible for the observed tree invasion. Local tree-rings record numerous fluctuations between wet and dry periods extending back to 598 A.D. (Dean and

Robinson 1977) with 22 periods with 3 or more consecutive years of above average tree growth (probable wet periods) between 1600 and 1920 (Dean and Robinson 1978), and 5 such periods of above average precipitation observed since 1920 (Figure 4-6, LANL - records on file) - yet absolutely no conifer establishment is seen from the 19th century, and only occasional savanna relicts persist from earlier centuries. The striking onset of conifer encroachment occurred at different times between 1920 and 1932 for different grasslands. Further, tree invasion has generally continued to the present, although commonly at reduced rates over the past 40 years (Figure 5-37). For climate to be the primary causal factor for tree invasion would require a 20th Century climate change from the conditions which prevailed over at least the previous several hundred years to occur in different years on different mountains within the Jemez range. In addition, these altered climatic conditions must have persisted until today. No such distinct and long-lasting climatic change is known to have occurred here (but see Neilson [1986] for a southern New Mexico interpretation). Pollen cores from Alamo Bog, located in the Jemez Mountains, indicate a "fairly stable" climate over the 4600 year period of record (Stearns 1981). Dendroclimatic reconstructions do not show any apparent modern climate anomalies which would explain the observed tree invasion pattern (Fritts 1965, Dean and Robinson 1977, Rose et al 1981). A convergence of adequate supplies of irregularly produced seed (Larson and Schubert 1970) and moist spring conditions is likely a necessary, but not sufficient, condition for pulses of conifer invasion. Moist spring conditions are tied to cyclic El Niño events (Andrade and Sellers 1988) that have recurred for at least hundreds of years (Quinn

et al 1987, Swetnam - in press). Over time spans of hundreds or thousands of years it is unlikely that climatic conditions have precluded forest establishment on montane grassland sites.

Conifer Invasion: Fire Interactions

Perhaps modern fire suppression allowed the recent tree invasion. The majority of grasslands worldwide are subject to encroachment by woody plants, with fire acting to maintain the herbaceous status quo (Sauer 1950, Vogl 1974, Kucera 1981). Reductions in fire frequency are often described as a major cause of tree invasion of meadows (Vankat and Major 1978, Strang and Parminter 1980, Vale 1981-b, Arno and Gruell 1986, Sugihara and Reed 1987). Local fire histories reveal a nearly complete cessation of a high-frequency ground fire regime in the late 1800's adjacent to, and presumably in, montane grasslands (Figure 4-20).

The topographic situations of montane grasslands are consistent with the idea that fires were important to their longterm maintenance. Fires would have spread freely across the smooth terrain of grassland sites; where interrupted by ravines or other potential fire breaks, old (pre-1920) stands of trees are often observed. Fires would have spread easily up the south-facing grassland slopes to the ridge crests, but would have had difficulty backing down into the cool, moist understories of spruce-fir forests on north-facing slopes; thus the observed development of the abrupt, ridge-crest, grassland/forest ecotones. Prescribed fires conducted on Cerro Grande in 1985 and 1987 displayed this hypothesized fire behavior.

The local cessation of widespread surface fires preceded the onset

of massive conifer invasion into grasslands. The absence of significant conifer encroachment prior to 1900 indicates that high-frequency fires helped maintain these grasslands by killing invasive conifer seedlings. The few, older, fire-singed savanna relicts may have become established during occasional longer intervals between fires (e.g., 1773-1796). Yet recent fire suppression alone does not explain the timing of the initial pulse of tree invasion, which occurred several decades after widespread fires ceased.

Conifer Invasion: Livestock Grazing Interactions

The dense grass cover of ungrazed montane grasslands almost certainly inhibits tree establishment (cf. Pearson 1942, Larson and Schubert 1969, Madany and West 1983). The immense numbers of livestock grazed locally in the late 19th and early 20th centuries must have seriously degraded the herbaceous vegetation of these montane grassland sites. Even under today's relatively low stocking levels, grazed grasslands display greatly altered species composition, reduced ground cover and grass vigor, and accelerated erosion rates (Allen 1984).

I propose that the changeover from high levels of sheep to lower levels of cattle stocking served as the specific trigger for the massive pulse of tree invasion into grasslands in the 1920's and 1930's. Dramatic decreases in sheep numbers roughly coincide with the abrupt onset of conifer tree invasion (Figures 5-36, 5-37). Severe losses of sheep occurred in the winter of 1918-1919 on the Baca Location, which then included Cerro Grande (Scurlock 1981). Sheep grazing has been thought to prevent tree invasion by browsing and trampling, with the removal of sheep serving as a trigger for tree

invasion (Ellison 1960, Vankat and Major 1978, Vale 1981-a). Moderate levels of cattle stocking may promote tree invasion (Rummel 1951, Dunwiddie 1977, Strang and Parminter 1980). The absolute lack of observed conifer invasion of Jemez grasslands in the late 19th and early 20th Centuries may reflect intense grazing pressure by large sheep populations (cf. Cooper 1960). Captioned photos taken nearby in 1927 clearly show and state that exclosure protection from "goats" and sheep allowed dense ponderosa pine reproduction that was absent from outside the exclosure (Washington negative #'s 36-30-86 and 36-30-88, on file at the Santa Fe National Forest Supervisor's Office). The massive pulse of tree encroachment observed in the early invasion period (Figure 5-37) probably reflects the suitability of degraded pastures for tree establishment immediately after the reduction of sheep numbers, when the vigor of the herbaceous plant cover was lowered and more bare mineral soil was likely exposed. This would explain the somewhat variable date of initial tree influx, as sheep grazing undoubtedly faltered at different times on various grasslands. Site-specific vagaries of local tree seed crops and spring moisture conditions may also have contributed to the variability in tree establishment dates between grasslands.

Trees continue to invade these grasslands in the presence of cattle. Ongoing conifer encroachment is often at lower rates, despite the increase in seed sources from the maturing first wave of invasion, likely due to recovery of grassland sod cover, litter, and overall vigor with lower livestock pressure. Cañada Bonito and several adjacent grasslands display the lowest rates of tree invasion, perhaps largely due to their 46-year reprieve from all livestock grazing.

Ongoing tree invasion may depend upon pocket gopher-disturbed microsites for seedling establishment in such ungrazed grasslands.

In summary, the climatic conditions of montane grasslands appear suitable for conifer establishment and growth, but high fire frequencies and vigorous grass competition maintained largely treeless grasslands prior to 1900. Overgrazing by many sheep from the 1880's to about 1920 caused failure of the previous ground fire regime, yet precluded all conifer establishment by direct browsing. Subsequent reductions in sheep numbers apparently allowed conifers to establish in degraded grasslands where appropriate seed sources existed. Moist spring weather may have promoted pulses of tree establishment.

Aspen "Invasion"

The even-aged nature of aspen groves in Jemez grasslands indicates clonal sprout regeneration (Barnes 1966), likely stimulated by fire damage to aboveground stems (Schier et al 1985, Jones and DeByle 1985-b). The large number and rapid growth of root sprouts sent up by regenerating aspen clones apparently allowed them to establish stands on many grassland sites in the late 19th Century, probably in response to fires in the pre-suppression period. Old individuals and snags sometimes display basal fire scars. The current sharp grove boundaries are probably maintained by an absence of extensive sucker development due to fire suppression, combined with intensive browsing by elk, deer, and domestic livestock (Frederickson 1975, DeByle 1985) and numerous other causes of mortality (Hinds and Shepperd 1987) to the few sprouts which are produced.

Many aspen stems date from the 1920's (Figure 5-37), perhaps

reflecting a release from intensive sheep browsing. However, it is difficult to determine when and how aspen clones initially became established in Jemez montane grasslands. It is not currently possible to date accurately the origin of an aspen clone, and such clones "may live almost indefinitely" (Barnes 1966). Modern aspen clones have been assigned origin dates from the Pleistocene (Blake 1964, Schier 1985) and even the Pliocene (Barnes 1975). Tree ring counts show that many Jemez aspen groves are well over 100 years old, and the large clone size of many of these stands strongly suggests an ancient date of origin (Kemperman and Barnes 1976). On the other hand, the existence beneath many of these aspen groves of Mollisols that are similar to the soils of surrounding grasslands implies either a recent aspen invasion or the presence of a vigorous grass cover in the aspen understory during soil formation (Jones and DeByle 1985-c).

Aspen clones may have been opportunistic invaders of old montane grasslands over centuries or millennia, encroaching periodically when conditions were favorable. In this scenario the old grassland soils simply persist beneath the aspen groves. While this process has likely occurred to some extent, I generally favor an alternative explanation similar to that proposed by Maini (1960) for the establishment of aspen parklands in the northern Great Plains. Many aspen clones observed today may not be invasive in nature, but rather the remnants of early Holocene forests that predate the origin of the grasslands. The shift after 8000 B.P. to the modern southwestern climatic regime of spring drought followed by midsummer thunderstorms (VanDevender and Spaulding 1979) may have sufficiently increased fire frequencies and/or intensities to have promoted the development of montane grasslands at

the expense of the former forest. Perhaps episodes of warmer, drier conditions during the variable "Altithermal" interval of about 7000 to 5000 B.P. (Petersen 1982, Hevly 1985) led to severe crown fires; numerous charcoal deposits and associated decreases in total tree pollen influx are observed even after 4600 B.P. from the local Alamo bog site (Stearns 1981). While conifers were eliminated from these relatively xeric and fire-prone sites, the irregularly distributed aspen stands persisted as brushy patches in the grasslands. Similar Mollisols then formed beneath the open, shrubby aspen stands and the surrounding grasslands. Extended fire return intervals or decreased fire intensities, due to historic grazing practices and modern fire suppression, then allowed these aspen clones to develop into the mature groves of trees found today. The absence of pre-1920 conifers in these aspen stands supports this idea, since if the aspen stands had contained mature trees for centuries one would expect to find some old individuals of mixed conifer species present that had established through the years in these favorable sites. Many "climax" aspen stands elsewhere may have a similar origin (e.g. Johnston and Hendzel 1985, Mueggler 1985).

A similar history of long persistence as brush fields may hold for the smaller clones of Gambel oak (Quercus gambelii) found less frequently in some Jemez montane grasslands, as on Cerro Grande.

Age of Montane Grasslands

Local opinion regarding the origins of these grasslands has ranged from a nebulous volcanic cause to recent post-fire derivation. Interest in this question is not new:

> Many folks have mentioned the bare spots on Caballo Mt. and other peaks to the west and wondered just how and when they got there. I was advised by J.L. Wang of the Forest Service that they are the result of huge forest fires which cccurred between 1850 and 1875. In this country, due to rapid evaporation, low humidity, and considerable wind, the burned-over areas just dried up and disappeared, leaving the open grassland... Incidently the next time you drive over into the Valle Grande look north along the range just after you get over the top. Every peak in the line, five of them, is barren on the southern shoulder right to the summit (Hawkins 1946).

Several lines of evidence indicate that montane grasslands are ancient features of the Jemez Mountains landscape. First, consider the consistent and distinctive landscape pattern of grasslands and pachic Mollisols on south-facing slopes, with old forests and Inceptisols or Alfisols on adjacent north-facing slopes (c.f. Ugolini and Schlichte 1973). Mollisols are typically considered to have formed under grassland vegetation (Buol et al 1980), although they may also form where trees and grasses share dominance (Bakeman and Nimlos 1985). The great thicknesses of montane grassland Mollisol A-horizons indicate that these soil-vegetation systems have likely occupied their present sites for thousands of years (Scharpenseel 1971, Buol et al 1980:168, Allen 1984).

Forest "artifacts" are absent in Jemez montane grasslands. Widespread tree invasion is apparently a 20th Century phenomenon. There are no tree stumps or tree-tip mounds, and woody charcoal is not common in grassland soils. The few old trees that exist indicate that at most a savanna-like density of trees has been present in grassland margins over the last 300+ years.

The names of such grassland-capped mountains as Cerro Pelon (Bald Peak) and Cerro Pelado (Baldy Peak) further suggest that these

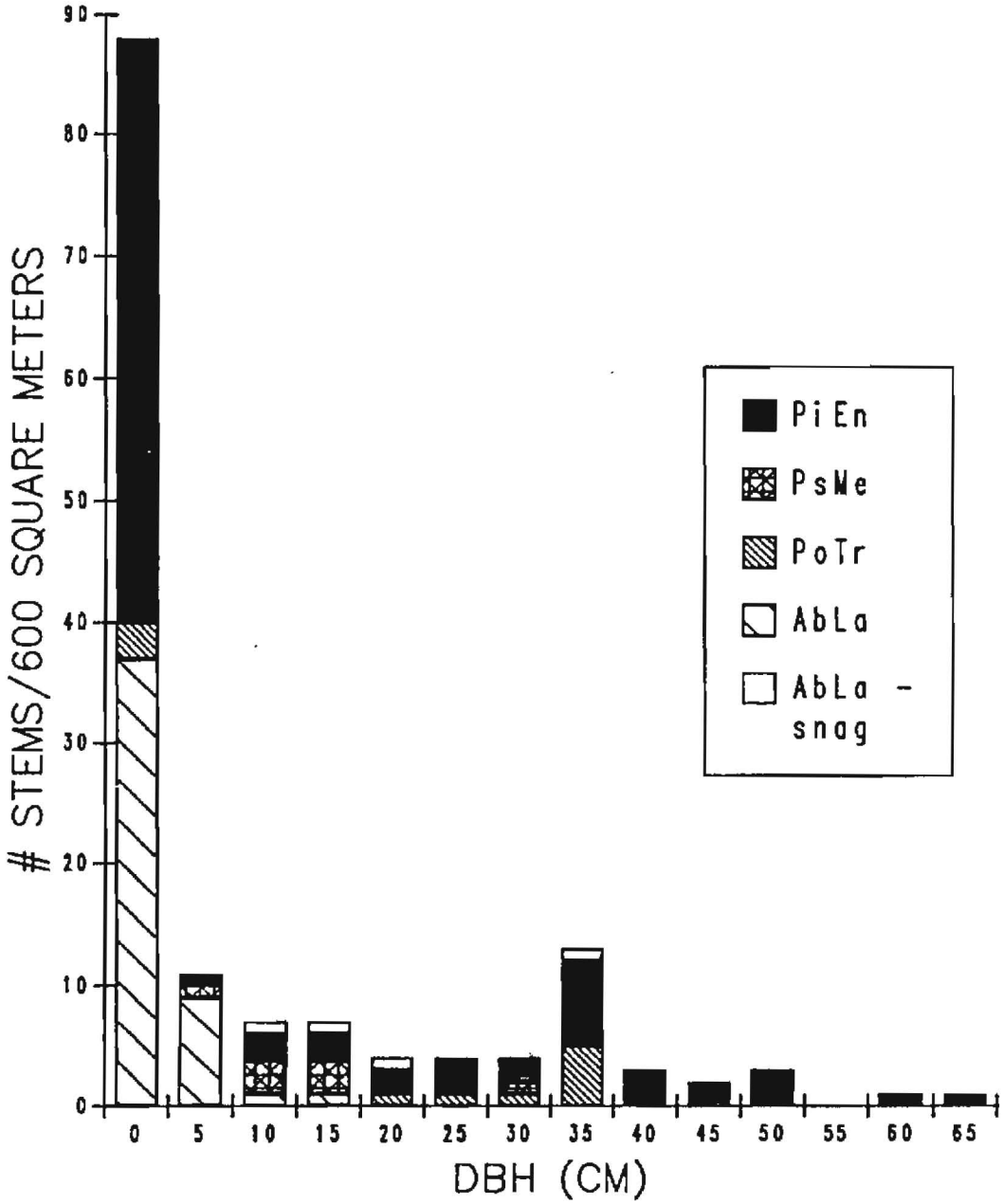
grasslands have been present since Spanish settlers arrived locally nearly 400 years ago.

I believe that Jemez montane grasslands were maintained by recurrent fires, their warm and dry microclimate, the paucity of drought-tolerant tree seed sources at these elevations, the difficulty of tree seedling establishment in the dense sod and thick litter of the grasslands, competitive exclusion of tree seedlings by vigorous grasses, and animal damage to young trees (Allen 1984). Despite higher-frequency climatic regimes (Neilson 1986), the relative continuity of the modern southwestern climatic regime over the last 4000 to 8000 years (VanDevender and Spaulding 1979) provides no reason to suppose that forests would have been able to establish and maintain themselves on these modern grassland sites for any great length of time during this period. Fire has apparently been locally common throughout this time period, as 21% of the Alamo Bog core is composed of charcoal lenses dating back to 4600 B.P. (Stearns 1981), and fire scars reveal high-frequency fires back to 1480 A.D. Precise grassland forest ecotones were undoubtedly dynamic to some extent, and savanna or forest vegetation may have developed at times through the millenia. Still, all available evidence indicates that Jemez montane grasslands have occupied their present sites for many hundreds, and most likely thousands, of years, and thus are ancient features of this landscape. Ongoing tree invasion is converting these sites to forest vegetation.

SPRUCE/FIR FORESTS OF THE SIERRA DE LOS VALLES

Figure 5-38 is representative of the spruce/fir forest structure found on the highest, north-facing slopes of Cerro Grande. Engelmann

FIGURE 5-38. Engelmann spruce forest size-class distribution near sample point #734, Cerro Grande slopes.



spruce dominates the canopy, with aspen from the last major disturbance event persisting in the overstory. All larger corkbark fir stems are dead from an uncertain cause, but vigorous reproduction of this shade-tolerant species foreshadows increased prominence in this forest.

Prior to 1900 the Engelmann spruce/corkbark fir forests of the Jemez Mountains likely experienced high intensity fires at mean intervals of over 150 years. Thus these forests have probably been less affected by modern fire suppression than any other local landscape element. The greatest human impacts on local spruce/fir forests have resulted from clearcut logging activities on the Baca Location and especially around Chicoma Peak on the SFNF, where windthrow, spruce beetle infestation, and regeneration problems have occurred.

MEADOWS OF THE VALLES CALDERA AND SIERRA DE LOS VALLES

Low-lying meadows throughout the the Jemez Mountains have undergone radical transformations in herbaceous species composition, with alien species now commonly found as dominants. The extensive meadows of the Valles Caldera have been little-studied but the long history of livestock grazing in the Baca Location has likely led to analagous changes in these meadows (cf. Pilz et al 1979). Tree encroachment along meadow edges, probably related to fire suppression and grazing histories, has slowly reduced the area of open meadow in the Jemez Mountains. The boundaries of the Valle Grande meadow have been visibly altered by marginal tree invasion from upslope forests between 1935 and 1981 (Figures 5-9, 5-10), the primary cause of the 7% decline in mapped meadow area (Tables 5-5, 5-6). Frost damage associated with cold air drainage, fine-textured soils, water-logged

soils, absence of needed mycorrhizal symbionts, and distance from tree seed sources are speculative possible explanations for the lower rates of tree invasion observed in these meadows relative to higher-elevation montane grasslands.

THE LOCAL LANDSCAPE OF BANDELIER NATIONAL MONUMENT

Like all reserves, Bandelier National Monument is part of a local landscape. This section focuses on that portion of the Jemez Mountains landscape in and immediately around Bandelier, discusses the boundary interactions between the park and its surrounding landscape, and briefly considers landscape changes at this scale.

ECOSYSTEM PATCH (ECOPATCH) MAP

Figure 5-39 displays the outline of the 1981 high resolution map of ecosystem patches (cover-type/landform combinations) in and adjacent to BNM. Figure 5-40 displays combinations of these patch types in a color map, while Table 5-10 summarizes the cover-type information contained in this map. I will refer to this detailed cover-type/landform map as an "ecopatch" map to distinguish it from the similar, but lower resolution, 1981 landscape cover-type map previously discussed (see Figure 3-1).

The ecopatch map area covers the entire watershed area of BNM, with the exceptions of the outlying Tsankawi Unit and a portion of the Sanchez Canyon drainage in the southwest (Figure 5-39). Quadrangle boundaries were inadvertently imbedded into this map, resulting in the same flawed values for polygon perimeters and perimeter-derived indexes that are discussed in detail in **LANDSCAPE COVER-TYPES**. Still,

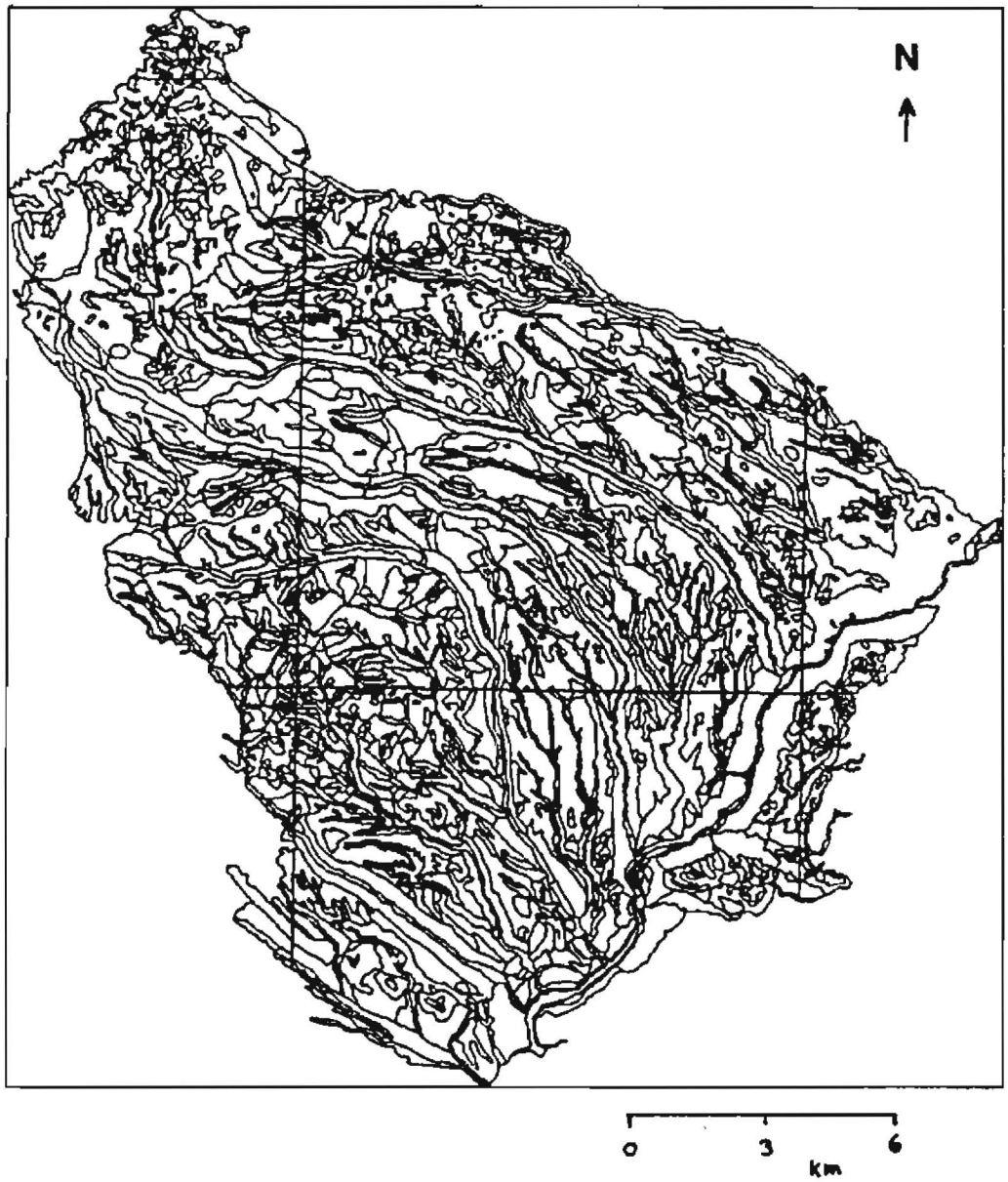


Figure 5-39. Outline map of 1981 ecosystem patches.

Figure 5-40. Color map of 1981 ecosystem patch cover-types. Mixed conifer forests = light blue, montane grasslands/meadows/grasslands = magenta, aspen stands (in the northwest quarter) = yellow, ponderosa pine forests = white outlined with black, grass/shrublands and shrublands are grey patterns, industrial areas of LANL and orchards (southwest corner) = orange, piñon-juniper woodlands = green, juniper woodlands/savannas (southeastern half) = yellow, canyon walls/felsenmeers = black, and water = dark blue.

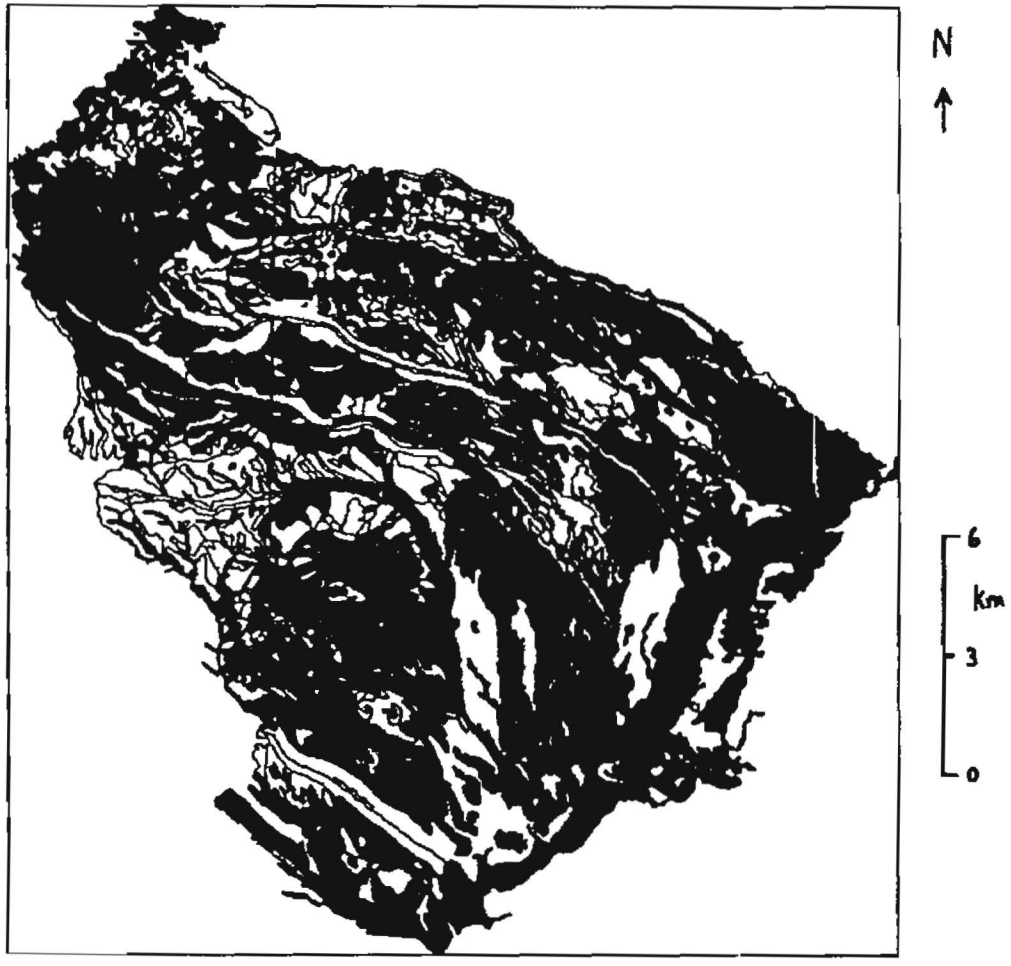


Figure 5-40.

TABLE 5-10. Summary cover-type information for the 1981 ecosystem patch (ecopatch) map. Cover-type abbreviations are described in Tables 3-2 and 3-3. N is the corrected number of polygons, % area is the percentage of the total map area, mean area is the corrected mean polygon area (ha), % P is the uncorrected percentage of the total map perimeter, mean P is the uncorrected mean polygon perimeter (km), mean P:A is the mean perimeter:area ratio (km/km^2), and mean PDI is the mean patch dissection index (unitless). Total map area is 28,683.7 ha, uncorrected total map perimeter is 4333.1 km, and corrected total map perimeter is 4239.1 km. T (U) provides totals using corrected total area but uncorrected perimeter data, while T (C) uses corrected area and perimeter data.

Cover Type	N	% Area	Mean Area	% P	Mean P	Mean P:A	Mean PDI
ARCH	8	0.04	1.3	0.10	0.5	39.9	1.30
COMM	1	0.01	4.2	0.03	1.2	28.8	1.67
CWCX	88	13.36	43.6	9.95	4.9	11.3	2.09
DROW	12	0.26	6.2	0.55	2.0	32.1	2.25
FELS	40	0.07	0.5	0.32	0.4	66.0	1.36
GRAS	48	2.35	14.0	1.87	1.7	12.1	1.27
GRSH	47	4.80	29.3	3.48	3.2	11.0	1.68
IND	20	0.85	12.2	0.65	1.4	11.7	1.15
J	189	10.21	15.5	10.58	2.4	15.7	1.74
J-S	10	0.56	16.0	0.66	2.8	17.8	2.00
JSAV	30	0.68	6.5	1.10	1.6	24.6	1.76
MC	135	11.58	24.6	9.51	3.1	12.4	1.73
MS-S	1	0.06	17.9	0.09	3.9	21.7	2.59
MEAD	27	0.21	2.3	0.51	0.8	36.4	1.54
MG	14	0.45	9.2	0.46	1.4	15.4	1.32
MINE	1	~0	0.7	0.01	0.4	51.4	1.23
ORCH	3	0.08	7.4	0.10	1.5	20.5	1.57
PJ	320	20.43	18.3	19.40	2.6	14.4	1.73
PJ-S	10	0.36	10.4	0.54	2.3	22.4	2.04
PMCS	5	0.43	24.4	0.44	3.8	15.8	2.19
POND	2	~0	0.6	0.01	0.3	53.5	1.15
PP	183	7.60	11.9	8.99	2.1	17.9	1.74
PP-S	14	1.51	30.9	1.25	3.9	12.6	1.97
PPJS	10	0.36	24.5	1.20	4.0	16.3	2.28
PPMC	113	7.12	18.1	7.46	2.9	15.8	1.90
PPPJ	197	11.15	16.2	13.06	2.9	17.7	2.01
REMC	1	0.37	106.8	0.85	36.8	34.4	10.04
REPP	3	0.49	46.9	0.68	9.8	21.0	4.05
RESI	2	0.02	3.2	0.04	0.8	25.9	1.31
RI-E	2	0.08	11.2	0.15	3.3	29.9	2.81
RI-M	9	0.39	12.4	0.69	3.3	27.0	2.68
ROCK	3	0.04	3.5	0.07	1.0	27.1	1.43
SAND	7	0.04	1.8	0.11	1.7	94.3	3.50
SH-O	42	0.80	5.5	1.27	1.3	24.0	1.58
SHRU	17	0.59	10.0	1.04	2.7	26.8	2.39
STRE	1	0.56	159.9	0.91	39.6	24.8	8.83
ASPE	57	1.32	6.7	1.79	1.4	20.3	1.48
WRSW	4	0.02	1.2	0.05	0.6	47.1	1.46
T (U)	1679	100.0	17.08	100.0	2.58	15.1	1.76
T (C)	1679	100.0	17.08	100.0	2.52	14.8	1.72

inclusion of these extraneous quadrangle boundaries only causes changes of about 2% in total map perimeter, average polygon perimeter, mean map P:A ratios, and mean map PDI. This indicates that the perimeter-associated data for individual cover-types reported here (Table 5-10) are close to their true values.

It is interesting to compare this ecopatch map to the coarser-grained, 1981 landscape cover-type map (Figures 5-8, 5-10, Table 5-5), as both are centered on the same area and were interpreted from identical aerial photographs. Mean polygon area is five times smaller for the ecopatch map, reflecting the smaller grain size and higher resolution of this map. This increased resolution is largely due to more-detailed interpretation of forest and woodland map units, which decreased markedly in size. The few cultural cover-types present within the ecopatch map boundary confirm the coarser-grained finding that cultural polygons have relatively simple, compact shapes, based upon low PDI values. High PDI values are found in both maps for the elongate patches associated with riparian forests, streams, canyon walls, and the various shrub cover-types that are typically found on canyon walls. The nearly identical mean PDI values for these two landscapes indicates that the patches mapped in the high-resolution ecopatch map maintained similar patterns of shape as those found in the landscape cover-type map.

The relatively doubled mean P:A ratio of the ecopatch map reflects its smaller average polygon size and does not imply that edge effects have necessarily increased in this landscape simply by mapping cover-types in greater detail, as different phenomena respond in diverse ways to different types of edge (cf. Lovejoy et al 1984). Thus assessment

TABLE 5-11. Landscape diversity (H), dominance (D), and evenness (E) indices for the 1981 ecopatch and landscape cover-type maps, calculated using natural logarithms. The number of cover-type categories = m. The comparison with m = 7 was achieved by combining cover-types into the following categories: forest, woodland, canyon walls, grass/shrub lands, water, urban, and agricultural.

Map	m	Hmax	H	D	E
Ecopatch	38	3.638	2.510	1.128	0.690
Cover-type	19	2.944	1.564	1.380	0.531
Ecopatch	7	1.946	1.314	0.632	0.675
Cover-type	7	1.946	1.436	0.510	0.738

of edge effects based upon comparative landscape mapping must be careful to consider the resolution of the map data, as well as the specific type of edge effect(s) of interest.

Comparison of landscape diversity, dominance, and evenness indices for the 1981 ecopatch and landscape cover-type maps (Table 5-11) shows that the number of cover-type categories used in the calculations can markedly affect the results. When both maps are reduced to seven basic cover-type categories the ecopatch map displays lower H and E and higher D values than the landscape cover-type map, primarily because water and urban types are smaller components of the ecopatch map. Increasing m to its maximum value for each map reverses the relative magnitude of their index values, with ecopatch now apparently the more diverse, evenly distributed map. These results again point out the need for caution when comparing these information theory-based indices between landscapes with different values of m.

ANALYSIS OF BANDELIER'S ADMINISTRATIVE BOUNDARIES

The importance of boundary processes for the design and management of nature reserves has been highlighted by Schonewald-Cox and Bayless (1986) and Schonewald-Cox (1988). A brief review of the ecological and management significance of the administrative boundaries of BNM follows.

Bandelier's boundaries may be divided into nine different segments based upon land ownership and management activities (Figures 2-2, 5-41). Segment one is the park boundary with USFS land along the Rio Grande. This is a logical, natural boundary. Difficult access to the White Rock Canyon edge of the Cerros del Rio, and the canyon and river themselves, restrict human and animal fluxes across this boundary into BNM. The SFNF plans to maintain the area along the eastern rim of White Rock Canyon as a visual buffer for the Bandelier Wilderness (USDA Forest Service 1987-c). The major issue associated with this segment is the occasional inundation of areas adjacent to the river by Cochiti Reservoir. Prolonged flooding is ecologically disruptive and has extirpated native plants and introduced alien species over large areas. Lake conditions also cause some loss of administrative control over the area as boats facilitate easy human access to this otherwise remote part of BNM which is difficult to monitor. Cochiti Reservoir impacts upon BNM exemplify the creation of a "generated edge" (Schonewald-Cox and Bayless 1986) which intrudes into the monument's administrative boundary. The current memorandum of agreement between the U.S. Army Corps of Engineers and BNM actually formalizes the loss of some park control over this area.

Segment 2 is the monument's southern boundary, adjoining the

Figure 5-41. Map of Bandelier National Monument boundary segment numbers, 1981 cultural cover-type patches, and 1981 road networks. The dotted line is the boundary of Bandelier National Monument, with segment numbers displayed. Cover-type patch key: Los Alamos National Laboratory technical areas and Cochiti Dam = magenta, residential = yellow, commercial = dark blue, ponds and Cochiti Lake at maximum flood pool stage = blue, pumice mines = black, agricultural fields = green, golf courses = orange, and ski area = horizontal striping. Road type key: paved = black, improved = yellow/green speckles, dirt = magenta, and primitive = blue.

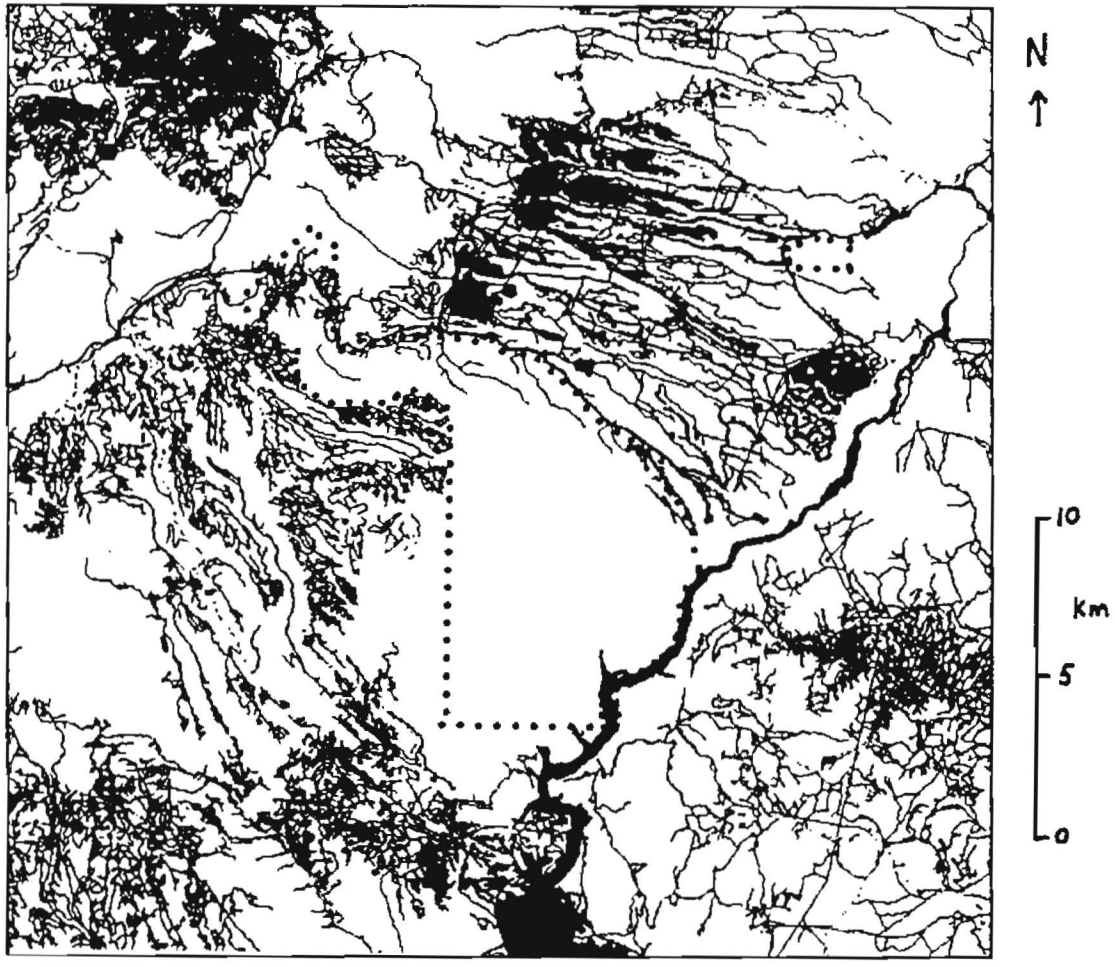


Figure 5-41.

Cañada de Cochiti Grant. This boundary cuts across the grain of the landscape and contains the mouths of three park canyons (Capulin, Medio, and Sanchez) and the ends of the mesas which separate them. The undeveloped Cañada tract acts as a buffer between BNM and the Town of Cochiti Lake developments on Cochiti Pueblo land. Remoteness from BNM's headquarters (in lower Frijoles Canyon) reduces administrative control over this backcountry region, which is a favorite entry point into the park for deer poachers. Burros continue to invade the park across this boundary from refugia on other lands. Currently owned by the University of New Mexico, adjacent portions of the Cañada de Cochiti Grant were authorized by Congress for addition to BNM in 1976, but this acquisition remains unrealized.

Segment three is another straight line boundary, separating the Bandelier Wilderness from the Dome Wilderness in the Santa Fe National Forest. Although this segment also ignores natural landscape patterns, the compatible land use outside the park boundary reduces potential problems - note the absence of roads on either side of the boundary in this area (Figure 5-41). The small Dome Wilderness contains headwater portions of the Capulin, Medio, and Sanchez watersheds and acts as a buffer for the monument. BNM and the SFNF have a joint "natural prescribed fire" program for this pair of wilderness area. Burros also cross into BNM through this segment, despite an expensive boundary fence.

Segment four continues the straight SFNF/BNM boundary north across Alamo Canyon to the rim of Frijoles Canyon, and then extends irregularly west to the park corner with private land. The roads visible in Figure 5-41 reflect the differing land uses across this

boundary, as logging, hunting, and livestock grazing occur on SFNF land adjacent to the Bandelier Wilderness. Headwater portions of Capulin and Alamo Canyons occur on SFNF land along this boundary. The SFNF is currently planning its next cycle of management activities for this area (the "Dome Diversity Unit") - BNM is participating in this planning process. A timber sale is scheduled for this area in 1991 (USDA Forest Service 1987-c).

Segment five follows the angular boundary between BNM and the private lands of the Baca Location, a National Natural Landmark containing the Valles Caldera. This straight-line boundary ignores topographic and vegetation patterns which may cause future management problems. For example, the boundary cuts through the margins of a relict montane grassland and comes to an apex exactly on the open summit point of Cerro Grande. This boundary complicates prescribed burning to maintain the grassland, will be a major aesthetic intrusion into this highly scenic area when fenced, and will force visitors to trespass onto the private land to partake of the majestic views of the Valles Caldera available 20 m across the fenceline. The conservative management policies of the Baca Location's current owners are maintaining the forests which adjoin the park in pristine condition, thus acting as a buffer for this caldera rim portion of the park. Major elk migration routes cross this boundary. The long-term management of these private lands remains uncertain, as development is only limited by the preferences of the land owners - land use regulations are absent in this area. The potential for future development of these lands is exemplified by the recent sale of a small tract at the segment four/five intersection; the owner of this small

tract has already built a residence here and offered of portion of this property for sale.

Segment six is another boundary between the SFNF and BNM. The current Forest Plan (USDA Forest Service 1987-c) has designated a threatened and endangered species emphasis area along the northern portion of this boundary, which in practice means a low probability of logging activity for at least the next several decades. However the rest of this segment is included in a planning area that is scheduled for logging activity in 1997. In 1983 this boundary was a source of conflict between the two agencies, as despite repeated objections by the park's superintendent BNM was forced to permit the SFNF to open and use a road across park land to log pristine forest along the boundary (BNM - numerous memos on file). The SFNF is proposing to acquire the small tract of private land along this segment through a land exchange, which would defuse the plan of the current owner to create a 48-parcel subdivision on this 20 ha tract. If the land exchange founders and the subdivision occurs, both BNM and the SFNM expect a variety of management problems to develop from the presence of a sizable human community within the surrounding wildlands (USDA Forest Service 1989).

Segment seven is the boundary between BNM and Los Alamos National Laboratory, which largely follows State Highway 4. This boundary follows the grain of the landscape across the Pajarito Plateau from the base of the Sierra de los Valles to White Rock Canyon, and roughly marks the northern extent of the Rito de los Frijoles watershed. The highway provides easy human access along this segment. The development and weapons research activities that occur on LANL lands causes some intrusion across this boundary into the park, for example, the near-

daily sounds of explosive tests. The park is somewhat buffered from LANL activities by the several canyons (Chaquehui, Ancho, and Water) that run between the park and most LANL facilities.

The last two boundary segments are associated with the small outlying portion of Bandelier, the Tsankawi Unit. The distance of this 334 ha unit from the park headquarters has reduced monitoring of visitor activities, resulting in significant amounts of illegal artifact collecting and vandalism on this archeologically rich tract. Segment eight comprises the southern and eastern borders with San Ildefonso Pueblo. The absence of development on these Pueblo lands has acted as buffer for Tsankawi. However, a major new highway linking Los Alamos to Santa Fe is being planned, and the two most probable routes cross San Ildefonso land south of the Tsankawi Unit. The development of a campground and other facilities near the south boundary has been considered by the Pueblo since the 1960's, and might occur with the new highway.

The ninth segment is the western and northern border between the Tsankawi Unit and LANL, which again follows State Highway 4. The major impacts associated with this boundary come from the ever-increasing traffic flow on this busy road, which intrudes upon the serenity, grand vistas, and illusion of prehistoric isolation that the Tsankawi Mesa often provides. One proposed route for the new Los Alamos/Santa Fe road would parallel the segment eight southern boundary, which would then enclose the Tsankawi Unit with major highways on 3 sides. All proposals for the new highway include an elevated interchange at the southwest corner of the unit. Despite its compact shape, the small size of the Tsankwi Unit leaves it highly vulnerable to external

impacts. Other external threats to park resources are described in Bandelier's nearly-completed revised Statement for Management (BNM - on file).

CUMULATIVE IMPACTS OF HUMAN ACTIVITY

The incremental effects of activities through time, when considered as a whole, constitute cumulative impacts. Large areas like landscapes are vulnerable to cumulative impacts from actions which appear minor when considered individually but which become significant when considered collectively (Odum 1982). Standardized methodology is currently nonexistent for assessing cumulative environmental impacts of human activities, although prominent recent efforts have used time-series analysis of air photos in conjunction with GIS technology to document landscape change through time (Dickert and Tuttle 1985, Walker et al 1987). Figure 5-41 combines the 1981 cultural cover-types of Figure 5-14 with the 1981 road network map of Figure 5-18 to provide one such picture of the cumulative impacts of recent human activity upon the local landscape of BNM. This image provides a perspective of human landscape alteration that is unavailable from ground observations alone. However, assessing the impact of the human activities implied by the mapped roads and cover-types upon a specific issue of concern, e.g., the affect of logging practices upon the reproductive success and population viability of spotted owl populations in the Jemez Mountains, requires fieldwork to integrate landscape-level patterns with the on-the-ground phenomenon of interest.

CHANGES WITHIN THE LANDSCAPE ELEMENTS OF BANDELIER

Landscape elements within BNM have changed greatly over the past century or so, mirroring the pervasive transformations previously described in LANDSCAPE ELEMENTS OF THE JEMEZ MOUNTAINS. From the Rio Grande to the summit of Cerro Grande, every major landscape element in the park has been altered through the interactions of human agency and natural processes (with the possible exception of certain canyon wall sites). The Rio Grande riparian zone has been remade by the effects of Cochiti Reservoir; mesa-top woodlands of piñon and juniper have become overstocked with trees and depleted of herbaceous ground cover, leading to severe soil erosion; ponderosa pine forests became unnaturally dense thickets that were catastrophically "thinned" by the 1977 La Mesa Fire, creating large expanses of open grass and shrubland; mixed conifer forests of the Frijoles Canyon headwaters have undergone changes in species dominance and forest structure that foreshadow future crown fires; and meadows and montane grasslands of long-standing site fidelity have been overrun by invasive trees. The landscape observed in and around BNM today is in many ways a cultural landscape, an artifact of historic human land use practices, not the pristine, "natural" wilderness envisioned by most park visitors.

CHAPTER VI. SUMMARY AND CONCLUSIONS

A. THEORETICAL AND METHODOLOGICAL CONTRIBUTIONS

1. LANDSCAPE ECOLOGY

My research in the Jemez Mountains focuses on two of the three core landscape characteristics emphasized by landscape ecology (Forman and Godron 1986), namely landscape structure and landscape change. The results presented here provide a multiple perspective framework for describing the structure of the Jemez Mountains landscape. This framework allows landscape-level changes to be documented and their management implications assessed. My findings contribute to the growing literature on the potential uses of and problems with indices of landscape pattern (O'Neill et al 1988).

The work reported here represents one approach for translating landscape ecology research into management-oriented results. The provision of an integrative landscape perspective establishes a context to organize and improve the management usefulness of the eclectic array of past and probable future ecological research in the Jemez Mountains. Specific landscape-level issues of management concern were examined in detail, e.g., fire history and vegetstion change. Current GIS technology has been utilized to create managerially useful maps and associated databases which will serve as the core of Bandelier National Monument's evolving natural resource inventory. The park's ability to justify and secure agency funding and public support for needed management programs, ranging from fire and vegetation management to erosion control, will be enhanced by a capacity to graphically present these and other research findings. The methodology developed in this case study is applicable to other National Park Service units.

My research has explicitly promoted the collection and integration of data from multiple spatial scales and points in time in order to produce a more holistic perspective of the Jemez Mountains landscape (C. Allen 1988). My reasons for advocating a multiple perspective approach to landscape ecology research are several. First of all, from a purely intellectual standpoint, multiple perspectives are required to more fully understand landscape structure, function, and change, as different structures, processes, and changes are observable at different hierarchical levels (Urban et al 1987). For example, important changes in landscape structure and function are associated with both road networks and soil erosion in the Jemez Mountains, but assessing landscape-wide road patterns requires the broad and elevated perspective of aerial photos, while soil sheet erosion that is apparent on the ground is nearly invisible from the air.

Secondly, resource managers need information at multiple spatial scales because they engage in management activities at different hierarchical levels. Manipulative management activities, e.g., timber harvest, prescribed fires, and wildlife habitat enhancement, primarily take place at the individual organism and ecosystem patch levels, influencing patterns and processes at these levels and the higher landscape scale. Yet management concerns are increasingly focused upon landscape-level issues, such as management of migratory wildlife species like elk, maintenance of viable populations of threatened and endangered species such as spotted owls, assessing the impacts of planned development projects like new highways, and cooperative interagency management of processes like wildfire. Ecological information is required at each level to assess the effects of current

practices and policies, and to determine what ,if any, management changes are needed.

Moreover, an agency like the NPS needs information on temporal landscape changes (both inside and outside park boundaries), especially those that have been caused by humans, in order to meet its mandate of maintaining resources "unimpaired" for future generations. Without scientifically sound information on landscape change, no management action will occur. For example, prior to my research on tree invasion of montane grasslands BNM believed that they knew enough about these grasslands to manage them, i.e. they benignly neglected them, not realizing that a grossly unnatural transformation was taking place. Based upon the solid evidence now available the park has initiated prescribed burning and tree cutting to maintain at least portions of the ancient Cerro Grande grassland.

Finally, I have a few comments about the limitations and usefulness of integrating these multi-level data sets. The human brain remains the best integrator of such eclectic information that I know of, as such integration is in part a creative act, blending the transferrable and useful parts of multiple perspectives to form a coherent whole. Integration is not a precisely repeatable procedure, and thus will never be be completely satisfying to ecologists who suffer from "physics envy" (cf. Egler [1986]). The ability of individuals to merge information depends upon many personal variables, including innate thinking style, training, and breadth and depth of background experiences. It requires familiarity with ecosystem components and processes at several different levels, both in the abstract classroom sense and on-the-ground in the particular landscape

under study. Differences in type, precision, and scale of information derived from multiple sources makes integration of such data challenging. Despite these limitations, conceptual integration allows eclectic, multi-level, spatiotemporal information to be considered together as a whole, to most fully describe and explain landscape patterns and processes. Scenarios of landscape-wide change cannot practically be created any other way.

2. DISTURBANCE ECOLOGY

I find that natural and human disturbances are ubiquitous in this landscape, and have attempted to weave a review of multiple disturbance regimes into a landscape perspective. From drought-induced bark beetle outbreaks to recurrent fires, natural disturbances have clearly influenced every landscape element in the Jemez Mountains. Historic human disturbance, notably livestock grazing, fire suppression, and timber harvesting, have also affected this entire landscape.

Several findings on local disturbances merit specific mention here. The landscape perspective provided by this study resolved widespread shifts in overstory vegetation ecotones caused by the 1950's bark beetle outbreaks. I am unaware of analagous documentation of bark beetle disturbance affects form other locations, although it seems probable that similar ecotone shifts have occurred in other Southwestern mountain ranges during drought-induced beetle outbreaks. The sampling of fire scars in several different landscape elements across a 1000+ m e'evational gradient has provided one of the most detailed and locally extensive Southwestern fire histories ever conducted. This study provides the first fire history data from

riparian mixed conifer and ponderosa pine/piñon-juniper forests in the Southwest. These data reveal one of the clearest relationships between dry years and fire years ever noted, a correlation that has been surprisingly hard to pin down (T. Swetnam, Tree-Ring Lab - personal communication). The frequent, widespread fires that occurred throughout the Frijoles watershed, and the onset of de facto fire suppression that accompanied the grazing of large livestock populations by the late 1800's, explain much of the historic vegetation change observed in the Jemez Mountains. Finally, extensive road network development in the Jemez Mountains has left a surprisingly strong signature of human activity, and potential impacts, upon the wildlands of this landscape; it deserves further investigation.

3. VEGETATION ECOLOGY

My work in the Jemez Mountains finds close linkages between disturbances and vegetation/landscape patterns. Every major vegetation type in my study area has been transformed in varying degrees by recent changes in the disturbance regimes operating in this landscape; as a result, contemporary vegetation is anomalous when viewed from the perspective of the past several hundred years. These findings support the view that vegetation may be quite dynamic over even the short time scale of decades, and challenge static perspectives of "climax" vegetation that speculate about successional patterns in the absence of disturbance (cf. Stewart and Hann 1983). In particular I find untenable the notion that there is a single, readily identifiable vegetation association that will dominate each site in the Jemez Mountains in the absence of future disturbance, an underlying premise

of the "habitat type" classifications that have become popular in most western USFS regions, including the Southwestern Region (Moir and Hendzel 1983, USDA 1987-a). I contend that individualistic patterns of plant species establishment and growth (Gleason 1926, Whittaker 1953), chance factors like concurrence of mast years with unusual weather conditions (Watt 1947), initial floristic composition (Eggler 1954), lag times between climate change and vegetation change (Watt 1947, Brubaker 1988), variability in site history associated with past disturbance events (Pickett and White 1985, Hamburg and Sanford 1986), and the ubiquity and variety of local disturbances (Christensen 1988) result in a variety of successional pathways leading to multiple possible vegetation types (species associations) on any given site in the Jemez Mountains. For example, several tree species grow well in Jemez montane grasslands and clearly have the potential to dominate these sites, and yet various lines of evidence show that grasslands have persisted as the dominant vegetation at these specific locations for hundreds or thousands of years. The site history associated with human-mediated changes in two disturbance regimes, fire and livestock grazing, has been vital in determining rates of historic tree invasion, while variability in the availability of local tree seed sources and climate has affected the species composition, timing, and rates of tree establishment. I believe that alteration of just one key disturbance regime, fire, could allow a variety of vegetation types to dominate the upper south-facing slopes of the Sierra de los Valles, ranging from the montane grasslands of the past or aspen stands to conifer forests with dominants ranging from ponderosa pine to Engelmann spruce. Speculation about potential natural vegetation in the absence of

disturbance seems unrealistic in a landscape like the Jemez Mountains where disturbances are ubiquitous and integral determinants of observed vegetation.

Failure to consider disturbance regimes or site history leads to observations of current vegetation that lack perspective about vegetation dynamics. For example, in describing the "subalpine grassland" type on Cerro Grande, Potter and Foxx (1981) state:

> This grass stand extends under the adjacent forest of ponderosa pine and mixed conifer in a most unusually high productivity and coverage for a forest stand. In fact, it is the densest grass cover the writer has ever seen under such a forest stand. Therefore, the vegetative type is really more extensive than the outline shown on the map (8 acres) but elsewhere it represents a forest understory.

The dense and highly productive grass understory occurs because it was present on the site long before the advent of recent tree invasion, unrecognized by the investigators. It is only now, as the invasive trees develop into closed-canopy forests, that the relict grass understory appears unusual. As the canopy closes the bunchgrasses eventually decline and most will die, as evidenced by the rotting skeletons of former grass clumps now found under some of these dense, young forests. Attempts to explain the transformed patterns of contemporary Jemez Mountains vegetation in isolation from disturbances and site history will produce simplistic results which lack explanatory power.

B. MANAGEMENT IMPLICATIONS

1. BANDELIER NATIONAL MONUMENT

My research finds that the landscape of the Jemez Mountains in and around BNM has been greatly altered by the interaction of human and natural processes over the last century or so. With respect to BNM these changes are of two general types: 1) direct modification of surrounding lands by human development activities; and 2) pervasive landscape-wide transformations of the major landscape elements within BNM. Both sets of landscape changes pose challenges for BNM managers which are examined below.

Changes in the Landscape Surrounding Bandelier

External threats to park resources from human activity on surrounding lands are of increasing concern at BNM, as they are throughout the National Park system (Schonewald-Cox and Bayless 1986, Stottlemyer 1987). Persistent external impacts to Bandelier are associated with ongoing activities at LANL, managment of SFNF land in the headwaters of several park watersheds, the operation of Cochiti Reservoir, and regional air and water pollution. The growing list of potential external threats to the park includes the construction of a major new highway between Santa Fe and Los Alamos, possible development of a new regional airport near Santa Fe which would bring increased air traffic over the Bandelier Wilderness, development of private tracts of land which adjoin the park, and the long-term effects of fragmentation and alteration of the surrounding landscape on resources like biodiversity and scenic vistas. Mitigation of external threats to BNM requires cooperative management at a landscape level.

Changes in the Internal Landscape of Bandelier

Recognition of pervasive, human-induced changes within the landscape elements of the park itself presents philosophical and logistical challenges for Bandelier's managers. The central guiding policy statement for natural resource management of National Park Service units states: "The National Park Service will manage the natural resources of the national park system to maintain, rehabilitate, and perpetuate their inherent integrity" (USDI National Park Service 1988). Philosophically, evidence of widespread landscape change forces managers to confront the difficult questions of "what is natural?" and "what specific management goals should we be pursuing to preserve a natural landscape?" The sometimes vociferous national debate over proper answers to these questions (cf. Bonnicksen and Stone 1985, Parsons **et al** 1986, Agee and Huff 1986, Christensen **et al** 1986, Chase 1987, Bonnicksen 1988, Agee and Johnson 1988, NPCA 1989, Peterson and Krumanaker 1989, Sellars 1989) has left managers in the parks with surprisingly little guidance on how to determine what is "natural". My research shows that very little of the Bandelier landscape is strictly natural in the sense of unaltered by interactions with human beings, but this still leaves BNM managers the task of determining if the documented landscape changes require management intervention "to maintain, rehabilitate, and perpetuate (the) inherent integrity" of park resources.

How much management intervention is justified in our national park and wilderness landscapes? The National Park Service is a conservative agency; NPS recognition of past mistakes, such as elimination of large predators, absolute fire suppression, and widespread application of

pesticides, has bred a justifiable sense of caution against excessive intervention in national park landscapes. Unfortunately, the magnitude of human-mediated changes to even our most pristine natural areas requires active management responses (Leopold et al 1963, Chase 1987, NPCA 1989).

The issues at Bandelier are further complicated by occasional philosophical conflicts between management of the cultural resources for which the park was originally established and management of most of the park as a natural zone (as explicitly required by the 1976 designation of the Bandelier Wilderness). Cultural resource management fights the natural processes of weathering, decay, disturbance, and succession in an effort to preserve human artifacts or a specific cultural landscape, while natural resource management in the NPS is now mandated to emphasize the maintenance of natural **processes** rather than the perpetuation of a particular **structure**. As stated in the new Management Policies (USDI National Park Service 1988): "... change (will) be recognized as an integral part of the functioning of natural systems. The National Park Service will not seek to preserve natural systems in natural zones as though frozen at a given point in time." How much management protection from natural degradational processes should be given to the thousands of archeological sites in the Bandelier Wilderness? What about protection from human-initiated, but now naturally self-sustaining, erosion?

Another philosophical dilemma may arise soon from the current policies which favor management of "natural processes" over managing for a target natural system structure. The problem I foresee with management for natural processes remains the determination of what is

"natural" in a world of fragmented landscapes and human-mediated climate change. Impending (ongoing? [Kerr 1989]) global climate change in particular is a wild card that has not been accounted for in current management scenarios (Roberts 1988, Graham 1988). The transitory nature of plant and animal communities will be highlighted by shifts in species ranges in response to changes in climate and disturbance regimes (Brubaker 1988, Hunter et al 1988). Rare orchids and giant sequoias, old-growth forest reserves and their spotted owls, all may be forced into extinction on their current human-designated landscape locations by altered environmental conditions. It will become extremely difficult to determine which extinctions are "natural", or what constitutes a "natural" fire regime, if current models of rapid climatic change are realized. The unresolvable uncertainties that rapid changes in climate could induce in our national park landscapes may become so overwhelming as to restrict NPS resource managers to frustrated inaction if maintenance of undeterminable "natural processes" is the key to action. More likely the NPS will comply with the general guidelines of their management policies ("The extent and degree of management actions taken to protect or restore park ecosystems or their components will be determined in light of management objectives and prevailing scientific theory and methodologies" [USDI National Park Service 1988]) by following the pragmatic advice of Lovaas (1989) to "do the best we can". The decisions faced by natural resource managers in the coming decades are not going to get any easier...

Pervasive landscape change within BNM also presents logistical problems for managers. Internal changes caused by external influences,

such as Cochiti Reservoir impacts or landscape-wide decline of an endangered species (pushing it below minimum viable population levels within the park), are outside direct park control and require cooperative arrangements with other land managers. Some changes may be irreversible and thus beyond full correction, such as past soil erosion in piñon-juniper woodlands. The extensive scale of management intervention required to control ongoing soil erosion or effectively re-institute pre-1900 fire regimes through prescribed burning are severely constrained by limited funding and personnel. The wilderness designation of most of the park also limits the tools and techniques available to treat a problem like widespread soil erosion. Finally, certain management techniques may be difficult to implement due to lack of public acceptance (e.g., cutting of trees to control unnaturally dense forest stands).

Landscape Management

I believe that the implications of pervasive external and internal landscape changes indicate that the National Park Service in general, and BNM in particular, must learn to manage **landscapes**. Others have called this expanded, holistic perspective "ecosystem management" (Agee and Johnson 1988), but my preference for the term landscape management stems from the broad array of definitions and spatial scales applied to the term ecosystem (cf. Lidicker 1988). I think that ecosystem management is a "fuzzy" term which will mean different things in the minds of different resource managers and will never be clearly understood by the public. In contrast, landscape is a relatively well-defined level of spatial scale that is understood by professionals and

the public alike. People are familiar with their local landscapes as this is the scale at which they live and work, and this is also one level at which most natural resource agencies operate. Regardless of what it is called, landscape management must increasingly be practiced if semblances of natural landscapes containing full complements of native structures and processes are to maintained (Agee and Johnson 1988).

The probability of rapid climatic change in coming decades is another reason why landscape management is going to flourish. Resource managers are going to have to find ways to tie our increasingly fragmented landscapes back together again or accept the large losses of biodiversity that will result as climate-forced changes in species distributions confront anthropic barriers to dispersal. Though a small park, Bandelier is fortunate to cover a 1500 m elevational gradient which will allow some species range shifts to occur internally as climate changes.

Current management policy in the NPS is moving toward landscape management, with increased emphasis on ecosystem management and cooperative management with adjoining agencies (USDI National Park Service 1988, Agee and Johnson 1988). Bandelier is increasingly interacting with its neighbors in discussions of land management and potential impacts upon the park, and BNM has cooperative agreements with other agencies covering issues ranging from fire management to resource protection on adjacent lands.

2. OTHER LAND MANAGERS IN THE JEMEZ MOUNTAINS

The emerging importance of landscape management applies to other landscape management agencies as well. There are numerous landscape-level concerns shared by resource management agencies and private land owners in Jemez Mountains that could serve to focus coordinated landscape research and management actions. For example, cooperative agreements between the USFS, US Fish and Wildlife Service, and the New Mexico Department of Game and Fish already structures coordinated research on and management of Jemez Mountains salamanders and spotted owls. Perhaps a landscape-wide system of old-growth forest patches and connecting corridors could be established to maintain dependent species and other associated resources and values. Discussions are underway between the NPS, USFS, and Corps of Engineers regarding cooperative efforts to rehabilitate and manage the reservoir-altered riparian zone of the Rio Grande. Accelerated soil erosion is occurring on piñon-juniper woodlands throughout the Jemez Mountains and is another candidate for cooperative investigation of treatment strategies. Management of burgeoning local elk populations clearly requires coordination between the state, local land owners, and land management agencies. From maintenance of biodiversity and planning of major new roads to protection of surface water quality, the future landscape of the Jemez Mountains will be increasingly shaped by landscape-level interactions among the diversity of local land owners and managers.

C. FUTURE PROSPECTS FOR THE JEMEZ MOUNTAINS LANDSCAPE

The wildlands of the Jemez Mountains have in many respects become a cultural landscape. Landscape form and function in the Jemez Mountains are not simply inscribed in stone and genome, but increasingly reflect human conceptions about what this landscape should be like that arise from the "inscapes" of our minds (cf. Dansereau 1975, Tuan 1976). A landscape perspective may provide one framework to escape the people/nature dichotomy that has become increasingly unsustainable both conceptually and materially. Widespread recognition that all landscapes today result from a blend of cultural and natural interactions may allow land management agencies, local communities, and the general public to acknowledge and act upon the increasingly obvious fact that humans are now responsible for the fate of what we once called the natural world.

Perhaps a biosphere reserve type of landscape will develop in the Jemez Mountains, with complementary zones of land ownership and use working together to meet various landscape-level goals (cf. IUCN 1985). Yet in the absence of an overall landscape vision of the Jemez Mountains, current opportunities to mold a desirable landscape by conscious choice and awareness of tradeoffs will be lost to the cumulative impacts of piecemeal 'development'. Many obstacles to landscape management certainly exist. Improved communication and relationships between land management agencies and local communities are essential to the development of concensus landscape goals. Reconciling the different perspectives and interests of groups ranging from Pueboloans and federal bureaucracies to cattlemen and environmentalists will certainly be challenging. It is still unclear

if the landscape which enters the 21st century will be touted as an example for others to emulate or mourned as an opportunity lost.

Despite the obstacles, I remain optimistic. Given northern New Mexico's long history of people viewing the land as a community to which they belong, and the diverse cultural and natural components of the Jemez Mountains, the potential for the emergence of a harmonious cultural/natural landscape here is great. Concensus on many issues is possible if an honest, cooperative attitude is displayed by all parties. The recent land management planning efforts of the SFNF, while flawed by inadequate information and an over-reliance upon abstract, black-box FORPLAN computer models, have been a step in the right direction by outlining long-term alternatives at a landscape level and in soliciting public input. Cooperation between local land management agencies has been increasing. If enough people come to care and be informed about this landscape, and if local communities and land management agencies take landscape-wide, long-term perspectives seriously, I believe that the Jemez Mountains will become known as a special place in northern New Mexico where loggers and recreationists learned to co-exist with Jemez Mountains salamanders and with each other - a landscape which offers hope that our civilization can replace its carpetbagger economics and attitudes with a land ethic in time for us to live as nurturing members of a balanced landscape community. The Jemez Mountains are a good place to explore the creation of such a landscape.

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