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# **Assessment of Top Down and Bottom Up Controls on Fire Regimes and Vegetation Abundance and Distribution Patterns in the Chihuahuan Desert Borderlands: A Hierarchical Approach**

## **Final Report to the Joint Fire Science Program**

Project # **03-3-3-13**

Project Location:

Big Bend National Park, the Maderas del Carmen Protected Area in Coahuila, Mexico, and the Davis Mountains Preserve of the Nature Conservancy

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## Executive Summary

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### Overview

The goal of this study was to quantify historical fire regimes, climate forcing of fire events, plant population dynamics, and fuel abundance and distribution patterns in the Chihuahuan Desert Borderlands. Our results provide information about recent changes in fire regimes, forest structure and species composition as a result of fire exclusion (grazing) and direct fire suppression. This study area straddled the US-Mexico border, and included Big Bend National Park (BIBE), and The Davis Mountains Preserve of The Nature Conservancy (DMTNC) in Texas, and the Maderas del Carmen Protected Area (MCPA) in Coahuila, Mexico (Fig. 1). These preserves were chosen for this study because they comprise a regional-scale representation of the variation in fire regime characteristics and forest vegetation types in a region where no fire or forest inventory data existed. Furthermore, they represented understudied pinyon-juniper and pine-oak vegetation assemblages in southwestern North America. Each of the three preserves is run by a different management agency, which enabled the sharing of experiences across agency boundaries, and provided a large-scale perspective of fire-climate-vegetation interactions in this region. Data from this study provide critically needed reference conditions and targets for ecological forest restoration to lower the risk of future high intensity fire.

**Figure 1: Location of the three study sites in the Chihuahuan Desert Borderlands.**



## Objectives

The primary objective of this study was to characterize the relationship between climate, fire, fuel loads, and vegetation in the Chihuahuan Desert Borderlands to provide managers from the region with reference conditions for forest restoration and management. Specific objectives were to:

1. Assess the range of the historical natural variability of fire regimes at the landscape and regional scales in Chihuahuan Desert Borderlands.
2. Determine the extent to which climate influences fire regimes (frequency, magnitude, seasonality, and spatial extent) and vegetation patterns (structure and composition) in this region.
3. Describe the relationships between environmental site conditions and tree species composition and diversity at the local-, landscape-, and regional-scales
4. Quantify the extent to which fire suppression has altered fire regimes, fuel loads, and plant population abundances and distribution patterns.
5. Develop reference conditions for historical fire regimes and forest stand structure in BIBE, DMTNC, and MPCA.
6. Use forest inventory data, landscape metrics, and spectral information to model vegetation and fuel loads across the landscapes of the three study areas.

## Methods

Extensive fire history and forest inventory data were collected using a uniform sampling design at each study area to enable between-site comparisons. Two hundred fifty fire scar samples were collected across the three sites to reconstruct historical fire regimes. Fire history data were used in combination with tree ring indices and direct measurements of precipitation to quantify fire-climate interactions. Vegetation and fuels data were collected using a gridded sampling design across the forested areas of each of the three sites to quantify overstory stand structure, tree species composition, and fuel loads. Age data from 1387 trees were collected to determine the extent to which contemporary forests date back to the last major fire events in each site. Environmental data were collected at each vegetation plot and with the aid of a geographic information system to relate forest and fuel distribution patterns to local environmental variables. Finally, spectral information from satellite imagery was used along with the environmental data to generate spatially explicit predictive maps of vegetation and fuels for each study site.

## Key Findings

### Fire history

- All three study areas experienced frequent fires before the onset of fire exclusion. Mean fire intervals prior to fire exclusion for fires scarring 10% or more of sample trees were: DMTNC 7.2 yrs (range 1-16), BIBE 7.5 yrs (range 2-34), and MPCA 7.7 yrs (range 2-24).
- Fire return interval data match the range of historic natural variability of fires in other regions of the Southwest that were also characterized by frequent, low intensity fire.

- Fire return intervals were frequent in all forests, and even in pinyon pine and juniper dominated vegetation types, a finding that informs the current debate about fire periodicity in pinyon-juniper woodlands.
- Fires were predominantly small in size; the majority of fires did not scar 25% of the sample trees used for the fire history reconstruction.
- The last major fire dates differed in each site. Frequent fires ended after 1926 in DMTNC and 1937 in BIBE, but continued in MCPA until 1951. These dates differ from other sites in the Southwest where fire exclusion occurred in the late 1880s in response to cattle grazing. While our study sites also supported large numbers of cattle during this era, fire exclusion in this region occurred after the introduction of sheep and goats in the mid-1900s. Direct fire suppression in BIBE after the park was founded in 1944 also contributed to the lengthening of the fire return interval there.
- After the introduction of goats to the region, fire return intervals increased two-fold.
- Fires occurred predominantly in the early season according to the intra-annual ring position of fire scars. Fire scars were concentrated in the dormant (90%) and earlywood (8%); fewer than 2% of fires occurred in the latewood or ring boundary.
- Superposed epoch analysis indicated that fire years were unusually dry. Wet climatic conditions in years preceding fire events probably promoted the accumulation of fine fuels that were subsequently dried in fire years; however only the dryness in the year of the fire was significant.
- Large fires were synchronous across the three sites, and occurred on years in the positive phase of the Southern Oscillation Index (La Niña years).

#### Stand Structure

- Age structures of trees in the three sites reflect the regeneration response to fire exclusion. The age structure of trees in MCPA may have also been influenced by logging activities during the early 1900s.
- The dates of tree regeneration correspond closely to the introduction of sheep in the early to mid-1900s.
- Reverse-J size-class distributions characterized the size-structure of conifer species, indicating high numbers of small, potentially young trees in all vegetation types. Total tree densities were also high, and ranged from 700-3100 trees ha<sup>-1</sup>.
- Sprouting species including oaks had multi-modal size-class distributions that most likely represented the life-history characteristics of these trees in relation to fire. Oaks readily regenerate via post-fire sprouting, and the high density of intermediate-sized oaks is probably an artifact of the last major fire events in each site.
- Surprisingly, while MCPA was the least fire-suppressed site in our study, it had higher densities of small diameter trees than BIBE and DMTNC. This suggests that factors other than the cessation of frequent, low intensity fires may control tree abundance in this region.

### Vegetation Distribution Patterns

- Nine dominant vegetation types were identified, and their distribution varied in association with local environmental site conditions.
- Mesic valley bottoms supported vegetation types composed of mesophytic tree species.
- Drier sites on steep, upper topographic positions with higher incident solar radiation supported sparsely populated vegetation types composed of more drought-tolerant tree species.
- Species richness (diversity) was highest for mesic vegetation types (gallery forest, Arizona cypress-fir, and Graves oak), they represented a small fraction of the landscape.
- Total species richness was highest in MCPA, followed by BIBE and DMTNC. This species richness pattern follows a latitudinal gradient, and may also be related to the size, orientation and topographic complexity of the different mountain ranges.

### Fuel Distribution Patterns

- Ours was the first study in the western United States to quantify fuel distribution patterns in relation to physiography and topography.
- This is also the first study to investigate the abundance and distribution patterns of snags in forest types other than Jeffrey and ponderosa pine in southwestern North America.
- Five dominant fuel load types were identified in the study area. Fuels distribution varied with local environment, rather than vegetation type.
- Valley bottoms with low incident solar radiation had higher productivity and fuel accumulation. Transport of fuels from upper topographic positions to lower topographic positions by gravity and sediment flow was also a probable factor influencing fuel distribution patterns.
- Steep slopes and upper topographic positions had low fuel accumulations as a result of the lower productivity and tree density of these sites, and the transport of fuels downslope from these locations.
- Snag densities varied widely across the landscapes of DMTNC, BIBE and MCPA, suggesting the heterogeneity in sites for cavity nesting birds and bats across a range of forest types.
- Snag diameters were smaller in our sites than reported in previous research due to the dominance of smaller trees including pinyon pines, junipers and oaks.

### Predictive Mapping of Vegetation and Fuels

- The fuels maps differed greatly from the vegetation maps at all three sites. This highlights the importance of mapping fuels separately from vegetation in fuels.
- The link between the forest inventory data and the vegetation and fuels maps can be used by managers to target locations on the landscape for biodiversity maintenance, wildlife habitat modeling, and fuels reductions treatments.

### **Key Implications for Management**

- The information presented in this study represents one of the only fire scar-based fire history studies to include pinyon pine and juniper woodlands, as well as better-known fire-scarring species like ponderosa pine, southwestern white pine, and Douglas fir. Fire history information in the pinyon-juniper forest type could be used as a reference for historical fire behavior in other similar pinyon-juniper woodlands of the southwestern United States and northern Mexico.
- Historical information on fire regimes can inform management activities directed towards restoring forest stand structures to their pre-fire exclusion states to reduce future risk of uncharacteristically high intensity fire/
- The later onset of fire exclusion in our study areas relative to other regions of the southwest suggests that these forests may be more easily restored through the use of prescribed fire than other forests in the Southwest. Caution is warranted because contemporary stand densities and fuel loads may cause future fires to behave quite differently than historical fires.
- The distribution of vegetation (and hence diversity) on the landscape suggests that mesic vegetation types may be of particular interest for management oriented toward the preservation of rare vegetation assemblages that contain high species diversity.
- Uplands and valley bottoms are at greater risk of high intensity fire than other locations on the landscape because of increased fuel loads.
- Our snag data for pinyon-juniper and pine-oak woodlands from represent new information about wildlife habitat in these understudied forest types.
- The vegetation and fuels maps will be useful as decision support tools for a variety of management activities.

## **Table of Contents**

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<b>Chapter 1</b>	<b>Page 8</b>
<b>Project Justification</b>	
<b>Chapter 2</b>	<b>Page 12</b>
<b>Fire Regime Characteristics, 20<sup>th</sup> Century Change in Fire Frequency and Forest Age Structure</b>	
<b>Chapter 3</b>	<b>Page 24</b>
<b>Climate as a Forcing Mechanism on Fire</b>	
<b>Chapter 4</b>	<b>Page 28</b>
<b>Vegetation Types, Forest Size Structure, and Species-Environment relationships</b>	
<b>Chapter 5</b>	<b>Page 42</b>
<b>Tree Species Diversity</b>	
<b>Chapter 6</b>	<b>Page 46</b>
<b>Fuel Distribution Patterns</b>	
<b>Chapter 7</b>	<b>Page 53</b>
<b>Predictive Mapping of Vegetation and Fuels</b>	
<b>Chapter 8</b>	<b>Page 59</b>
<b>Conclusions and Management Implications</b>	
<b>References</b>	<b>Page 65</b>

## Chapter 1 Project Justification

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Sustaining vegetation characteristics of National Parks and other protected areas within fire-regulated landscapes requires fundamental knowledge about the role of climate and fire in developing and maintaining local, landscape, and regional vegetation patterns. Reference conditions represent the range of historic (or natural) variability in ecological structures and processes. They reflect the recent evolutionary history and the dynamic interplay of biotic and abiotic conditions and disturbance patterns that can be used to form the basis for comparison with contemporary ecosystem processes and structures (Landres et al. 1999). This frame of reference is often used as a benchmark for designing ecological restoration treatments and management plans (Fulé et al. 1997). Establishing reference conditions to assist fire management activities is difficult. However, information about the historical controls on fire regimes can generate process and structural objectives for restoring or maintaining sustainable vegetation patterns (Stephenson 1999). Predicting and responding to current and future climate impacts on disturbance regimes and vegetation patterns demands a more comprehensive understanding of the interactions among climate, fire, vegetation, and fuels.

Reference conditions and knowledge about how climate and fire have shaped vegetation patterns in the Chihuahuan Desert Borderlands is lacking, jeopardizing the sustainable management of National Parks and protected areas on both sides of the US - Mexico border. Current vegetation structures and compositions are strongly influenced by interannual- to multidecadal-scale climate variability and corresponding variation in fire regimes (Swetnam and Betancourt 1998). Processes influencing local and regional fire regime parameters (frequency, magnitude, seasonality, and spatial extent) include top down (climate) and bottom up (species composition, fuel loads, topography, environment and human activity) controls. Knowledge about the individual, let alone synergistic, contributions of these controls is almost non-existent for this region (but see Moir 1982 and Brown et al. 2001).

BIBE and DMTNC are separated from MCPA by only the Rio Grande River, which forms the US- Mexican border between the states of Texas, Chihuahua, and Coahuila (Figure 1). Divergent fire and land management histories in the U.S. and Mexico provide a unique opportunity for comparing vegetation effects under different levels of fire exclusion (grazing) and direct fire suppression. The proximity of all three reserves allows for the investigation the effects of natural and anthropogenic fire regime variability on vegetation dynamics. This study generates new knowledge about relationships among climate, fire and vegetation at local, landscape and regional scales for the Chihuahuan Desert Borderlands. Results from this study provide a scientific basis for management decisions within the three reserves as well as in nearby areas with similar forest types. The scientific quantification of forest stand structure in relation to fire regimes is desperately needed by fire managers to justify the use of prescribed fire in this region where resistance to the reintroduction of fire is strong at local and regional levels.

### Project Description

We collected forest inventory data, tree ring data, and fire-scar samples from the three sites in our study to 1) quantify the historical range of variability of fire regimes in

the Chihuahuan Desert Borderlands, 2) determine when fire exclusion began in the region, and whether it occurred synchronously across the three sites, 3) quantify the relationship between extreme climatic events and fire occurrence, 4) assess the changes in forest stand structure and fuel loads resulting from the disruption of frequent, low intensity fire regimes, 5) quantify species-environment and diversity-environment relationships for tree species, and 6) generate predictive maps of vegetation and fuels. The products from this project provide site-specific and regional-scale information about the relationship between fire and climate, as well as reference conditions for the implementation of future forest management activities.

### Study Area Description

The three reserves are within the “sky island” mountain ranges, a floristically diverse region of relict mixed conifer forests. Each range has unique combinations of topographic extremes with a corresponding diversity of habitats. These mountain ranges are biologically significant because of their high species diversity and large numbers of endemic taxa. Located on the southeastern edge of the Basin and Range Geographic Province in the Trans-Pecos, these mountains share biological affinities with Rocky Mountain and Madrean floras. Chihuahuan Desert grasslands exist at lower elevations. Upper elevations contain post-Pleistocene refugia composed of relict montane conifer forests. The mountains are volcanic and originated 35-39 million years ago in the same Eocene to Oligocene orogeny that formed most of the Front Range of the Rocky Mountains. High elevation forests are composed of oak-juniper-pinyon and conifer woodland. Oak-pinyon-juniper woodland occurs at elevations of 1370-1707m and consists of evergreen oak (*Quercus sp.*), juniper (*Juniperus flaccida* and *J. deppeana*), and Mexican pinyon pine (*Pinus cembroides*). Conifer woodland occurs at elevations between 1677 and 2287 m, and contains ponderosa pine (*Pinus ponderosa*), southwestern white pine (*Pinus strobiformis*), upland oaks (*Q. hypoleucoides*, *Q. gravesii*, and *Q. sideroxylla*), aspen (*Populus tremuloides*) and other evergreen oak associates.

The modern climate is arid, characterized by cool winters and warm summers. Annual precipitation ranges from < 200 to > 480 mm and is distributed bimodally. The majority of the precipitation falls during late-summer “monsoons” (mid-July through mid-September), with moist Pacific and Arctic fronts responsible for precipitation occurring between December and March. Summer monsoons are produced by low-level moisture advection from the eastern tropical Pacific-Gulf of California, with upper-level moisture contributions from the Gulf of Mexico (Adams and Comrie 1997). Moisture from these two sources is thought to mix over the Sierra Madre Occidental in Mexico prior to being transported northward into the United States. Winter storms occurring from December to March are generally from frontal systems coming from the northwest. A pronounced dry season usually occurs between winter storms that end in March or April and the onset of the summer monsoon in mid-July. Prevailing winds are southerly to southwesterly from October to April and southeasterly from May to September.

Climate strongly influences fire seasonality in much of the southwestern US and northern Mexico, no data exist for west Texas or northern Coahuila. Most fires occur just prior to the onset of the North American Monsoon System (NAMS) in Arizona and New Mexico (Swetnam and Betancourt 1998), they occur earlier in the year in much of northern Mexico (Heyerdahl and Alvarado 2003). The fore-summer period prior to the

NAMS onset is usually very dry, with low humidity and high temperatures that reduce live and dead fuel moisture levels, rendering highly flammable vegetation and fuels. Fires also occur during mid-summer monsoons as a result of lightning ignitions during rainstorms.

Climate variation affects fundamental ecosystem processes on local, landscape and regional scales. Warming climates tend to support more frequent disturbance and changes in forest composition (Overpeck et al. 1990). Fuel accumulation and fuel moisture levels are also highly reactive to changes in climatic conditions (Miller and Urban 1999). Cyclic and synoptic climate patterns caused by the El Niño Southern Oscillation (ENSO) may have pronounced effects on fire frequency, magnitude, synchrony, and spatial scale (Swetnam and Betancourt 1990 and 1998; Grissino-Mayer and Swetnam 2000). There appears to be a strong linkage between wet climate cycles during which fuels accumulate, and dry climate cycles when the accumulated fuels burn in fire events. However, current knowledge about such climate-fire relationships is limited to a small number of sites. Understanding how climate forcing of fire regimes affects vegetation patterns over longer temporal and wider spatial scales can help better predict how recent warming trends in climates may impact the magnitude and size of future fires in the western United States.

Top-down and bottom-up processes influence fire regimes and vegetation patterns. Climate is a top-down control that has a pronounced effect on vegetation structure and composition because of its influence on tree regeneration patterns. Variations in climate and fire influence forest stand structure by promoting tree recruitment during wet periods, and inhibiting tree establishment during dry climate cycles (Mast et al. 1998; Barton et al. 2001). Bottom-up controls that include species composition, environment and topography have significant effects on fire regime parameters (Heyerdahl et al. 2001, Taylor 2000, Miller and Urban 1999, Camp et al. 1997). Effects of top-down versus bottom-up controls on fire regime and vegetation structure and composition have been studied in only a few locations. Deciphering these relationships requires additional research at multiple scales, with a payoff for land managers that includes a scientific basis for developing management plans to sustain desired forest ecosystem structure and species composition.

Human interactions with fire and vegetation also affect ecosystem processes. Direct fire suppression, livestock grazing and logging interact synergistically with variations in climate to alter fire regimes, as well as forest structure and composition (Savage 1991, Camp 1999, Donnegan et al 2001, Conard et al. 2002). Biomass accumulation, reduction of fine fuels through livestock grazing, and fire suppression can exacerbate the effects of climatic conditions on fire regimes and vegetation. There is general consensus that removing fire from forested landscapes in the western United States has increased the density of young trees, shifted forest composition towards shade tolerant, but fire intolerant tree species, and increased accumulations of live and dead fuels. In contrast, the montane conifer forests of northern Mexico exist under a variety of fire regimes (Fulé and Covington 1997, Heyerdahl and Alvarado 2003, Stephens et al. 2003, Fulé et al. 2005). Some of these forests have not been widely subjected to fire exclusion and could provide reference conditions for sustainable vegetation patterns. These reference conditions are invaluable templates for restoring National Parks and other protected areas to more sustainable conditions. While it is difficult to determine the

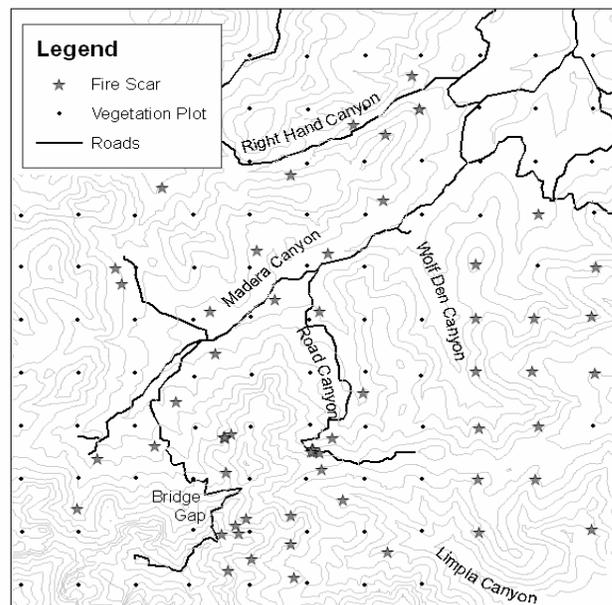
extent to which human-caused changes in disturbance regime affect natural ecosystem processes, a regional-scale interpretation of climate variation, fire regime and land-management history will provide additional insight into how synchronous and interrelated agents of disturbance influence ecosystem structure and function.

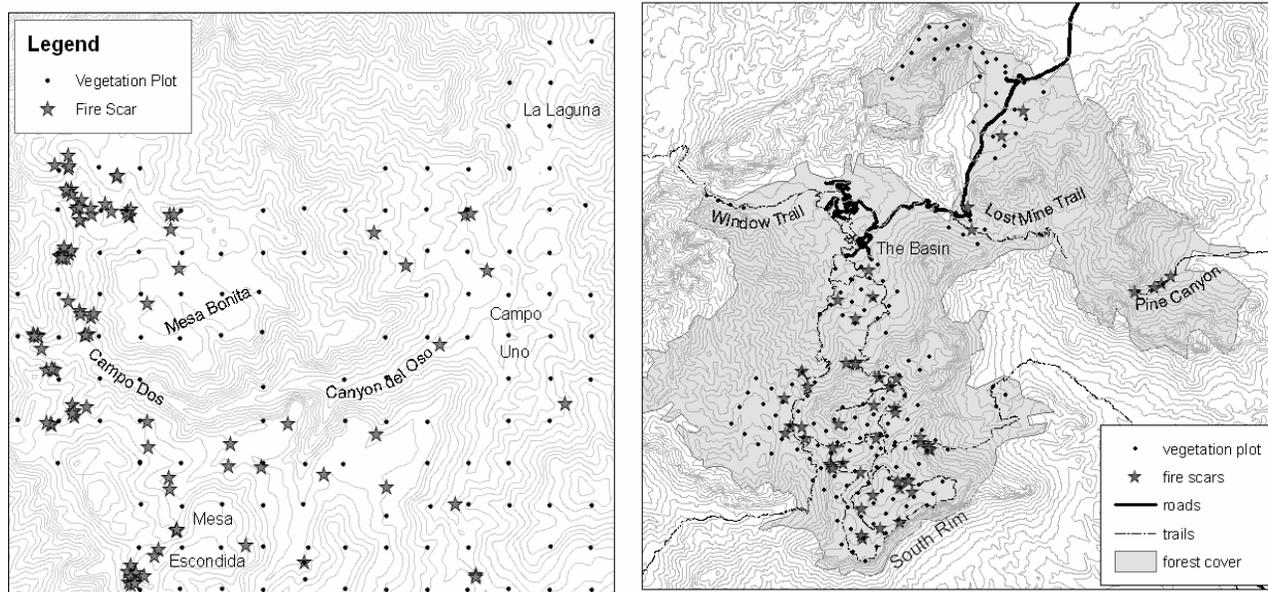
## Chapter 2 Fire Regime Characteristics, 20<sup>th</sup> Century Change in Fire Frequency and Forest Age Structure

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We investigated the historical range of natural variability of fire regime characteristics and recent changes in fire frequency in each of the study areas by sampling two hundred fifty fire-scarred trees. At each site, we searched in a 75-m radius around the center of each vegetation plot (vegetation sampling is described in detail in chapter 4). Full or partial cross-sections were cut from scarred “catfaces” on trees, snags, logs, and stumps of conifers that form consistent annual growth rings. Trees with multiple fire scars were preferentially sampled to extend the fire record as far back as possible. Samples from living trees were collected as partial cross-sections, a method that does not require felling the tree (Agee 1993). Samples locations were geo-referenced and were well distributed throughout the three study sites (Fig. 2). Most samples were from *Pinus strobiformis*, followed by *P. ponderosa*, *P. cembroides*, *Pseudotsuga menziesii*, and *Abies coahuilensis*, respectively.

**Fig 2. Fire-scar and vegetation plot locations in a) MCPA, b) BIBE, and c) DMTNC**





Fire-scar samples were aged by sanding them to a high polish and visually cross-dating them under a binocular microscope using standard dendrochronological techniques (Stokes and Smiley, 1968). Master chronologies for each site were used to aid in cross dating. Master tree-ring chronologies were developed for DMTNC and MCPA for use in cross-dating the tree cores and fire-scar samples since no chronology existed for that site (Villanueva et al., unpublished data). A master chronology for BIBE site was obtained from the international tree-ring database. The season of fire occurrence (Baisan and Swetnam 1990) was estimated based on the relative position of each fire lesion within the annual ring according to the following categories: early earlywood, earlywood, middle earlywood, late earlywood, latewood, and dormant. Dormant season scars were assigned to the year following the earlywood (i.e. spring fires), a convention that appears valid for the spring drought-summer monsoonal climate pattern of northern Mexico (Fulé and Covington 1997, 1999, Heyerdahl and Alvarado 2003). The season was listed as “not determined” when it could not be distinguished clearly.

Fire history data were analyzed with FHX2 software (Grissino-Mayer 2001). Analysis at each site began with the first year with an adequate sample depth which was 1772 (Grissino-Mayer et al. 1994), defined as the first fire year recorded by 10% or more of the total sample size of recording trees at each site. For our sites this year was 1772. “Recorder” trees are those with open fire scars or other injuries (e.g. lightning scars), leaving them susceptible to repeated scarring by fire (Swetnam and Baisan 1996). We considered fire scars only as proxy evidence for dates of fire events, and not as a true representation of burning across the landscape (*sensu* Baker and Ehle 2001). We assumed that during any fire there were many trees that had fire at their base but did not record a fire scar that invalidated assessments of precise spatial patterns of burning from fire-scar evidence. However, we assumed that percentages of trees recording fire years were representative of the relative spatial scales of burning for those years (Swetnam and Baisan 2003).

Fire-return intervals were analyzed statistically in different subcategories or filters. First, the statistical distribution of fire intervals using all fire years, even those represented by a single scar, was assessed. Then only those fire years were included in which 10% or more, and 25% or more of the recording samples were scarred, respectively. In contrast with the all-scarred category, these filters captured fires that were relatively large in size or more intense than those fires that scarred only one or a few trees (Swetnam and Baisan 1996). The statistical analysis of fire-return intervals included the minimum, maximum, and mean fire return intervals (MFI) and the Weibull median probability interval (WMPI), used to model asymmetric fire interval distributions and to express fire-return intervals in probabilistic terms (Grissino-Mayer 1995). After assessing each study site individually, all the data were combined to produce a master fire chronology for the region using a 10% filter for scarred samples. A 10% filter was chosen rather than a 25% filter based on the limited sample size of fires that scarred 25% of the samples.

### Forest Age Structure

Six hundred 0.03-ha permanent field plots (200 at each site) were established using a systematic sampling grid to stratify sample plots at 250-m intervals at BIBE and at 600-m intervals across DMTNC and MCPA at the intersection of grid lines. The center of each plot was marked with rebar and the location of the plot center was geo-referenced using a global positioning system (GPS). Age data for all trees is presented in this chapter and species composition and size data are presented in chapter 4.

Stand age was determined by coring a subset (2-3 per size-class) of trees in each plot using an increment borer ( $n = 1387$ ). *Pinus cembroides*, *P. strobiformis*, *P. ponderosa*, and *Pseudotsuga menziesii* were the only species cored, because the majority of tree species in the region do not produce annual growth rings (Cook et al. 1999, Floyd et al. 2000). Three seedlings (< 5 cm dbh) were destructively sampled just outside the perimeter of each plot. Cross-sections were taken of each seedling at the base and at 30 cm high to determine their age and to correct for the number of years lost by coring at of 30 cm. Tree cores and seedling cross-sections were aged by sanding them to a high polish and visually cross-dating them under a binocular microscope using standard dendrochronological techniques (Stokes and Smiley 1968). Additional years to the center were estimated with a pith locator (concentric circles matched to the curvature and density for the inner rings) for cores that missed the pith (Appelquist 1958).

## Results

### Fire Regime Characteristics

Two hundred twenty four of the 250 fire-scar samples we collected were cross-dated and used for analysis. The remainder of the samples could not be cross-dated; and were therefore eliminated from the study. Of the samples collected, 80% were sampled from dead trees (mostly downed logs). A total of 556 fire scars were dated, and the season of fire occurrence was estimated on 70% of the scars. The seasonality of the remaining scars was not able to be determined due to damage or the small size of the annual growth ring. Dormant season scarring (i.e. spring burning) was the most common (90%), followed by early-earlywood scarring (8%). Fewer than 2% of the burns were

middle earlywood or late earlywood. No scars were found in the early latewood or latewood.

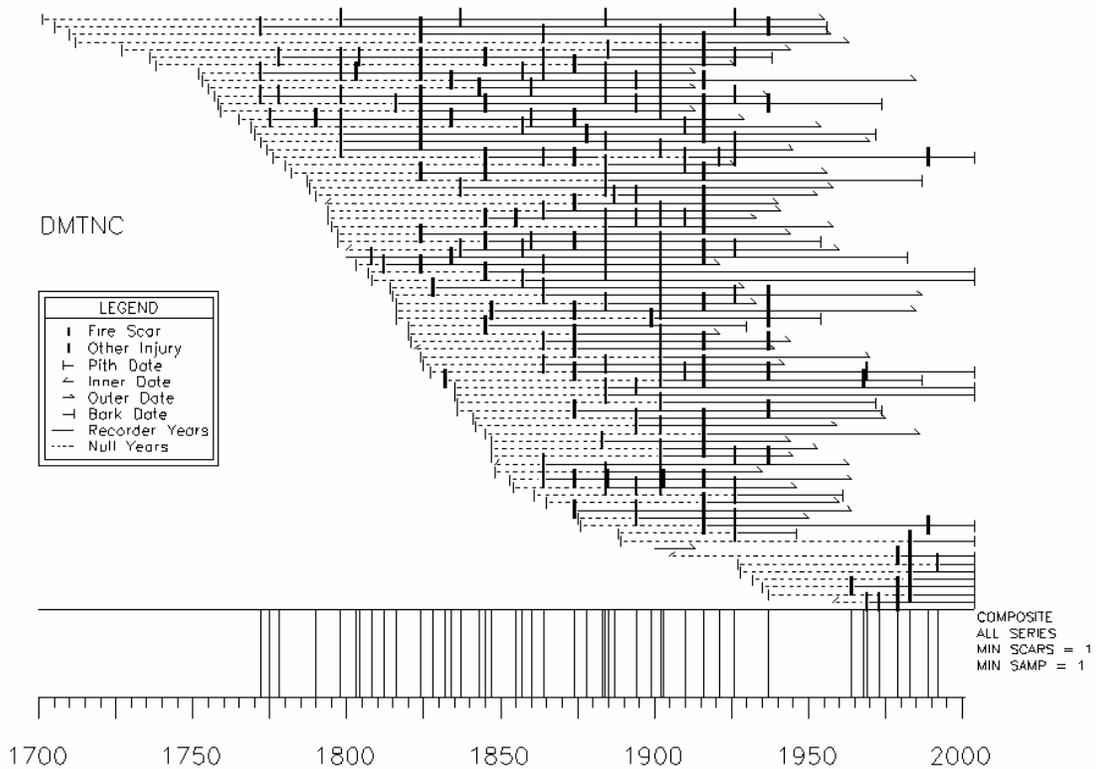
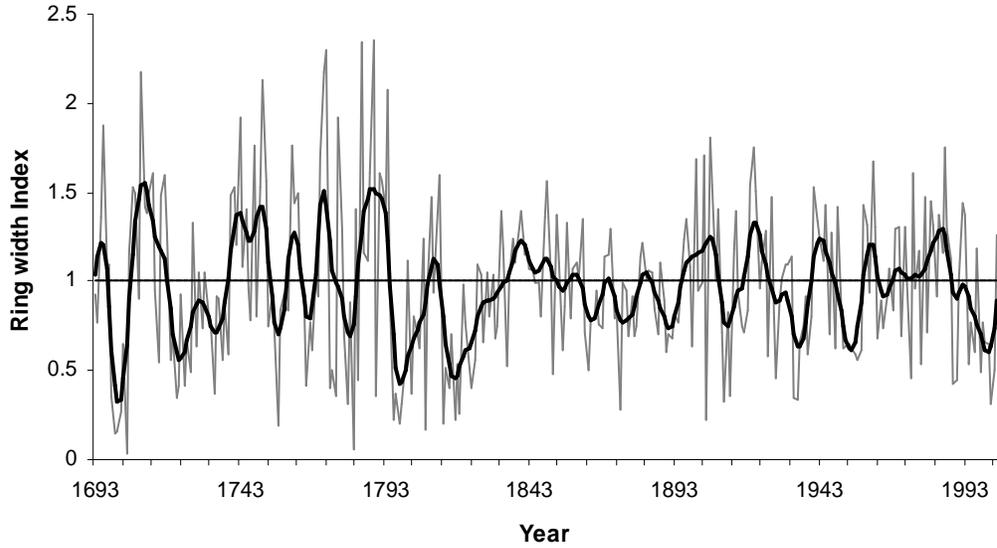
Fires were frequent at all three sites from 1772 (the onset of sufficient sample depth) (Table 1, Figs. 3-5) until 1913 in BIBE, 1926 in DMTNC, and 1951 in MCPA. Fires recurred at mean intervals of 4.4-9.0 years, using all scars at all sites. Point fire return intervals prior to fire exclusion ranged from 1-27 years in DMTNC, 2-19 years in BIBE, and 1-24 years in MCPA. The filtered data suggested that fires were predominantly small in size since most fires did not scar > 25% of the samples in each site. Mean fire return intervals for the filtered fire data using the 10% scarred criterion, averaged 1.5 to 2 times longer than for all fire scars, and were considered a conservative estimate of mean fire return interval (MFI) (Baker and Ehle 2001). The Weibull median probability interval (WMPI) values were very similar to MFI values, with all values being within a 1.0-year difference.

The historical pattern of frequent fire ceased at different times for each site. Major fires ended in 1937 in DMTNC and BIBE, but continued until 1951 in MCPA. Infrequent, smaller fires occurred in the three sites during the rest of the 20<sup>th</sup> century, mostly at MCPA. Twentieth century MFIs doubled, demonstrating a general cessation of fire (Table 3).

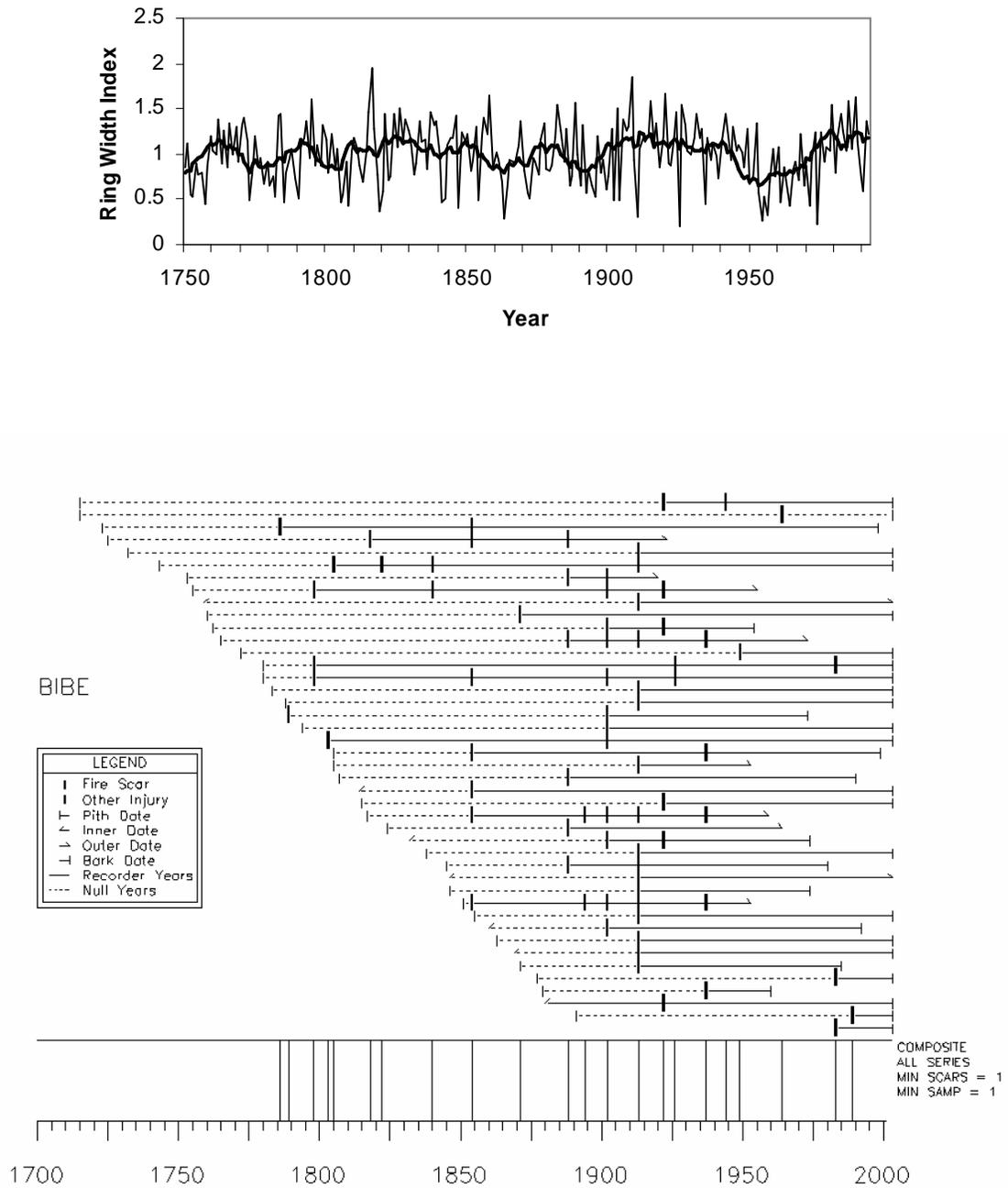
**Table 1. Fire intervals (in years) at DMTNC, BIBE, and MCPA. Analysis was carried out from the first fire date with a depth of recording samples of 10% of total sample size until the final data. All three sites recorded fires in 1937; thereafter only MCPA recorded fires. Statistical analysis was carried out in three categories: (1) all fire years, including those represented by a single fire year; (2) fire years in which 10% or more of the recording sample trees were scarred; and (3) fire years in which 25% or more of the recording sample trees were scarred. Abbreviations correspond to: MFI = mean fire interval, and WPMI = Weibull median probability interval. WPMI values are not shown (—) where the Weibull model did not fit the fire interval data adequately (Kolmogorov-Smirnov test  $\alpha = 0.05$ ).**

Site	Analysis period	Category of analysis	No. of intervals	MFI	Min.	Max.	Average per-sample interval	WPMI	Interval since last fire
TNC	1772-2004	All scars	44	5.0	1.0	27.0	24.4	4.2	12
		10% scarred	24	8.0	1.0	42.0		7.6	
		25% scarred	11	19.8	3.0	67.0		15.6	
TNC	1772-1926	All scars	29	4.4	1.0	12.0		4.0	
		10% scarred	17	7.2	1.0	16.0		5.6	
		25% scarred	8	14.5	3.0	26.0		12.6	
TNC	1926-2004	All scars	9	7.3	1.0	27.0		5.5	
		10% scarred	4	15.8	4.0	42.0		11.8	
		25% scarred	0	—	—	—		—	
BIBE	1786-2003	All scars	21	9.0	2.0	19.0	15.3	9.1	5
		10% scarred	15	13.1	2.0	46.0		10.6	
		25% scarred	7	14.0	3.0	41.0		15.5	
BIBE	1786-1900	All scars	11	9.8	2.0	18.0		8.9	
		10% scarred	10	7.5	2.0	34.0		8.8	
		25% scarred	5	14.0	3.0	42.0		16.7	
BIBE	1900-2003	All scars	9	9.7	4.0	19.0		9.3	
		10% scarred	4	20.3	9.0	46.0		17.8	
		25% scarred	0	—	—	—		—	
MCPA	1715-2003	All scars	62	4.4	1.0	24.0	24.7	—	14
		10% scarred	33	7.0	2.0	24.0		6.3	
		25% scarred	14	16.6	5.0	34.0		15.6	
MCPA	1715-1900	All scars	35	4.0	1.0	24.0		4.3	
		10% scarred	24	7.7	2.0	24.0		6.9	
		25% scarred	12	14.1	5.0	28.0		13.5	
MCPA	1900-2003	All scars	27	3.0	1.0	10.0		3.2	
		10% scarred	9	5.2	2.0	11.0		3.9	
		25% scarred	0	—	—	—		—	
All sites	1715-2003	All scars	99	2.0	1.0	24.0	21.5	—	14
		10% scarred	28	8.0	1.0	24.0		7.2	
		25% scarred	10	18.7	2.0	40.0		16.9	
All sites	1715-1900	All scars	63	2.0	1.0	24.0		—	
		10% scarred	22	8.1	1.0	24.0		6.8	
		25% scarred	9	20.0	2.0	40.0		16.7	
All sites	1900-2003	All scars	36	2.6	—	—		2.4	
		10% scarred	5	9.0	3.0	11.0		9.1	
		25% scarred	0	—	—	—		—	

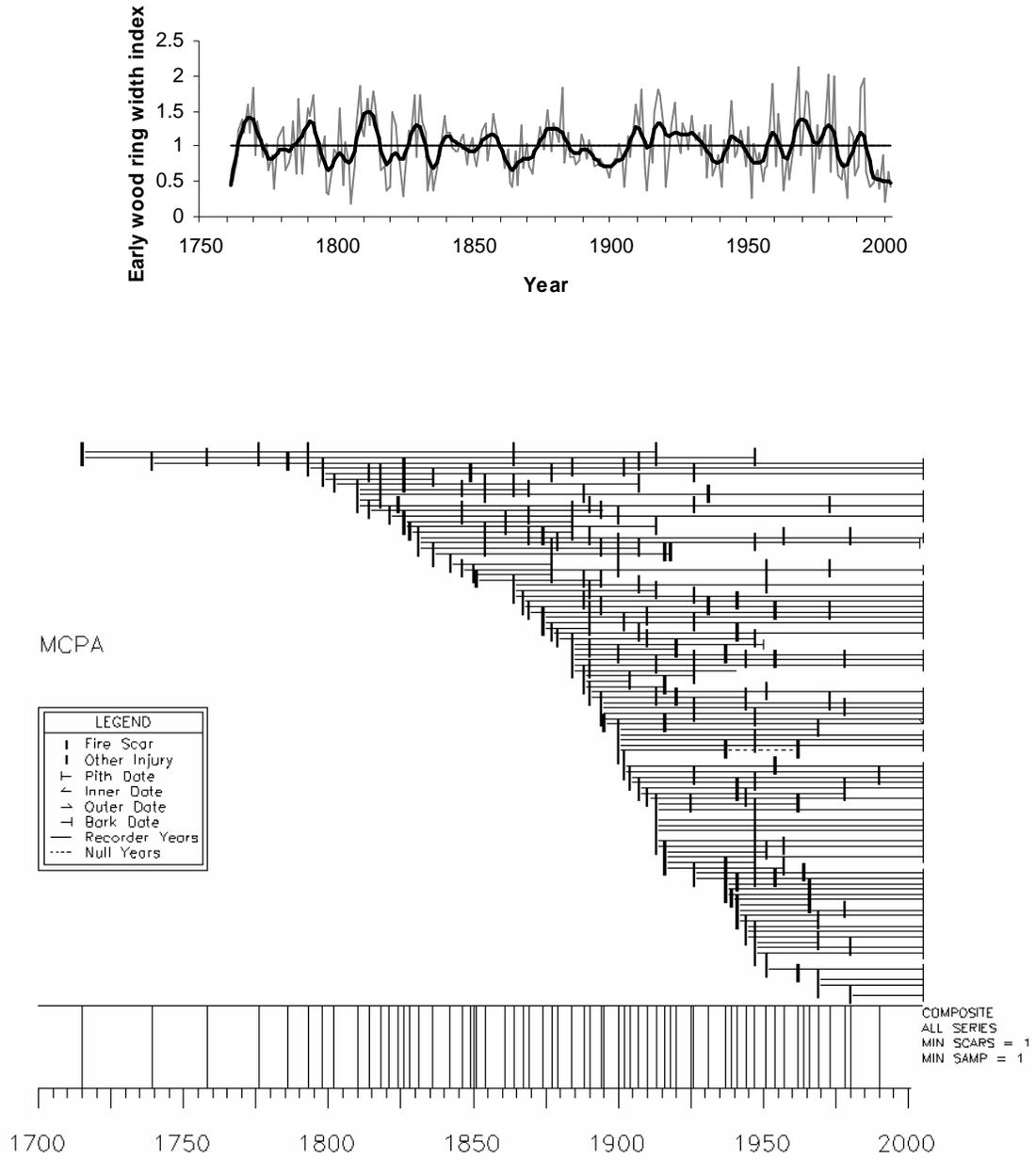
**Figure 3: Master chronology of fire events for 10% or more samples scarred and corresponding ring width index of climatic variation for DMTNC.**



**Figure 4: Master chronology of fire events for 10% or more samples scarred and corresponding ring width index of climatic variation for BIBE.**



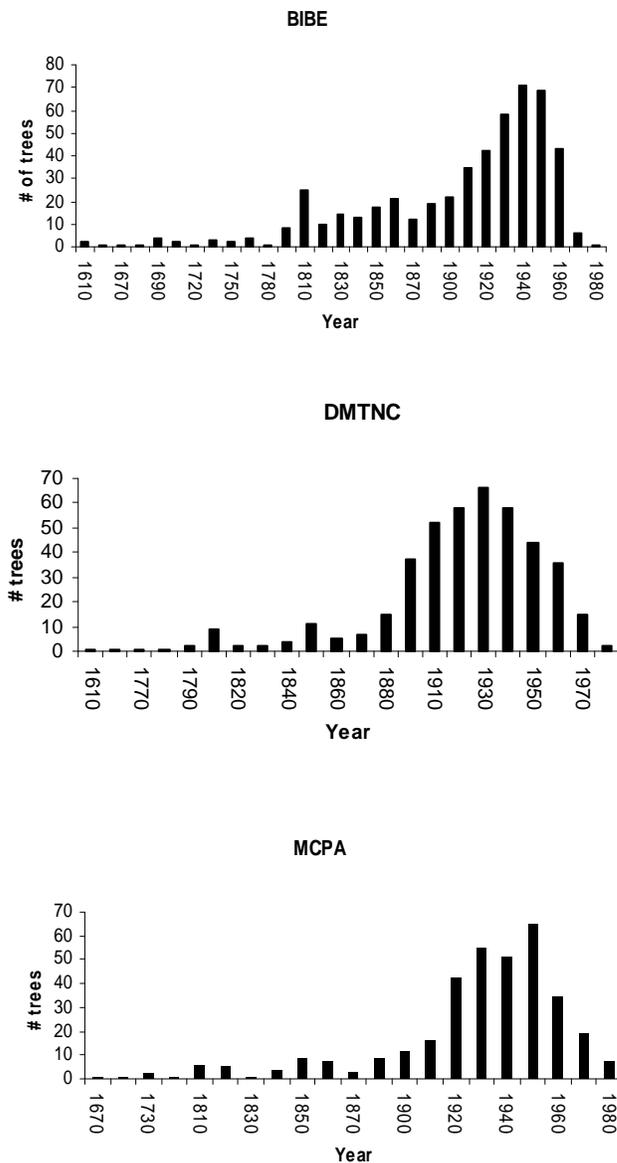
**Figure 5: Master chronology of fire events for 10% or more samples scarred and corresponding ring width index of climatic variation for MCPA.**



## Tree Age Structure

Tree age structures in the three sites showed regeneration pulses following cessation of the historical pattern of frequent fire (Fig. 6). The timing of recruitment pulses differed by site and corresponded with the last major fire event and settlement dates for each area. The regeneration pulse in MCPA occurred later than in the other two sites, in congruence with the later disruption of the historical fire regime. The recruitment pulse in BIBE followed a fire in 1913, while tree regeneration in DMTNC closely corresponded to the introduction of sheep to the Fort Davis area in the 1930s and their subsequent removal of the fine fuels necessary for fire ignition and spread across the landscape.

**Figure 6: Innermost ring dates of mature conifer trees in BIBE ( $n = 508$ ), DMTNC ( $n = 429$ ), and MCPA ( $n = 450$ ).**



## Discussion

Fire seasonality at all three sites was more typical of Mexico than other parts of the US Southwest. Fires in DMTNC, BIBE, and MCPA occurred predominantly during the dormant season, or in the spring prior to the onset of the North American Monsoon System. Most historical fires in the southwestern United States have been recorded in the early wood, probably as a result of lightning ignitions during summer monsoon rainstorms (Swetnam et al. 2001). In contrast, most modern fires in the Sierra Madre Occidental and Oriental in Mexico burn during the dry spring when lightning is most common (Mosiño Alemán and Garcia 1974, Turman and Edgar 1982, SEMARNAP 2000). This pattern of historical early spring has been reported for the Mexican portion of the Sierra Madre Occidental by Heyerdahl and Alvarado (2003) and Fulé et al. (2005), and in the southern Sierra Madre Oriental by Fule and Covington (1997).

The location of our sites south of Arizona and New Mexico may influence the strength of the NAMS and the seasonality of cloud-to-ground lightning in the Trans-Pecos. A phase shift in climate is known to occur between the Southwest and Mexico, though the exact location of this shift is not known (Stahle et al. 1998). The fact that our study area is located several hundred miles to the southeast of the center of the NAMS over Arizona suggests that the Chihuahuan Desert Borderlands may experience climatic conditions that are much more similar to Mexico than to the rest of the United States. Future studies that examine the contemporary spatial and temporal variability of lightning across the southwestern United States and northern Mexico would provide information about the mechanisms behind the observed differences in fire seasonality.

The small, frequent, low-intensity fires that historically impacted the forests of the Chihuahuan Desert Borderlands were similar to other reported historical fire regimes in southwestern North America. Historical MFIs were similar in our study sites to other regions of the Southwest, which ranged from 3.85-14.33 years for fires that scarred > 10% of the samples (Kaye and Swetnam 1998, Swetnam and Baisan 1996, Fulé et al. 2002). Like elsewhere in the Southwest and northern Mexico, our sites probably experienced fire exclusion as a consequence of the introduction of large numbers of livestock that preferentially grazed the fine fuels necessary for fire ignition and spread. However, the date of onset and the historical factors responsible for the cessation of fire in our study areas differed from elsewhere. Much of the Southwest on the US side of the border was settled by Euro-Americans around 1880 after the successful extirpation of Native American populations (Bahre 1991, Welsh 1996). The completion of the Texas and Pacific railroads brought the “cattle boom” to the southwest in the late 1800s. This era was characterized by tremendously high stocking rates ranging from 4.1-12.3 acres/animal unit in west Texas, compared to present day stocking recommendations of 75-200+ acres/animal unit (Downie 1978, Clayton 1993). This huge infiltration of cattle to the Southwest generally coincided with the subsequent reduction in the amount and continuity of fine fuels from grasslands and forest understories (Leopold 1924), and the removal of fire throughout the majority of Arizona and New Mexico in the late 1800s (Cooper 1960, Swetnam and Baisan 1996).

At our sites, the introduction of cattle did not seem to change the fuel bed characteristics sufficiently to incur a drop in fire frequency. Instead, the introduction of sheep and goats to the region in the 1920s and 1930s coincided with and may have been responsible for the onset of fire exclusion. Drought and subsequent cattle starvation

caused major losses of livestock between 1885 and 1895 in the west Texas, leaving ranchers looking for hardier livestock that could tolerate the harsh conditions (Carlson 1982). Sheep and goats were subsequently introduced in large numbers to the region in the early 1900s and local residents in the Big Bend Region recall high numbers of sheep in forest uplands during the 1930s and 1940s (William Dodson, personal communication). Grazing habits of sheep and goats are more damaging to grasses than those of cattle, and this was likely a key factor in the drop in fire frequency in DMTNC and BIBE since the early 1900s.

Fire exclusion dates in MCPA in the mid-20<sup>th</sup> century corresponded to other fire-suppressed sites in Mexico (Heyerdahl and Alvarado 2003, Matthews 2003, Fulé et al. 2005). Fire exclusion dates ranged from the 1930s to the 1950s in most sites in the Sierra Madres, though it occurred somewhat later in the 1970s in the Sierra San Pedro Martir in Baja California (Stephens 2003), and did not occur at all in several locations of the La Michilía Protected Area in the southern Sierra Madre Oriental (Fulé and Covington 1996). The 1940s and 1950s corresponded to the redistribution of lands to ejidos by Lázaro Cárdenas between 1934 and 1940 (López and Bernardino Mata 1992). The formation of the ejidos, literally the redistribution of lands to community landowner groups, caused the overexploitation of the majority of arid Mexico because ejidos were generally given unproductive lands located in remote regions. People from lowlands were relocated to uplands of the Sierra Madres Oriental and Occidental by ejido formation. Their relocation resulted in the introduction of livestock to mountainous regions of Mexico that were probably not utilized previously for grazing. Increases in livestock (especially in goats and sheep) in MCPA as a result of ejido occupation probably played a major role in the cessation of fire in the middle of the 20<sup>th</sup> century.

Commercial logging was prevalent between 1950 and 1970 in MCPA, and may have also played a role in fire exclusion by changing the live and dead fuel bed characteristics (SEMARNAP 1997). Prior to the formation of the ejidos, roads were limited to a few small mining roads in the lower elevations of MCPA. Logging during the mid-to-late 1900s resulted in an extensive road system that still scars the landscape today. We speculate that roading in MCPA disrupted fuel bed continuity and limited fire spread throughout the mountain range. The synergistic effect of intense livestock grazing due to the formation of the ejidos and logging was most likely responsible for the exclusion of frequent, low intensity fire in MCPA after 1953.

A key finding in our study was that pinyon pine forests experienced frequent, low intensity fire. The fire scar samples from BIBE and DMTNC were predominantly from pinyon pines, a species not thought to readily produce fire scars because of its intolerance to fire (Floyd et al. 2000, Floyd et al. 2004). Our pinyon pine fire scar samples contained between 1 and 4 fire scars each, suggesting that the pinyon pines in our study area were able to survive multiple fire events. Baker and Shinneman (2004) suggested that the use of frequent, low intensity prescribed fire in pinyon-juniper woodlands of the West is misguided, a contention based solely on the lack of actual fire history data that quantifies the historical range of variability of fires in these forests. Floyd et al. (2000 and 2004) used stand age data from pinyon pine to quantify the fire history of this species in Mesa Verde, Colorado. Their results suggested that fires in pinyon pine woodlands were infrequent; with fire return intervals between 200-400 years. Absence of fire scar data from Floyd et al. (2000 and 2004) may explain their high fire return intervals, but that

they were working in a mesa system, rather than in a topographically dissected landscape, may also explain the dramatic differences between their results and ours.

Our results represent the second pinyon pine fire history study based on dendrochronologically dated fire scars in the western United States. Our study suggests that fires occurred much more frequently in this forest type than believed by Baker and Shinneman (2004) and Floyd et al. (2000 and 2004), and are similar to Brown et al. (2001) who also documented frequent historical fires in pinyon-juniper woodlands in the nearby Sacramento Mountains in New Mexico. These findings indicate that pinyon-juniper woodlands in the southeastern portion of the Southwest probably sustained frequent, low intensity fires similar to other vegetation types of the Southwest.

The age structure of the trees in our study sites supported the contention that fire exclusion resulted in extensive tree recruitment. Similar tree regeneration responses have been observed throughout the southwestern United States and parts of Mexico in response to the disruption of frequent fire return intervals as a result of fire exclusion from grazing and direct fire suppression (Savage 1994, Savage 1991, Swetnam and Betancourt 1998, Fulé and Covington 1997, Fulé et al. 2002).

## Chapter 3 Climate as a Fire-Forcing Mechanism on Fire

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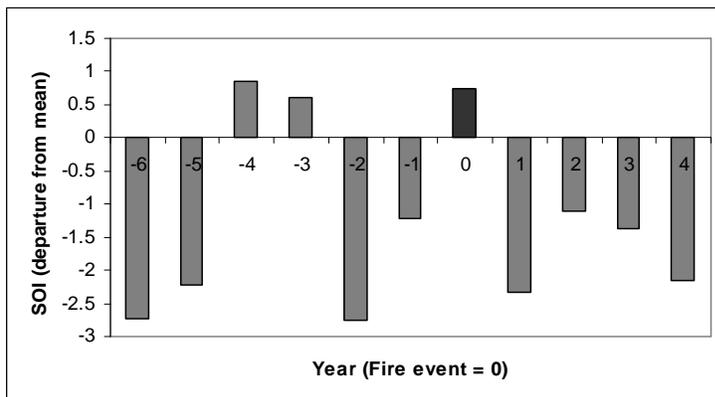
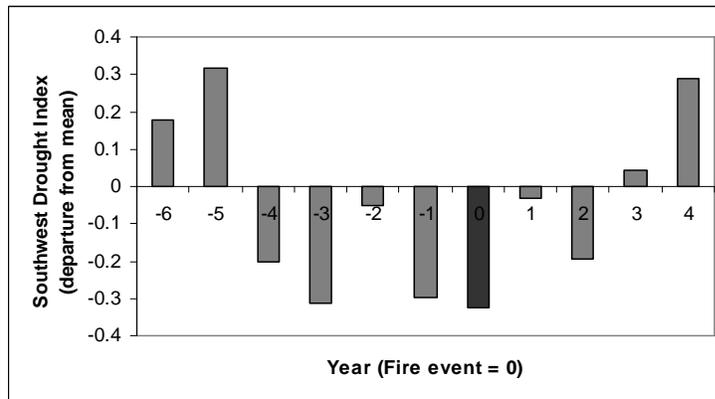
We determined the strength of the relationship between climatic fluctuations and fire occurrence using superposed epoch analysis (SEA: Swetnam and Baisan 1993) with FHX2 (Grissino-Mayer 2001). At each site, we used locally developed tree-ring chronologies as proxies for climate. SEA was used to compare average annual climate conditions for the set of fire years recorded on 10 % or more of the fire-scar samples. SEA uses a moving window to compare climatic conditions 6 years prior to and after a fire event. Significant climate conditions were determined using bootstrapped confidence intervals based on average annual climate values with the same number of climate years as the fire year datasets. A principal components analysis of the tree-ring chronologies explained 67% of the winter-spring (October-May) precipitation variance of the climatic station in the Chisos Basin in BIBE from 1944-2002, which is centrally located relative to the other two sites.

The regional-scale relationship between climatic fluctuations and fire occurrence was evaluated by performing an SEA of the master fire chronology (10% scarred filter) with independently derived tree-ring reconstructions of the Southern Oscillation Index (SOI; Stahle et al. 1998) and the Southwestern USA Drought Index Reconstruction (SWDI) (Cook 2000). SOI is a commonly used measure of El Niño Southern Oscillation (ENSO), and is the difference in surface air pressure between Darwin, Australia and Tahiti. The SOI is measured as the difference in sea surface temperatures between Tahiti and Darwin so that an extremely negative (near  $-1$ ) SOI constitutes an ENSO year (wet year in southwestern North America), and a positive (near  $+1$ ) SOI is associated with a La Niña event (dry year in southwestern North America) (Díaz and Kiladis 1992).

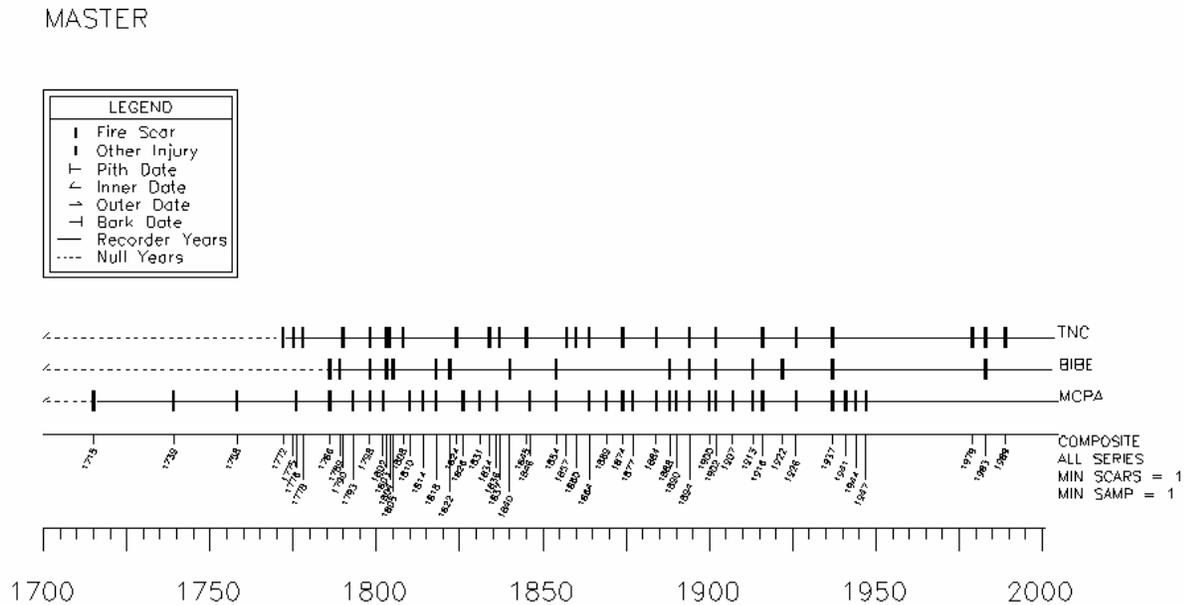
### Results

Large fires (fires that scarred  $> 10\%$  of the samples) burned across multiple sites during significantly dry years (outside 95% bootstrapped confidence intervals) according to the Southwestern Drought Index (SWDI) and the Southern Oscillation Index (SOI) (Fig. 7). Years preceding fire events were wet, though our results did not show a significant trend for pre-fire climate years. Eleven fire years coincided with positive extremes of the SOI (La Niña years), as reconstructed from tree rings in the Sierra Madre Occidental (Stahle et al. 1998), indicating that fires burned during cool, dry conditions.

**Figure 7: Superposed epoch analyses of ENSO anomalies and hydroclimate for fire years scarring > 10% of fire-scar samples in the master chronology of the three sites for (a) the Southwestern Drought Index (SWDI) and (b) the Southern oscillation index (SOI) (Stahle et al. 1998). SOI is reversed in terms of moisture relative to other proxies (dry years are high SOI). Dark gray bars mark significant departures ( $P < 0.05$ ) above and below the mean based on bootstrapped confidence intervals.**



**Figure 8: Compiled master chronology of fire events for the three study sites including fires that scarred 10% of the samples in each location.**



Unique fire dates were assessed by comparing fire events between sites (Fig 8). Of the 55 fire dates, 27 fires were unique to a particular site, with 16 unique fires at MCPA, 7 at TNC, and 4 at BIBE. Nineteen of the fires (20%) occurred at 2 sites, and 5 (9%) occurred at all three sites. The synchronous fire years across the three study areas (1798, 1802, 1884, 1902, and 1937) corresponded to extreme positives of the SOI.

### Discussion

A pattern of wet years preceding fire events, followed by dry fire years is typical for southwestern North America (Swetnam and Betancourt 1998, Heyerdahl and Alvarado 2003, Fulé et al. 2005, Brown and Wu 2005), elsewhere in the western United States (Veblen et al. 2000, Donegan et al. 2001, Heyerdahl et al. 2002, Norman and Taylor 2003, Floyd et al. 2004) and south America (Kitzberger et al. 1997, Veblen et al. 2000). While our data don't completely support this pattern (wet years were not statistically significant), neither do they refute it. Continental-scale teleconnections are known to drive climatic variation at the regional scale throughout the United States and Mexico, and they can potentially explain the link between regionally synchronous fire

events and climate anomalies like the El Niño Southern Oscillation (ENSO) (Stahle and Cleaveland 1993, Lough 1992, Meko 1992, Cleaveland et al. 1992).

The current year's climate affects fuel moisture and fuel availability, while a combination of wet years followed by dry years can lead to regionally synchronous fires (Swetnam and Betancourt 1998). Fine fuels such as understory grasses are key components of fire ignition and spread; they are known to be particularly sensitive to seasonal and annual variation in moisture availability in the Southwest (Cable 1975, McClaran and VanDevender 1995). Winter precipitation probably affects fire by influencing soil moisture and the correspondent growth of live surface fuels. Subsequent dry years typical of the positive phase of SOI (La Niña year) lower the moisture of fine fuels and larger woody debris, rendering these forests highly susceptible to regionally synchronous fire events. Persistent drought in the Sierra Madres Oriental and Occidental has been linked to La Niña episodes. The tropical rainfall index correlates well with instrumental and reconstructed winter precipitation, suggesting strong ENSO modulation of cool season climate over the Southwest and northern Mexico during La Niña years (Cleaveland et al. 2003).

While the process of fine fuel accumulation during El Niño years and the subsequent drying of fine and coarse woody fuels in La Niña years is considered the major mechanism controlling large-scale climatic forcing of fire events (i.e. Swetnam and Betancourt 1998, Heyerdahl and Alvarado 2003, Fulé et al. 2005, Brown and Wu 2005, Veblen et al. 2000, Donegan et al. 2001, Heyerdahl et al. 2002, Norman and Taylor 2003, Floyd et al. 2004, Kitzberger et al. 1997, Veblen et al. 2000), no quantitative comparisons exist to determine whether fine fuel accumulation and fuel moisture during El Niño/La Niña years differ dramatically from mean climatic conditions. Nor is there any information that quantifies potential variation in cloud-to-ground lightning frequency during El Niño/La Niña cycles, which could also explain why more fires are observed during these climate anomalies. Future research that investigates the mechanisms that drive the climate forcing of fire events could lead to causational, rather than correlational, explanations of the fire-climate relationship.

## **Chapter 4 Vegetation Types, Forest Size Structure, and Species-Environment relationships in the Chihuahuan Desert Borderlands**

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We quantified the dominant vegetation types and the size-structure of vegetation in the three study sites using nested, circular, fixed area plots. Trees  $\geq 5$  cm diameter at breast height (dbh) were measured in 10 m radius plots, and the species, dbh, total height, live crown height, and the live crown ratio were recorded for each tree. Seedlings were tallied by species on 5 m radius plots. Percent cover of grass, shrubs, and bare soil were recorded in six cover classes (<1 %, 2-5%, 6-25%, 26-50%, 51-75%, and 76-100%). Plot areas were corrected for slope. Species importance values were calculated for each tree species within each plot as the sum of the relative density and the relative basal area (BA) of each species (0-200 range). Forest size structure was quantified using size, height, live crown height, and live crown ratio measurements. Tree density ( $\text{ha}^{-1}$ ) was calculated in 5 cm size-classes for each species in each plot as a measure of stand structure.

Plots with similar woody plant composition were identified using cluster analysis. We clustered species' importance values using Ward's method, and relative Euclidean distance as the similarity measure with PC-Ord software (McCune and Mefford 1999). Differences in species composition in each group were determined using indicator species analysis (Dufrêne and Legendre 1997).

The strength of species-environment relationships were analyzed using non-metric multidimensional scaling ordination techniques (Kruskal and Wish 1978) and PC-Ord software (McCune and Mefford 1999). The potential contribution of site in explaining variation in species abundance was identified by correlating (Pearson product moment) NMDS axis scores with environmental variables. Differences in species importance values and environmental variables (Table 2) among groups were compared using a multiple response permutation procedure (MRPP).

**Table 2: Environmental variables used for quantifying tree species-environment and fuels-environment relationships using non-metric multidimensional scaling.**

Variable code	Definition
<b>Landscape Metrics</b>	
elevation	elevation (m)
N aspect	Cosine transformation of aspect (degrees) (Beers et al. 1966) 1.0 (southwest) to -1.0 northeast
E aspect	Sine transformation of aspect (degrees) (Beers et al. 1966) 1.0 (southwest) to -1.0 northeast
slope	lope (degrees)
ISF	incident sky factor calculated from hemispherical photographs using Hemiview Canopy analysis software, Version 2.1
DSF	diffuse sky factor calculated from hemispherical photographs using Hemiview Canopy analysis software, Version 2.1
TRMI	topographic relative moisture index calculated as the slope aspect and pitch, slope position, (mesic) configuration that ranges from 0 (xeric) to 60 Parker (1982)
PRR	cumulative potential relative radiation based on hourly solar position, topography and topographic shading (Pierce et al. 2005)
topopos 150	topographic position, calculated as the difference between a cell's elevation and the mean elevation of cells within a 150 m radius
topopos 450	topographic position, calculated as the difference between a cell's elevation and the mean elevation of cells within a 450 m radius
topo configuration	topographic configuration ranging from concave to convex calculated using the spatial analyst function in ArcMap 9.1
Landform	landform type derived from Terrain Analysis System software (Lindsay 2005) based on Pennock et al. (1987). Landform types include (1) convergent footslope, (2) divergent footslope, (3) convergent shoulder, (4) divergent shoulder, (5) convergent backslope, (6) divergent backslope, and (7) level
sediment transport	sediment transport capacity index = $(As/22.13)^{0.6} \times (\sin S/0.0896)^{1.3}$ where As = the specific catchment area and S = the local slope
wetness index	wetness index (Beven and Kirkby 1979) derived from Terrain Analysis System software (Lindsay 2005) defined as $WI = \ln(As/\tan S)$ .
network index	minimum wetness index value encountered along a flowpath (Lane et al. 2004). This value defines when a cell with a zero or negative saturation deficit is connected to the drainage network
flow direction	flow direction from ArcHydro extension in ArcMap 9.1 and 30-m DEM
flow accumulation	flow accumulation from ArcHydro extension in ArcMap 9.1 and 30-m DEM

## Results

Our data suggested nine dominant forest vegetation types across the three study sites: gray oak, gallery forest, graves oak, emory oak, pinyon pine, oak-pinyon-juniper, ponderosa-sw white pine, Arizona cypress-fir, and alligator juniper. Species composition differed significantly by vegetation type, and the names of the vegetation cover types were assigned using the names of species with high indicator values from indicator species analysis.

The vegetation types varied significantly by species composition and structure (MRPP  $P < 0.001$ ) (Tables 3 and 4). The spatial distribution of the vegetation types varied by environment (Tables 5 and 6, Fig. 9). DMTNC had only four vegetation types: pinyon pine, oak-pinyon-juniper, alligator juniper, and ponderosa-sw white pine forest. BIBE had six major vegetation types with several oak woodland types not found elsewhere in the study area. These included: gray oak, Graves oak, Emory oak, pinyon pine, oak-pinyon-juniper, and alligator juniper. The largest variety of vegetation types occurred in MCPA: gray oak, gallery, pinyon pine, oak-pinyon-juniper, alligator juniper, ponderosa-sw white pine, and Arizona cypress-fir vegetation types. The Arizona cypress-fir vegetation type was unique to MCPA.

**Table 3: Descriptions of dominant species and environmental characteristics for forested vegetation types in the study area.**

Vegetation Type	Dominant Species	Environmental Setting
Gray oak	<i>Quercus grisea</i> , <i>Q. arizonica</i> , <i>Pinus cembroides</i>	Dry sites Middle elevations Southerly aspects
Gallery forest	<i>Q. laceyi</i> , <i>Q. gravesii</i> , <i>Q. gracilliformis</i> , <i>Q. mohrinana</i> , <i>Acer grandidentata</i> , <i>Tilia Americana</i> ,	Low to middle elevations Mesic sites Valley bottoms Low incident solar radiation
Graves oak	<i>Q. gravesii</i> , <i>Arbutus xalapensis</i> , <i>Acer grandidentata</i> , <i>Juniperus flaccida</i>	Middle elevations Mesic sites Valley bottoms Low incident solar radiation
Emory oak	<i>Q. emoryi</i>	Low elevations Mesic sites Valley bottoms Low incident solar radiation Gentle slopes
Pinyon pine	<i>P. cembroides</i> , <i>Q. grisea</i>	Middle elevations Dry sites Convex slopes High incident solar radiation Upper topographic positions
Oak-pinyon-juniper	<i>Q. grisea</i> , <i>P. cembroides</i> , <i>J. deppeana</i>	Middle elevations Dry sites Convex slopes High incident solar radiation Middle topographic positions
Ponderosa-sw white pine	<i>P. ponderosa</i> , <i>P. strobiformis</i> , <i>Q. sideroxyla</i>	High elevations Mesic sites Low incident solar radiation
Arizona cypress-fir	<i>Cupressus arizonica</i> , <i>Abies coahuilensis</i> , <i>Pseudotsuga menziesii</i> , <i>P. strobiformis</i>	High elevations Most mesic sites Lowest incident solar radiation
Alligator juniper	<i>J. deppeana</i>	Middle elevations Gentle slopes High incident solar radiation High grass cover

Within a vegetation type, overall species composition differed among sites. Gray oak, pinyon pine, and oak-pinyon-juniper forests in BIBE contained a juniper complex (*Juniperus deppeana*, *J. flaccida*, and *J. pinchotii*) not found in the other two sites. While these juniper species also occurred to some extent in MCPA, *J. flaccida* and *J. pinchotii* primarily occurred at BIBE. The pinyon pine, oak-pinyon-juniper, and alligator juniper forests in MCPA had a greater incidence of oak species than did the other two sites. In MCPA, these forest types contained mixtures of *Quercus grisea*, *Q. arizonica*, *Q. mohriana*, and *Q. laceyi*. The primary oak species in these vegetation types at other two sites was *Q. grisea*.

The distribution of vegetation was dependent on local environment (Table 5, Fig. 9). The ordination of species composition and environmental variables identified

elevation, soil moisture (TRMI), incident solar radiation, slope, and topographic position as major factors driving species composition and distribution. Grey oak forests were found on dry sites at middle elevations on southern aspects. Gallery, Graves, and Emory oak forests were all found on sites with low incident solar radiation (mesic valley bottoms), but each dominated different elevational gradients. Emory oak forests were found at low elevations and on less steep slopes than the other two valley bottom forest types, and had higher shrub cover. Pinyon pine and oak-pinyon-juniper forests were found at low to intermediate elevations on dry, convex slopes with high incident solar radiation. Pinyon pine and oak-pinyon-juniper forest types were differentiated by their distributions on upper (pinyon pine) versus middle (oak-pinyon-juniper) topographic positions and oak-pinyon-juniper forests had higher bare soil, grass and shrub cover. Ponderosa-sw white pine and Arizona cypress-fir forests existed at upper elevations on mesic sites with low incident solar radiation. The two forest types were differentiated by moisture and insolation, with Ponderosa-sw white pine forests occupying wetter sites, with higher incident and potential relative radiation (PRR) than Arizona-cypress fir. Finally, Alligator juniper forests were found on gentle slopes at middle elevations, and were characterized by high incident solar radiation and low flow accumulation. This forest type also had high grass cover mixed with patches of bare soil.

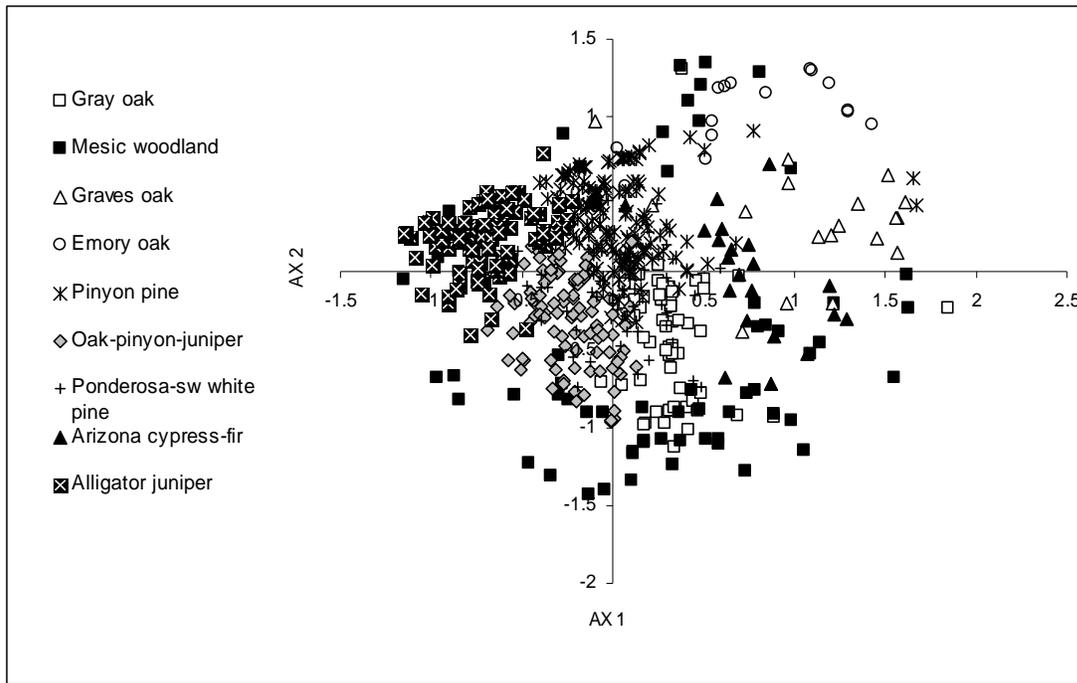
**Table 4: Basal area and density by species for each vegetation type in BIBE, MCPA, and DMTNC (n = 600).**

vegetation type	gray oak				gallery forest		graves oak		emory oak		pinyon pine					
	BIBE n = 38		MCPA n = 22		MCPA n = 58		BIBE n = 18		BIBE n = 20		BIBE n = 65		MCPA n = 14		DMTNC n = 68	
	BA	DENS	BA	DENS	BA	DENS	BA	DENS	BA	DENS	BA	DENS	BA	DENS	BA	DENS
<i>Abies coahuilensis</i>	0.0	0.0	0.0	0.0	0.0	1.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
<i>Acer grandidentata</i>	0.0	0.0	0.0	36.4	1.1	254.0	0.6	253.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
<i>Arbutus xalapensis</i>	0.0	4.2	0.5	1.4	0.2	16.9	0.9	48.4	0.0	13.3	0.1	10.8	0.1	10.3	0.0	1.9
<i>Calliandra conferta</i>	0.0	0.0	0.0	0.0	0.0	0.0	0.0	3.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
<i>Celtis laevigata</i>	0.0	20.1	0.0	309.1	0.0	28.7	0.0	15.0	0.0	4.2	0.0	0.0	0.0	0.0	0.0	0.0
<i>Cretagous traceyi</i>	0.0	0.0	0.0	18.2	0.0	79.2	0.0	25.0	0.0	0.0	0.0	0.0	0.0	4.5	0.0	0.0
<i>Cupressus arizonica</i>	0.3	5.0	0.1	5.8	1.1	30.2	4.5	39.8	0.0	0.0	0.0	2.4	0.2	4.3	0.0	0.0
<i>Alnus acuminata</i>	0.0	0.0	0.0	0.0	0.0	14.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
<i>Fraxinus cuspidata</i>	0.0	0.0	0.0	0.0	0.0	0.5	0.1	49.8	0.0	6.7	0.0	16.9	0.0	0.0	0.0	0.0
<i>Fraxinus gregii</i>	0.0	6.9	0.0	15.1	0.1	47.5	0.1	15.9	0.0	6.4	0.0	6.2	0.0	0.0	0.0	0.0
<i>Fraxinus velutina</i>	0.0	0.0	0.0	39.3	0.2	14.7	0.0	0.0	0.0	0.0	0.0	1.5	0.0	1.4	0.0	0.0
<i>Juniperus depepeana</i>	1.1	44.5	0.6	36.2	0.9	41.2	0.5	35.3	0.4	91.5	3.9	119.1	2.4	107.1	1.8	141.9
<i>Juniperus flaccida</i>	1.5	37.1	0.1	75.6	0.1	125.2	1.3	48.4	0.8	43.7	2.5	34.3	0.6	243.0	0.0	0.0
<i>Juniperus pinchotii</i>	0.5	10.9	0.4	26.7	0.0	0.0	0.0	0.0	0.2	8.8	0.1	3.9	0.0	4.5	0.0	0.0
<i>Ostrya virginiana</i>	0.0	0.0	0.0	0.0	0.1	10.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
<i>Pinus cembroides</i>	5.0	197.8	1.3	220.3	0.2	274.5	1.8	68.9	2.1	164.9	9.7	405.7	8.1	821.3	6.5	536.5
<i>Pinus ponderosa</i>	0.0	0.0	0.4	5.8	0.9	66.5	1.6	74.6	0.0	0.0	0.0	0.0	0.3	5.8	0.0	2.4
<i>Pinus strobiformis</i>	0.0	0.0	0.0	0.0	0.1	38.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
<i>Populus angustifolia</i>	0.0	15.8	0.0	0.0	0.1	1.1	0.0	0.0	0.0	0.0	0.0	4.6	0.0	0.0	0.0	0.0
<i>Populus fremontii</i>	0.0	0.0	0.0	0.0	0.1	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
<i>Prunus serotina</i>	0.0	0.0	0.0	13.6	0.0	51.4	0.0	14.6	0.0	20.0	0.0	1.0	0.0	0.0	0.0	0.0
<i>Pseudotsuga menziesii</i>	0.0	0.0	0.0	0.0	0.2	51.2	0.1	3.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
<i>Quercus arizonica</i>	0.0	157.9	0.9	143.6	1.6	153.2	0.0	73.0	0.0	66.7	0.0	136.9	0.2	29.6	0.0	0.0
<i>Quercus emoryi</i>	0.1	21.8	0.0	4.5	0.0	18.3	0.3	25.9	6.6	443.1	0.0	6.9	0.0	0.0	0.2	66.5
<i>Quercus gambelii</i>	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
<i>Quercus gracilliformis</i>	0.0	0.0	0.3	7.2	0.6	238.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.9
<i>Quercus gravesii</i>	0.3	25.1	1.0	179.2	2.1	356.0	10.9	789.6	9.3	67.6	1.0	55.4	0.0	20.9	0.0	0.0
<i>Quercus grisea</i>	6.4	470.2	8.3	549.8	1.6	136.1	2.0	72.1	0.5	32.1	3.8	219.0	1.7	166.0	4.3	302.3
<i>Quercus hypoleucoides</i>	0.0	0.0	0.1	10.3	0.2	73.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	7.4	0.0	8.2
<i>Quercus laceyi</i>	0.0	0.0	0.4	60.8	1.9	406.4	0.0	181.1	0.0	6.7	0.0	0.0	0.2	63.2	0.0	0.0
<i>Quercus mohriana</i>	0.0	0.0	0.4	42.8	2.4	195.3	0.0	5.0	0.0	20.0	0.0	0.0	0.4	51.7	0.0	0.0
<i>Quercus meuhlebergii</i>	0.0	0.0	0.0	0.0	0.1	20.5	0.3	1.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
<i>Quercus oblongifolia</i>	0.0	0.0	0.5	16.3	0.5	31.5	0.0	3.2	0.0	0.0	0.0	0.0	0.2	4.3	0.1	1.9
<i>Quercus pungens</i>	0.0	1.7	0.0	0.0	0.0	1.7	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0	0.0
<i>Quercus rugosa</i>	0.0	0.0	0.0	0.0	0.8	10.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	15.0	0.0	0.0
<i>Quercus shumardii</i>	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
<i>Quercus sideroxyla</i>	0.0	0.0	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0	18.5	0.0	0.0	0.0	3.7
<i>Quercus tardifolia</i>	0.0	0.0	0.0	22.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.4	0.0	0.0
<i>Quercus turbinella</i>	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.5	0.0	0.0	0.0	0.0
<i>Quercus vaseyana</i>	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
<i>Salix nigra</i>	0.0	0.0	0.0	0.0	0.0	1.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
<i>Stryax youngiae</i>	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
<i>Tilia americana</i>	0.0	0.0	0.0	9.1	0.8	28.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Total	15.2	1019.0	15.1	1849.5	18.8	2819.8	25.1	1846.6	20.0	995.7	21.2	1045.6	14.3	1562.0	13.0	1066.3

**Table 4: Continued.**

vegetation type	oak-pinyon-juniper						ponderosa-sw white pine				Arizona cypress-fir		alligator juniper					
	BIBE		MCPA		DMTNC		MCPA		DMTNC		MCPA		BIBE		MCPA		DMTNC	
	n = 14		n = 14		n = 78		n = 45		n = 14		n = 21	n = 13		n = 12		n = 72		
SPECIES	BA	DENS	BA	DENS	BA	DENS	BA	DENS	BA	DENS	BA	DENS	BA	DENS	BA	DENS	BA	DENS
<i>Abies coahuilensis</i>	0.0	0.0	0.0	0.0	0.0	0.0	0.5	212.4	0.0	0.0	2.6	427.0	0.0	0.0	0.0	0.0	0.0	0.0
<i>Acer grandidentata</i>	0.0	0.0	0.2	3.5	0.0	0.0	0.0	0.0	0.0	0.0	0.2	3.0	0.0	0.0	0.5	2.7	0.0	0.0
<i>Arbutus xalapensis</i>	0.1	0.0	0.7	28.3	0.0	11.4	0.1	96.4	0.1	133.0	0.0	126.8	0.0	0.0	0.0	11.0	0.0	0.4
<i>Calliandra conferta</i>	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
<i>Celtis laevigata</i>	0.0	0.0	0.0	0.0	0.0	0.0	0.0	8.9	0.0	0.0	0.0	14.3	0.0	0.0	0.0	16.7	0.0	0.0
<i>Cretagous traceyi</i>	0.0	0.0	0.0	0.0	0.0	0.0	0.0	124.4	0.0	0.0	0.0	590.5	0.0	0.0	0.0	0.0	0.0	0.0
<i>Cupressus arizonica</i>	0.0	0.0	0.0	3.5	0.0	0.0	1.2	63.7	0.0	0.0	14.0	465.6	0.0	0.0	0.0	2.7	0.0	0.0
<i>Alnus acuminata</i>	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
<i>Fraxinus cuspidata</i>	0.0	21.4	0.0	0.0	0.0	0.0	0.0	2.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
<i>Fraxinus gregii</i>	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
<i>Fraxinus velutina</i>	0.0	0.0	0.1	10.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.5	0.0	9.1	0.0	0.0	0.0	0.0
<i>Juniperus deppeana</i>	7.5	359.4	7.0	180.5	4.6	285.7	0.8	32.6	0.6	73.9	0.1	3.0	5.9	295.3	10.7	347.7	9.5	700.5
<i>Juniperus flaccida</i>	1.4	34.1	0.0	122.2	0.0	0.0	0.0	80.0	0.0	0.0	0.0	0.0	1.2	52.1	0.0	166.7	0.0	0.0
<i>Juniperus pinchotii</i>	0.1	9.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	23.2	0.0	0.0	0.0	0.0
<i>Ostrya virginiana</i>	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
<i>Pinus cembroides</i>	11.9	354.9	0.9	152.6	3.1	421.5	0.1	8.8	0.1	131.8	0.2	153.7	5.4	194.0	3.7	422.1	2.3	334.4
<i>Pinus ponderosa</i>	0.0	0.0	0.8	21.7	0.5	25.6	7.7	425.8	9.8	374.0	1.4	40.9	0.0	0.0	0.0	24.6	0.1	42.8
<i>Pinus strobiformis</i>	0.0	0.0	0.0	44.4	0.0	2.9	7.5	1169.9	2.7	341.1	3.8	258.8	0.0	0.0	0.0	0.0	0.0	0.0
<i>Populus angustifolia</i>	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	16.7	0.0	0.0
<i>Populus fremontii</i>	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
<i>Prunus serotina</i>	0.0	0.0	0.0	0.0	0.0	0.0	0.0	12.4	0.0	0.0	0.0	1.5	0.0	0.0	0.0	0.0	0.0	0.0
<i>Pseudotsuga menziesii</i>	0.0	0.0	0.0	0.0	0.0	0.0	1.2	535.5	0.0	0.0	2.3	324.6	0.0	0.0	0.0	8.3	0.0	0.0
<i>Quercus arizonica</i>	0.0	185.7	0.3	35.9	0.0	0.0	0.3	48.9	0.0	0.0	0.4	101.8	0.0	27.3	0.2	40.9	0.0	0.0
<i>Quercus emoryi</i>	0.2	38.7	0.0	0.0	0.0	40.4	0.0	3.5	0.0	50.0	0.0	0.0	0.1	14.5	0.0	0.0	0.3	162.3
<i>Quercus gambelii</i>	0.0	0.0	0.0	0.0	0.0	7.2	0.0	0.0	0.9	31.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
<i>Quercus gracilliformis</i>	0.1	4.5	0.0	0.0	0.0	0.0	0.0	0.7	0.4	8.0	0.1	1.5	0.0	0.0	0.0	11.0	0.0	0.4
<i>Quercus gravesii</i>	0.5	20.5	0.2	10.6	0.1	1.6	0.5	68.4	1.4	8.0	0.1	23.6	0.1	11.6	1.0	77.3	0.0	0.4
<i>Quercus grisea</i>	10.9	511.8	7.9	411.9	9.1	531.7	0.7	50.5	0.2	165.9	0.9	161.5	1.5	60.8	2.2	117.9	2.6	229.2
<i>Quercus hypoleucoides</i>	0.0	0.0	0.1	18.2	0.8	115.7	0.0	321.2	1.1	308.9	0.0	33.3	0.0	0.0	0.0	0.0	0.0	48.5
<i>Quercus laceyi</i>	0.0	0.0	0.4	69.2	0.0	0.0	0.2	25.0	0.0	0.0	0.3	58.1	0.0	0.0	0.0	0.0	0.0	0.0
<i>Quercus mohriana</i>	0.0	0.0	0.3	35.9	0.0	0.0	0.6	104.3	0.0	0.0	0.2	4.5	0.0	0.0	1.9	198.2	0.0	0.0
<i>Quercus meuhlebergii</i>	0.0	0.0	0.0	0.0	0.0	0.0	0.2	22.0	0.0	0.0	0.1	7.6	0.0	0.0	0.0	0.0	0.0	0.0
<i>Quercus oblongifolia</i>	0.0	0.0	0.2	18.2	0.0	0.0	0.0	1.4	0.7	15.9	0.0	6.1	0.0	0.0	0.9	31.8	0.0	0.0
<i>Quercus pungens</i>	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
<i>Quercus rugosa</i>	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0
<i>Quercus shumardii</i>	0.0	0.0	0.0	0.0	0.0	0.0	0.0	210.7	0.0	0.0	0.0	284.7	0.0	0.0	0.0	0.0	0.0	0.0
<i>Quercus sideroxyla</i>	0.0	0.0	0.0	0.0	0.0	0.0	7.3	0.0	0.0	0.0	2.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0
<i>Quercus tardifolia</i>	0.0	0.0	0.0	144.4	0.0	2.6	0.0	0.0	0.0	0.0	0.0	387.7	0.0	0.0	0.0	0.0	0.0	0.0
<i>Quercus turbinella</i>	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	14.9	0.0	0.0	0.0	0.0
<i>Quercus vaseyana</i>	0.0	0.0	0.0	0.0	0.0	0.0	0.1	2.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
<i>Salix nigra</i>	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
<i>Stryax youngiae</i>	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.7	0.0	0.0	0.0	1.5	0.0	0.0	0.0	0.0	0.0	0.0
<i>Tilia americana</i>	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Total	32.7	1540.2	19.1	1311.7	18.3	1446.4	29.2	3633.2	18.1	1642.3	29.8	3483.2	14.2	702.7	21.2	1496.2	14.8	1519.1

**Figure 9: Non-metric multidimensional scaling of vegetation samples (n=600) based on species importance values. Different symbols represent species composition groups identified by cluster analysis.**



**Table 5: Pearson product moment coefficients (n = 600) of environmental variables for NMDS axes 1, 2, and 3 from vegetation plots in DMTNC, BIBE, and MCPA. Values greater than 0.0877 are significant to  $p < 0.05$ .**

variable	axis 1	axis 2	axis 3
elevation (m)	-0.094	-0.024	-0.327
slope (degrees)	0.29	-0.188	-0.045
ISF	-0.165	0.16	0.468
DSF	-0.135	0.147	0.422
n aspect	0.03	-0.027	-0.093
e aspect	-0.01	0.099	-0.009
TRMI	0.07	-0.085	-0.31
PRR	-0.27	-0.019	0.164
convexity	-0.067	0	0.118
downslope neighbors	-0.142	-0.109	-0.014
downslope elevation	0.043	-0.214	-0.161
flow accumulation	0.115	0.072	-0.037
flow direction	0.035	0.087	-0.048
sediment transport	0.017	-0.164	-0.107
network index	0.005	0.223	0.15
relative elevation	0.137	0.2	0.037
shade relief	-0.009	0.123	0.036
topopos 450	-0.158	0.078	0.175
topopos 150	0.209	0.035	-0.073
wetness index	0.022	0.116	-0.015
grass cover (%)	-0.373	0.006	0.267
shrub cover (%)	0.154	0.043	0.31
bare soil (%)	-0.117	0.119	0.164

**Table 6: Mean environmental attributes for each vegetation type in sample plots in DMTNC, BIBE, and MCPA (n = 600).**

	<b>gray oak</b> n = 60	<b>gallery forest</b> n = 72	<b>graves oak</b> n = 18	<b>emory oak</b> n = 20	<b>pinon pine</b> n = 147	<b>oak - pinon</b> n = 106	<b>ponderosa pine - Arizona sw white pine</b> n = 59	<b>Arizona cypress - fir</b> n = 21	<b>alligator juniper</b> n = 97
elevation (m)	1917	1906	1834	1657	2037	2021	2364	2376	1924
slope (degrees)	20	23	22	12	20	17	18	19	12
ISF	0.573	0.333	0.381	0.585	0.555	0.568	0.349	0.282	0.597
DSF	0.625	0.367	0.406	0.632	0.589	0.597	0.399	0.312	0.620
e aspect	0.071	-0.194	0.224	0.157	0.072	0.010	0.149	0.156	0.074
n aspect	0.035	0.145	0.282	0.219	-0.101	0.048	0.065	-0.009	0.080
TRMI	29	34	39	34	25	29	34	31	31
PRR	18158	18266	15689	17963	18652	19417	18018	17771	19718
network	5.3	4.1	7.0	7.4	5.4	5.2	4.1	4.7	5.6
relative elevation	96.9	93.9	98.2	97.3	97.1	96.2	97.3	97.5	95.3
shade relief	0.48	0.45	0.48	0.49	0.52	0.48	0.51	0.54	0.49
topo pos 450	1.7	-16.6	-52.3	-22.3	11.9	-0.9	3.0	-4.6	-0.7
topo pos 150	-1.0	-0.7	16.2	4.7	-0.9	-1.0	-0.4	2.5	-1.2
wetness index	6.47	5.89	7.15	7.54	6.05	5.87	6.07	5.95	6.48
topo config	0.55	-0.37	-1.98	-0.54	0.41	0.22	-0.16	0.41	-0.05
downslope neighbor	3	4	3	2	4	4	4	4	4
down elev	8	16	5	4	10	12	13	13	7
flow accumulation	48	48	62	337	14	32	13	25	36
flow direction	30	23	53	38	31	27	36	32	31
sediment transport	13.5	24.3	9.6	6.4	13.8	16.3	21.6	16.3	11.5
grass (%)	45	41	26	50	52	59	37	32	66
shrub (%)	34	16	26	46	23	26	5	11	21
soil (%)	33	34	40	45	41	49	26	38	49

Figure 10: Mean size-class distributions (#/ha) in 5-cm classes of trees in the 6 vegetation types in DMTNC.

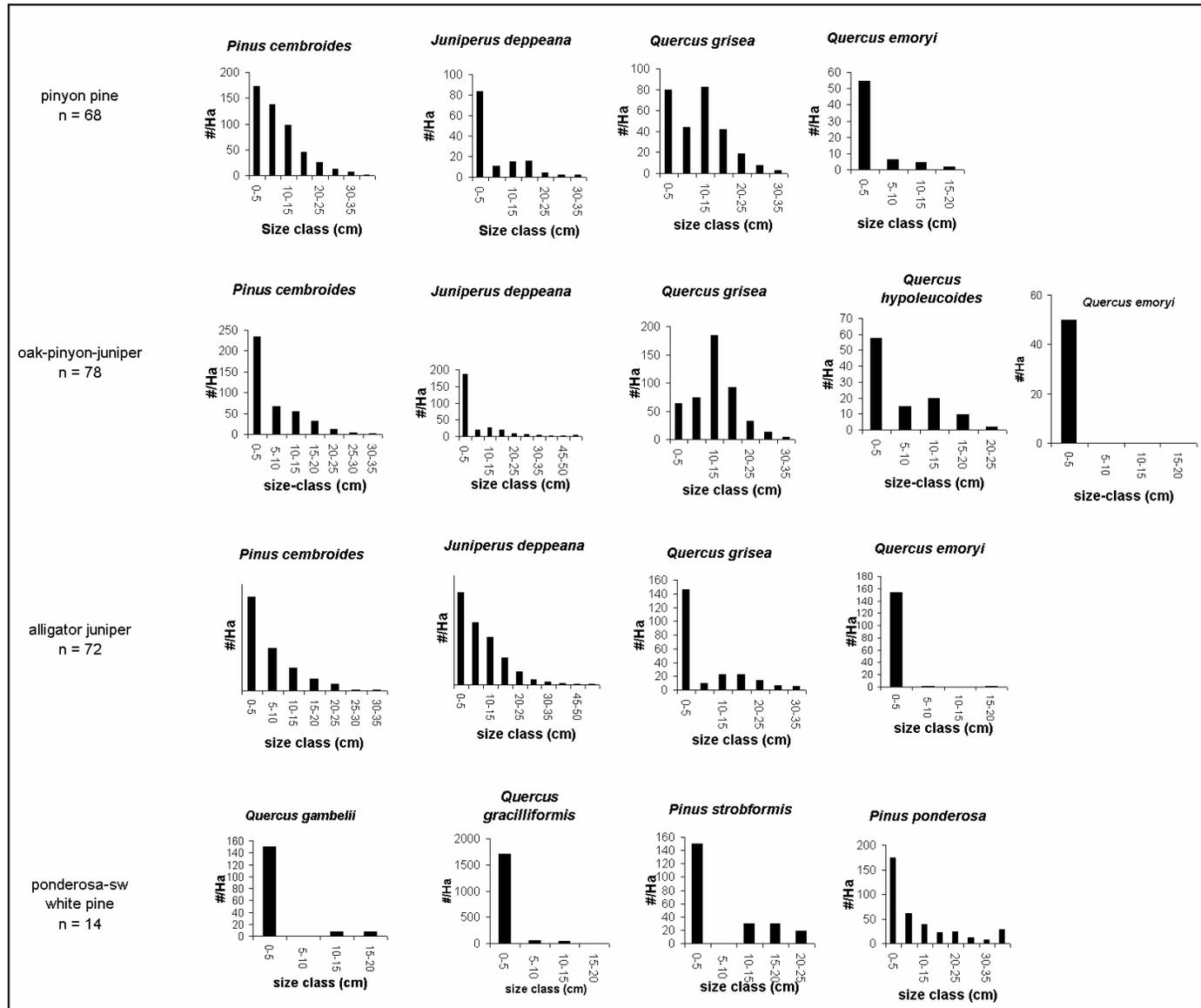


Figure 10: Continued. Mean size-class distributions (#/ha) in 5-cm classes of trees in the 6 vegetation types in MCPA.

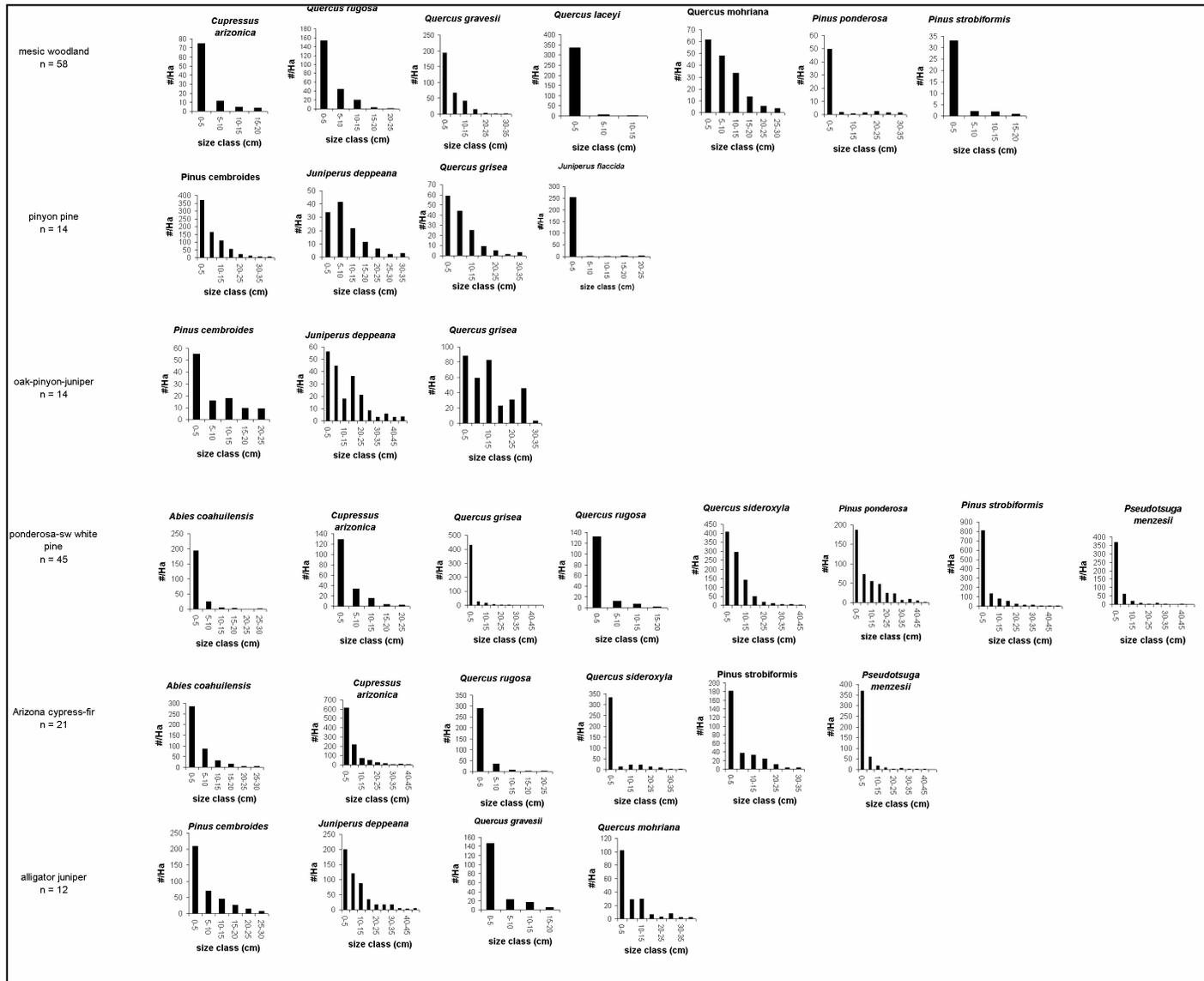
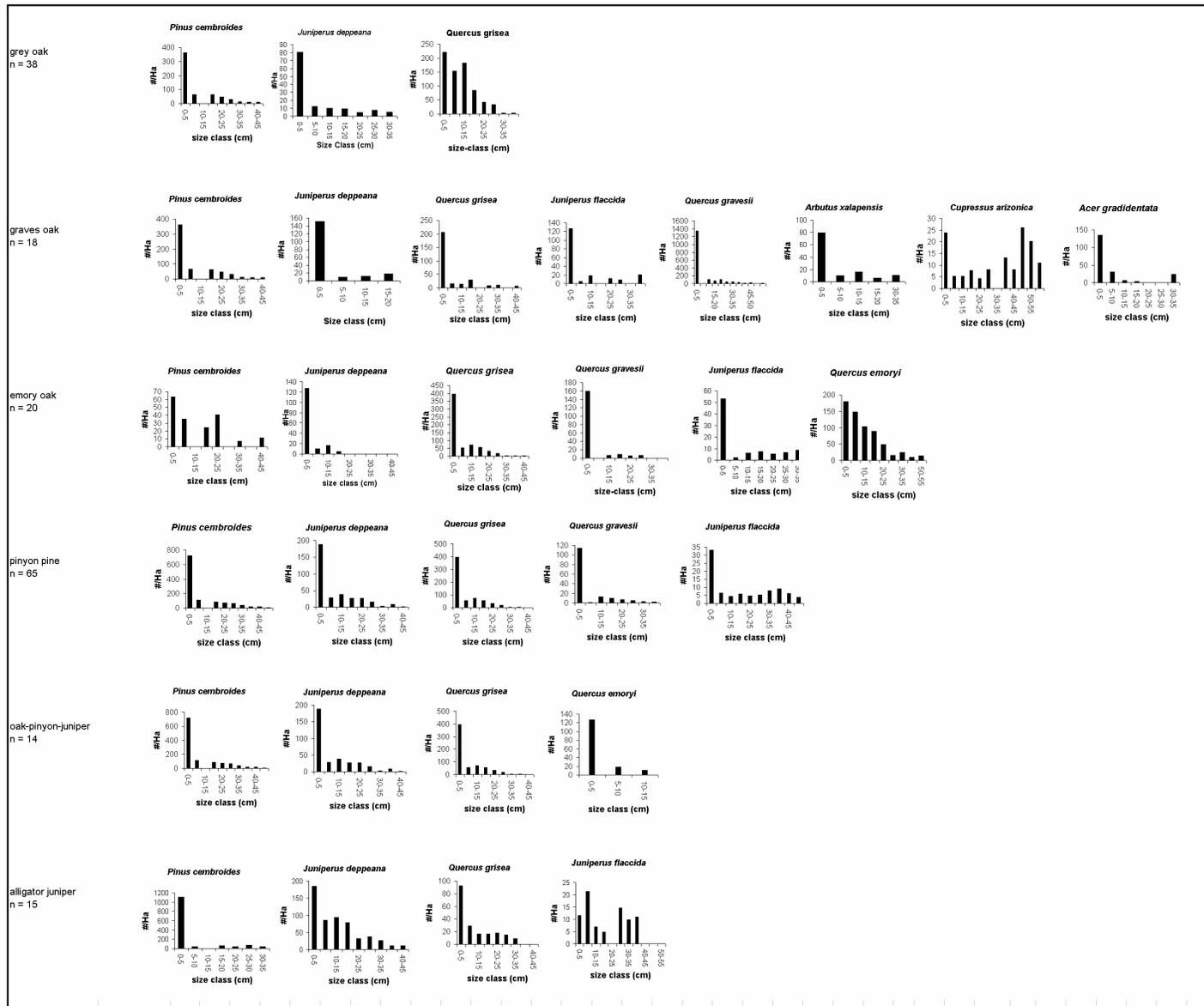


Figure 10: Continued. Mean size-class distributions (#/ha) in 5-cm classes of trees in the 6 vegetation types in BIBE.



All vegetation types in all three sites had high densities of young, small diameter trees (Table 4, Fig. 10). Reverse-J size distributions of conifers suggested a multi-aged population, while the size distributions of most oak species showed a peak at intermediate size classes. The high densities of small-diameter conifers ( $\leq 10$  cm DBH) in all vegetation types was consistent with the age data, and suggested that the majority of trees in the three study areas regenerated after fire was excluded as a disturbance process.

## Discussion

### Species Sorting Patterns

Environmental variation across landscapes is a well-known influence on tree species composition and forest stand structure. Our results are similar to prior research in Southwestern mountain ranges that identified elevation and soil moisture as the dominant influences on species sorting patterns (Whittaker and Niering 1965 and 1968, and 1975, Wentworth 1981, Heubner et al. 1999, Barton 1994, Poulos et al. in press). In addition to elevation and soil moisture, our results suggested that solar radiation and topographic position were also important factors influencing tree distribution patterns in our study region.

Elevation affects vegetation by decreasing temperature with increasing elevation according to the environmental lapse rate, generating lower temperatures and higher soil moisture at high elevations (Tranquillini 1964). This is why more mesic vegetation types such as Arizona cypress-fir forests and ponderosa pine forests occurred at higher elevations, and xeric forest types with pinyon pine and juniper dominated lower elevations.

Topographic position also affects forest microclimates because incident solar radiation varies across the surface of a slope. Lower topographic positions experience the least amount of solar radiation because it is often reflected upwards to mid-slopes, and because they receive shade relief from surrounding slopes. This further explains the occurrence of mesic vegetation types like gallery forests and Graves oak woodlands on lower topographic positions and valley bottoms. Mid-slopes have the highest incident solar radiation due to direct incoming radiation and reflection of radiation off other topographic positions. Drought tolerant vegetation types dominated these hotter and drier sites.

The tree species' distribution patterns we identified in our study were probably the result of the physiological ecology of trees in relation to drought and temperature tolerance. Junipers are known to be some of the most drought tolerant tree species in the Southwest (Padien and Lajtha 1992), which may explain why *Juniperus deppeana* dominated lower elevations that were characterized by low soil moisture and high solar radiation. Species such as *Pinus ponderosa* and *P. strobiformis* were restricted to upland valley bottoms because of their competitive advantage on wetter sites, but inability to survive and grow on the harshest sites in DMTNC (Barton 1993, Barton and Teeri 1993). Site generalists like *P. cembroides* spanned a wide range of site conditions due to their adaptability to a variety of moisture and radiation regimes (Poulos and Berlyn, in review).

## Forest Stand Structure

The high density of small diameter trees across our sites corresponds to the recent recruitment patterns observed in other fire-suppressed sites in the southwestern United States (Fulé et al. 1997, Fulé et al. 2002, Miller et al. 2003, Moore et al. 2004, Heinlein et al. 2005). Stand densities in BIBE, DMTNC, and MCPA were high, and they corresponded most closely to work by Sakulich and Taylor (*in review*), who reported tree densities of over 1500 stems per ha for pinyon and juniper-dominated portions of the Guadalupe Mountains, only 250 km to the north. While tree densities were similar in dry forest types, our study sites had much higher tree densities than the Guadalupe Mountains in mesic vegetation types dominated by ponderosa pine, southwestern white pine, Arizona cypress, and fir. Tree densities in DMTNC were similar to the fire-suppressed mixed conifer forests in the Camp Navajo Army National Guard Base and in the San Francisco Peaks of Arizona, which averaged 1436 stems ha<sup>-1</sup> and 1613 stems ha<sup>-1</sup>, respectively (Fulé et al. 1997, Heinlein et al. 2005).

Differences between conifer and oak diameter distributions in our sites are likely related to species life history. Ponderosa pine, southwestern white pine, pinyon pine and alligator juniper all regenerate readily from seed within 10-20 years after fire (Krugman et al. 1974, Moir et al. 1979, Evans 1988, Earl et al. 1996). This suggests that the pulse in tree recruitment beginning in the early 1900s in BIBE and DMTNC, and in the mid-1900s in MCPA may have occurred following the last fire events, which has persisted because of continued fire exclusion. In contrast, major oak recruitment events occur through post-fire sprouting in the western United States, with some seedling regeneration during fire free intervals (McClaran and Bartolome 1989, Nyandiga and McPherson 1992, Keeley 1992, McClaran and McPherson 1999, Barton 1999). The high densities of intermediate sized oaks in BIBE, DMTNC, and MCPA may be the remnant of post-fire sprouting after the last major fires in these sites.

Surprisingly, tree densities in MCPA were higher than the other two sites in all vegetation types but alligator juniper, suggesting that factors other than fire history may play an important role in determining forest stand density. While higher elevation areas comprised of ponderosa pine, southwestern white pine, Arizona cypress (*Curpressus arizonica*), Coahuila fir (*Abies coahuilensis*), and Douglas fir were logged during the 1950s and 1960s, lower elevation pinyon pine, juniper, and gray oak forests were never harvested. The higher density of trees across all vegetation types in MCPA relative to the other two sites suggests that historical logging (which occurred in only some vegetation types) is also not a probable cause for the differences in stand density in MCPA versus BIBE and DMTNC.

Environmental factors are the most likely explanation for the higher tree densities in MCPA. Variations in climate exist between the three mountain ranges because of the "Massenerhebungseffekt," (Schröter 1908 and 1926) or mountain mass buffering effect discussed by Barry (1992). The mountain mass buffering effect indicates that larger mountains have more stable climatic conditions than smaller mountains because of their size. The Sierra del Carmen is the largest and tallest of the three mountain ranges, which may equate to slower warming or cooling of this mountain range relative to the other sites. The greater climatic stability of the Sierra del Carmen may foster higher tree

regeneration and survival compared to the other smaller mountains that may be exposed to greater climatic variability.

Larger mountains have greater wet and dry adiabatic lapse rates and cloud cover than smaller mountains. Because of this, MCPA probably has greater precipitation and receives less incident solar radiation. Cloud cover and summer storms are often seen in MCPA from across the border in BIBE when BIBE, the smallest of the three mountain ranges, is experiencing clear sky conditions (Poulos, personal observation). The longer elevational gradient in MCPA could be responsible for these observations and could have the effect of stimulating greater thermal formation, cloud cover, and subsequently higher tree densities.

Lastly, mountain range orientation can influence climatic conditions. DMTNC and BIBE are east to west facing mountains, while MCPA is oriented north to south. Aspect effects on vegetation are well understood, and can be quite pronounced in the Southwest (Barry 1992). The orientation of mountain ranges modifies the regimes of temperature, evaporation, convection, and thermally induced wind circulations as a result of the augmentation (reduction) of solar radiation receipts on equatorward- (poleward) facing slopes. Plant species composition and density can change in response to such environmental variability. The fact that MCPA has a north-south orientation means that it receives reduced solar radiation relative to the other two mountain ranges. The reduced solar radiation could foster tree establishment and growth, explaining the higher tree densities in this site.

## Chapter 5 Tree Species Diversity

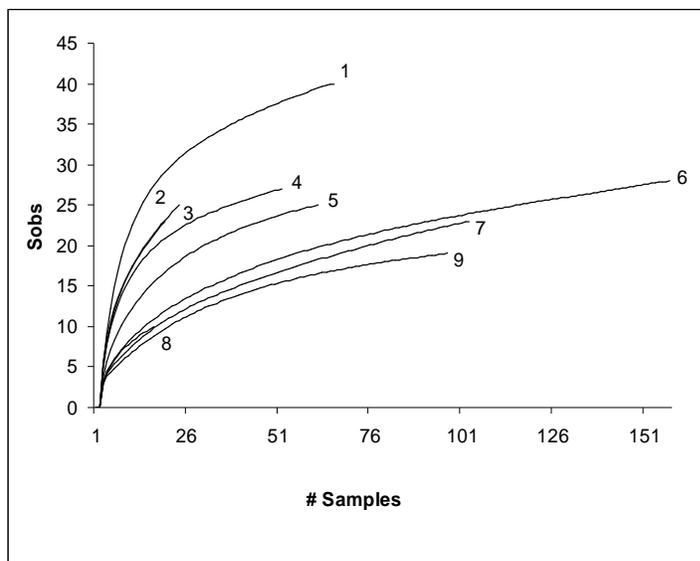
We assessed variation in species diversity at three scales: within vegetation types ( $\alpha$ ), among vegetation types ( $\beta$ ), and across the landscape ( $\gamma$ ). Species' count data for each plot were used in the diversity analysis (Whittaker 1972). Species richness ( $S_{obs}$ ), Simpson diversity ( $\tau$ ), Shannon diversity ( $H'$ ), were used as measures of alpha diversity ( $\alpha$ ), along with two non-parametric incidence-based species richness estimators, Chao<sub>2</sub> and Jack<sub>2</sub> (e.g. Whittaker 1972, Magurran 2004). We constructed sample-based rarefaction curves (species accumulation as a function of occurrence) with 95% confidence intervals (Colwell et al. 2004) to assess sampling completeness and to compare differences in species richness among vegetation types using *EstimateS* software (Version 7.5, Colwell 2005). Variation in Beta ( $\beta$ ) diversity (the degree to which different vegetation types shared species) was identified using Jaccard and Sørensen similarity indices that were modified and scaled by Chao *et al.* (2005) to accommodate for sample size differences among vegetation types. Gamma ( $\gamma$ ) (landscape diversity) was estimated and compared between sites using all of the plots at each site to quantify landscape-scale species richness. Gamma diversity was calculated by constructing a rarefaction curve for all samples and all species in DMTNC, BIBE, and MCPA, respectively.

### Results

#### Alpha Diversity

Vegetation types associated with mesic sites had higher species richness (more individual species) than vegetation types associated with dry sites (Figs. 11). Gallery forests had much greater species richness than all other vegetation types, while Emory oak and alligator juniper forests had the lowest species richness (Fig. 11).

**Figure 11: Sample-based rarefaction curve for richness of woody plant size structure for each vegetation type in the study. Numbers correspond to vegetation types as follows: 1 = gallery forest, 2 = Arizona cypress-fir, 3 = Graves oak, 4 = ponderosa-sw white pine, 5 = gray oak, 6 = pinyon pine, 7 = oak-pinyon-juniper, 8 = Emory oak, 9 = alligator juniper.**



## Beta Diversity

Chao-Jaccard and Chao-Sorenson similarity indices revealed that dry site forests had low beta diversity (more species in common), while wetter forest types had higher beta diversity (more distinct species assemblages) (Table 7). Gallery forests were an exception to this generality as they shared species with adjacent forest types including oak-pinyon-juniper, ponderosa-sw white pine, and Arizona cypress-fir forests. Ponderosa-sw white pine and Arizona cypress-fir forests had species in common, but shared few species with other vegetation types.

**Table 7: Chao-Jaccard and Chao-Sorenson similarity in woody plant species composition between vegetation types in the study region.**

CHAO-JACCARD									
	gray oak	mesic woodland	graves oak	emory oak	pinyon pine	oak-pinyon-juniper	ponderosa-sw white pine	cypress-fir	alligator juniper
gray oak	x								
gallery forest	0.772	x							
graves oak	0.84	0.66	x						
emory oak	0.90	0.31	0.75	x					
pinyon pine	0.96	0.81	0.87	0.95	x				
oak-pinyon-juniper	0.99	0.82	0.94	0.90	0.99	x			
ponderosa-sw white pine	0.64	0.89	0.34	0.06	0.83	0.57	x		
Arizona cypress-fir	0.56	0.93	0.59	0.06	0.61	0.91	0.98	x	
alligator juniper	0.98	0.79	0.98	0.95	0.98	1.00	0.67	0.72	x

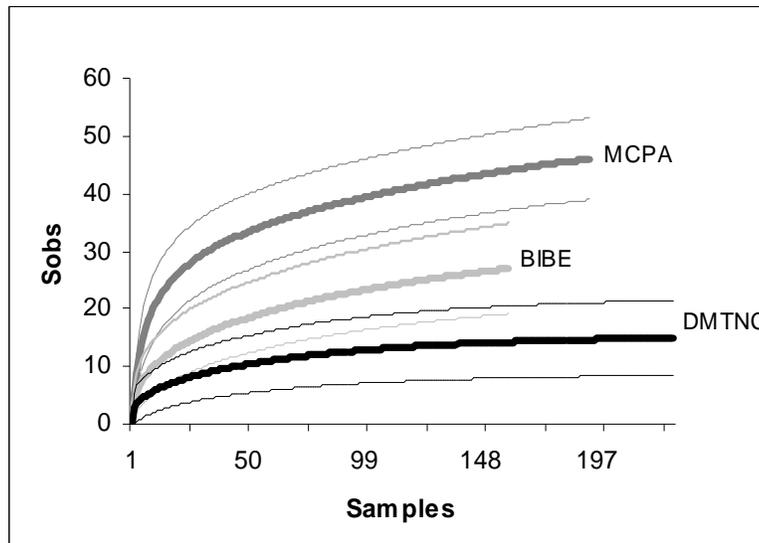
  

CHAO-SORENSEN									
	gray oak	mesic woodland	graves oak	emory oak	pinyon pine	oak-pinyon-juniper	ponderosa-sw white pine	cypress-fir	alligator juniper
gray oak	x								
gallery forest	0.87	x							
graves oak	0.91	0.79	x						
emory oak	0.95	0.47	0.86	x					
pinyon pine	0.98	0.89	0.93	0.97	x				
oak-pinyon-juniper	0.99	0.90	0.97	0.95	0.99	x			
ponderosa-sw white pine	0.78	0.94	0.50	0.12	0.91	0.73	x		
Arizona cypress-fir	0.72	0.96	0.74	0.12	0.76	0.95	0.99	x	
alligator juniper	0.99	0.88	0.99	0.97	0.99	1.00	0.80	0.84	x

## Gamma Diversity

Landscape-scale species richness was highest in MCPA, followed in order by BIBB, and DMTNC (Fig. 12). MCPA had significantly higher species richness than the other two sites.  $H'$  and  $\tau$  were also higher in MCPA than the other two sites. Sampling efficiency was adequate for the three sites, suggesting that an ample sample size was used for diversity estimation.

**Figure 12: Gamma diversity comparisons between the three sites. Thin lines represent 95% confidence intervals of rarefied species richness.**



## Discussion

### Species Richness

Species richness was highest in mesic vegetation types that comprised a relatively small proportion of the forested landscape. This suggests that these more rare vegetation types are important for sustaining biodiversity in this region. The high tree densities and basal area in mesic vegetation types also suggest these forests are also highly productive. Targeting these forest types for conservation will help ensure the maintenance of Sky Island biodiversity in an otherwise desert landscape.

The incidence of high biodiversity in mesic vegetation types in our study sites similar to that found in the Rocky Mountains (Peet 1978 and 1981), where species richness is tied to moisture regimes. In the rest of the Southwest, species richness has been correlated to topographic complexity and forest physiognomy (Whittaker and Neiring 1975, Poulos et al. in press). Similar to our study sites, wetter sites in the Rockies had the highest species richness (Peet 1978). In the Southwest, species richness was highest in short-stature vegetation types (pinyon pine and juniper) that spanned a wide range of elevations and topographic settings in southeastern Arizona. The variability of species richness across landscapes of the Southwest suggests the need for site-specific management to assure the protection of biodiversity hotspots.

### Species Turnover

Beta diversity (species turnover among vegetation types) highlighted that mesic vegetation types contained unique assemblages of tree species. Not only did mesic vegetation types have high biodiversity, but they also appeared to contain tree species that were found nowhere else on the landscape.

## Regional Diversity Patterns

Gamma diversity (landscape-scale diversity), comparisons suggested that the largest, most southerly preserve had the highest species richness in this eco-region. While our sample size was small, four potential explanations exist for this pattern; 1) the effect of larger mountain ranges on species richness; 2) the effect of topographic complexity; 3) the presence of a latitudinal gradient in species richness; and 4) the northward migration of species following the recession of the Laurentide and Cordilleran continental glaciers in the mid-Pleistocene (VanDevender 1977).

Larger mountains provide a wider range of potential habitats for species colonization, which may explain the higher species richness in MCPA. This explanation is based on the species-area relationship where species richness increases as a function of area (MacArthur and Wilson 1963). Larger preserves can expect greater diversity in species composition, while smaller preserves cannot realistically expect to encapsulate all of the species diversity that exists in a region.

Mountains with more broken topography may also provide a wider diversity of habitats for plant colonization. Gamma diversity appeared to increase with topographic complexity in our sites. DMTNC had the lowest species richness and was the least topographically complex mountain range in our study. BIBE was intermediate in both gamma diversity and topographic complexity. Gamma diversity was highest in MCPA, which contained the most broken topography, consisting mainly of incised towers and steep canyons. Poulos et al. (in press) found the same pattern in the Chiricahua National Monument, where species richness was highest in vegetation types on topographically-complex sites.

The latitudinal gradient in species richness in our study area suggests that mountain size and topography are the only explanatory patterns for species richness. BIBE is the smallest mountain range in our study, but contains more species than larger DMTNC. Our sites appear to follow “Rappaport’s rule”, or the mid-domain effect, which postulates the trend of increasing species richness with decreasing latitude (Colwell and Lees 2000). This pattern is most likely related to the northward migration of species from southern latitudes during the late Wisconsin (13,000 years ago) and early Holocene (8,000 ~ 10,000 years ago) (VanDevender 1977, VanDevender and Spaulding 1979, Lanner and VanDevender 1981, and Elias and VanDevender 1992). The climates of the Southwest originated in the late Wisconsin era, culminating in conditions suitable for species migrations from equatorial latitudes. The depauperate species richness of DMTNC is potentially a result of being the most northerly mountain range in our study.

## Chapter 6 Fuel Distribution Patterns

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The fuel loading of the forest canopy was quantified by taking a hemispherical photograph at the center point of each vegetation plot. Hemispherical photographs were taken with a Nikon Coolpix 900 digital camera with a Coolpix 900 fish-eye lens mounted on a self-leveling tripod that was positioned 1 m above the ground. Pictures were taken under cloudy sky conditions in the morning. The leaf area index (LAI) of each plot was calculated using HemiView canopy analysis software version 2.1 (Delta-T Devices 1999). LAI data were converted to crown bulk density (CBD) following Keane et al. (2005). Live fuel structure was determined by calculating the total BA ( $\text{ha}^{-1}$ ), biomass ( $\text{kg m}^{-2}$ ), and density ( $\text{ha}^{-1}$ ) of trees in 5 cm size-classes in each sample plot. Standing dead tree structure was summarized by the total standing dead BA ( $\text{ha}^{-1}$ ) and density ( $\text{ha}^{-1}$ ) in 5 cm size-classes. Dead and down fuels were sampled using the point relascope method for quantifying coarse woody debris (CWD) developed by Gove et al. (2001). Fine fuels were not sampled in this study. Instead, we chose to focus on quantifying potential fire intensity by sampling CWD to investigate the factors that drive interactions between larger fuels and climate.

### Fuels Analysis

Fuel types and distribution patterns were quantified using cluster analysis and non-metric multidimensional scaling in the same manner as the vegetation data.

### Results

We identified five fuel load types across the three study areas that differed by fuel load characteristics and local environment (Fig. 13, Tables 8-12) ( $\text{MRPP} < 0.0001$ ). While vegetation type influenced the accumulation of fuels, it was not the dominant influence on fuel accumulation. Environmental factors, including elevation, slope, topographic position, and incident solar radiation, were the dominant influences on fuel loadings. In general, the amount of CWD was low, with 205 of the 600 sample plots (34.2%) having no dead and down fuel. Fuel types 1, 3, and 5 were found in DMTNC; fuel types 1, 3, and 4 were found in BIBE; and Fuel types 1-4 were found in MCPA. Fuel loads increased with elevation and soil moisture, and decreased in relation to slope, topographic position and incident solar radiation. Aspect was not a significant factor related to the distribution of fuels

Differences among fuel load types were evident from the mean fuel load characteristics, and their distribution was dependent on the variation in local environment across the landscape ( $\text{MRPP} < 0.001$ ) (Tables 8, 10, and 11). Fuel load type 1 was characterized by low live and dead coarse woody fuels and high fine fuel cover (% grass cover). Type 1 fuel loads were found at middle elevations on steep, upper topographic positions (i.e. ridgetops). These sites were quite dry and exposed (low shade relief). Fuel type 2 was characterized by high crown bulk density and high numbers of small seedlings. This fuel type dominated steep mesic valley bottoms at low elevations that had a high capacity for transporting sediment. Fuel type 3 had low crown bulk density, live trees, and standing dead fuels. CWD volume CWD volume and 1000-hr fuels were intermediate for this group. This fuel type existed on straight slopes on lower topographic

positions having high incident solar radiation. Fuel type 4 had the highest amount of live, standing dead, and dead and down fuels. This type dominated high elevation mesic valley bottoms that had high sediment accumulation. Fuel type 5 had low canopy and dead and down fuels, but high fine fuel cover (i.e. grass and litter). Fuel type 5 was found on straight, gentle slopes at high elevations and high amounts of solar radiation.

Live and standing dead fuels were highest at MCPA, followed in order by BIBE, and DMTNC, a pattern similar to live tree density (Table 12). Dead and down CWD volume was highest in BIBE, followed by MCPA and DMTNC. One hundred hour fuels were highest at DMTNC, and 1000-hr fuels were highest at BIBE. Fine fuels including grass and litter were highest at DMTNC.

**Table 8: Fuel load characteristics for each fuel load type in the Chihuahuan Desert Borderlands.**

Fuel Load Type	Fuel Load Characteristics	Environmental Characteristics
1	<ul style="list-style-type: none"> <li>• Low live fuels</li> <li>• Low standing dead fuel</li> <li>• Low dead and downed fuel</li> <li>• High grass cover</li> <li>• Low snag density</li> </ul>	<ul style="list-style-type: none"> <li>• Middle elevations</li> <li>• Steep slopes</li> <li>• Exposed sites</li> <li>• Upper topographic positions</li> <li>• Dry sites</li> </ul>
2	<ul style="list-style-type: none"> <li>• High CBD</li> <li>• High seedling density</li> <li>• High 100 hr fuels</li> <li>• Low grass, shrub and litter cover</li> <li>• Low snag density</li> </ul>	<ul style="list-style-type: none"> <li>• Low elevation, valley bottoms</li> <li>• Steep, convex slopes</li> <li>• Mesic sites</li> </ul>
3	<ul style="list-style-type: none"> <li>• Low live and standing dead tree density</li> <li>• Low snag density</li> </ul>	<ul style="list-style-type: none"> <li>• Middle elevations</li> <li>• Lower topographic positions</li> <li>• High solar radiation</li> </ul>
4	<ul style="list-style-type: none"> <li>• High live fuels</li> <li>• High snag density</li> <li>• Abundant large dead and downed fuels</li> <li>• High intermediate-sized (20-25 cm) snag density</li> </ul>	<ul style="list-style-type: none"> <li>• High elevation valley bottoms</li> <li>• Mesic sites</li> <li>• High sediment accumulation</li> </ul>
5	<ul style="list-style-type: none"> <li>• Low coarse fuel loads</li> <li>• High grass and litter cover</li> <li>• High snag density</li> </ul>	<ul style="list-style-type: none"> <li>• High elevations</li> <li>• Ridgetops</li> <li>• High solar radiation</li> <li>• Gentle slopes</li> </ul>

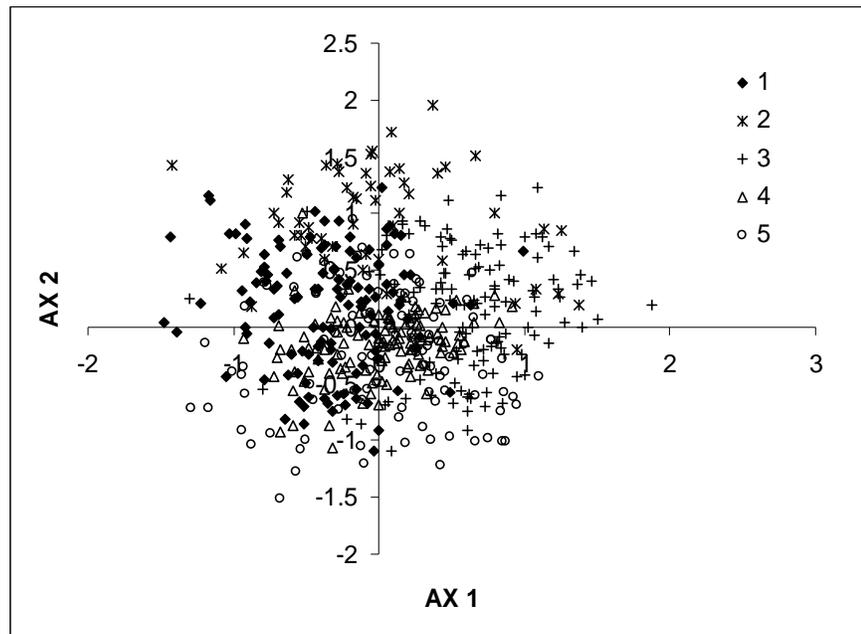
**Table 9: Mean summary statistics for fuel load types derived from cluster analysis of fuel characteristics of sample plots (n = 600) in the study area. Environmental variables differ significantly by fuel type according to MRPP ( $P < 0.001$ ).**

<b>fuel characteristics</b>	1			2	3			4			5
crown bulk density (kg m <sup>-3</sup> )	0.093	0.068	0.055	0.147	0.083	0.075	0.065	0.125	0.112	0.090	0.082
100-hour (kg ha <sup>-1</sup> )	10.6	0.0	0.0	240.9	70.4	1510.3	0.0	350.3	1170.3	340.7	46.1
1000-hour (kg ha <sup>-1</sup> )	2490.4	5120.5	7800.6	6340.8	9070.9	13640.2	2880.9	17010.7	8370.5	29350.7	596.6
logs (ha <sup>-1</sup> )*	16.9	44.8	37.3	48.8	88.6	206.3	66.0	134.3	412.5	315.0	119.1
volume (m <sup>2</sup> ha <sup>-1</sup> )	631.7	432.9	2028.4	1719.6	2162.4	1216.8	729.7	4830.2	762.7	7789.4	534.3
live BA (m <sup>2</sup> ha <sup>-1</sup> )	11.6	7.3	5.1	19.3	10.4	11.7	6.9	25.0	27.0	19.8	16.6
standing dead BA (m <sup>2</sup> ha <sup>-1</sup> )	0.6	1.2	0.0	1.5	1.0	2.2	0.3	4.6	4.3	3.5	2.0
grass cover (%)	72	52	39	19	24	58	38	47	57	47	62
shrub cover (%)	18	19	26	21	32	27	60	5	26	23	20
litter cover (%)	13	33	20	25	62	55	38	24	57	43	55
<b>standing dead trees (ha<sup>-1</sup>)</b>											
5-10 cm	24.4	5.7	3.1	36.7	18.3	0.0	7.0	38.6	0.0	19.5	5.6
10-15 cm	15.6	1.9	0.0	10.1	28.3	1.9	4.7	24.0	0.0	35.0	3.0
15-20 cm	8.9	0.2	2.5	6.7	5.0	0.4	5.0	16.3	0.0	35.6	1.3
20-25cm	2.2	3.0	0.0	4.4	6.7	16.3	8.0	12.2	14.7	23.0	58.2
25-30 cm	3.7	0.1	0.0	5.6	1.7	0.0	0.0	6.1	1.6	17.1	1.7
30-35 cm	4.4	1.5	0.0	1.1	3.3	0.0	0.0	4.1	0.0	10.9	0.0
> 35 cm	7.4	0.9	0.0	7.8	15.0	0.0	0.0	10.6	0.0	35.5	0.8
<b>live trees (ha<sup>-1</sup>)</b>											
0-5 cm	885.0	197.9	209.8	4121.2	720.2	515.7	471.3	1917.3	1316.7	1877.4	723.1
5-10 cm	229.1	68.1	126.2	84.0	74.6	99.7	156.0	693.2	678.1	282.4	293.2
10-15 cm	126.7	89.8	84.9	44.4	73.7	132.5	138.1	375.0	471.7	323.7	346.0
15-20 cm	66.8	63.9	44.2	28.7	47.0	89.3	71.6	158.0	157.6	251.5	197.4
20-25 cm	40.0	37.3	16.1	13.2	26.3	56.9	67.3	70.9	61.7	180.3	68.3
25-30 cm	28.3	14.0	15.7	16.1	19.2	26.7	57.9	46.7	31.3	132.4	21.7
30-35 cm	8.0	8.8	33.7	8.4	6.7	16.1	26.3	23.0	13.1	60.0	10.1
> 35 cm	9.1	10.1	47.6	4.6	8.1	14.1	15.2	34.5	34.3	107.6	10.3

**Table 10: Mean values of environmental variables for fuel load types identified by cluster analysis for the study area (n = 600). Environmental variables differ significantly by fuel type according to MRPP ( $P < 0.001$ ).**

<b>Environmental variable</b>	<b>1</b>	<b>2</b>	<b>3</b>	<b>4</b>	<b>5</b>
elevation (m)	1946	1808	1943	2121	2000
slope (degrees)	20	23	16	19	15
ISF	0.577	0.239	0.622	0.430	0.573
DSF	0.624	0.267	0.650	0.471	0.584
e aspect	-0.070	-0.365	0.099	0.104	0.140
n aspect	0.119	0.334	0.003	-0.043	-0.006
TRMI	27.4	39.5	29.1	29.6	29.3
PRR	18935	19308	18929	18056	19566
network	4.7	4.7	5.2	5.5	5.4
relative elevation	95	93	96	98	96
shade relief	0.487	0.402	0.503	0.501	0.509
topo pos 450	6.8	-8.4	-1.0	-2.6	2.5
topo pos 150	-0.4	1.7	-1.5	-0.3	0.6
wetness index	5.9	5.9	6.1	6.5	6.0
topo config	0.34	0.55	0.06	-0.05	0.07
downslope neighbor	3.5	4.0	3.5	3.3	4.0
down elev	12.9	16.2	10.2	7.8	11.1
flow accumulation	23.7	28.0	56.9	39.4	21.7
flow direction	29	17	30	31	36
sediment transport	17.4	26.1	17.7	11.3	14.8

**Figure 13: Non-metric multidimensional scaling of CWD samples (n=600) based on fuel characteristics. Different symbols represent fuel load groups identified by cluster analysis.**



**Table 11: Pearson product moment coefficients (n = 600) of environmental variables for NMDS axes 1, 2, and 3 from CWD plots in DMTNC, BIBE, and MCPA. Values greater than 0.0877 are significant to  $p < 0.05$ .**

<b>Environmental variable</b>	<b>Axis 1</b>	<b>Axis 2</b>	<b>Axis 3</b>
elevation	0.359	0.037	0.063
topopos_450	0.036	-0.202	-0.037
topopos_150	-0.040	0.119	0.071
slope	-0.009	0.164	0.183
N aspect	0.010	-0.058	-0.071
S aspect	0.002	0.086	0.065
ISF	-0.423	-0.415	-0.165
DSF	-0.372	-0.388	-0.119
TRMI	0.124	0.220	0.014
PRR**	0.005	-0.067	-0.145
flow_dir	0.012	-0.006	-0.080
flow accum	-0.064	0.088	-0.029
topographic configuration	-0.020	-0.057	-0.041
downslope neighbors	0.109	0.066	-0.180
downslope elevation change	0.041	0.241	-0.091
landform	0.005	0.169	-0.052
network	-0.143	-0.047	0.058
relative elevation	0.047	-0.188	0.257
shade releif	0.000	-0.112	-0.050
wetness index	-0.021	-0.035	0.046

**Table 12: Mean fuel characteristics for DMTNC, BIBE and MCPA**

<b>fuel characteristics</b>	<b>DMTNC</b>	<b>BIBE</b>	<b>MCPA</b>
crown bulk density (kg m <sup>-3</sup> )	0.08	0.08	0.12
100-hour (kg ha <sup>-1</sup> )	790.7	230.7	210.8
1000-hour (kg ha <sup>-1</sup> )	8790.1	21640.3	11210.3
logs (ha <sup>-1</sup> )*	158.1	232.7	83.3
volume (m <sup>3</sup> ha <sup>-1</sup> )	3.8	27.8	14.5
live BA (m <sup>2</sup> ha <sup>-1</sup> )	13.7	15.8	19.0
standing dead BA (m <sup>2</sup> ha <sup>-1</sup> )	2.1	2.6	2.6
grass cover (%)	58.0	44.5	46.0
shrub cover (%)	22.6	30.8	14.0
litter cover (%)	50.8	39.8	26.6
<b>standing dead trees (ha<sup>-1</sup>)</b>			
5-10 cm	0.0	15.0	32.7
10-15 cm	0.4	24.8	20.9
15-20 cm	0.2	26.2	11.1
20-25cm	31.0	17.3	7.1
25-30 cm	0.1	12.2	5.1
30-35 cm	0.0	8.7	3.5
> 35 cm	0.1	25.4	9.3
<b>live trees (ha<sup>-1</sup>)</b>			
0-5 cm	655.0	1401.3	1905.7
5-10 cm	207.8	240.1	401.1
10-15 cm	223.4	259.3	224.7
15-20 cm	126.1	193.6	101.0
20-25 cm	56.8	140.6	47.7
25-30 cm	22.8	104.6	33.2
30-35 cm	12.3	52.8	14.5
> 35 cm	13.9	85.9	19.7

## Discussion

To our knowledge, this is the first study investigating the distribution of fuels in relation to topography in the western United States, although Rubino and McCarthy (2003) and Graham and McCarthy (2006) investigated the distribution of CWD at the landscape-scale in Ohio. Our results suggested that the distribution of larger fuels across dissected landscapes was associated with some of the same factors controlling the distribution of vegetation, but that the factors and mechanisms responsible for these patterns were not identical. Elevation and incident solar radiation influenced site productivity for the reasons mentioned in chapter 4, with higher elevations having lower temperatures, greater cloud cover, and potentially decreased amounts of incident solar radiation under cloudy conditions. Higher productivity sites existed at high elevations in response to more favorable moisture and temperature regimes. This pattern of high fuel accumulation in high productivity sites is consistent with research from other parts of the western United States, where forests on high productivity sites averaged more pieces and volume per ha of down wood than low productivity sites (Harmon et al. 1986, Spies et al. 1988, Spetich et al. 1999).

Topographic effects on fuel distribution patterns were similar to those for vegetation, but the relationship between topographic position and gravity was responsible for the heavier fuel loads in valley bottoms than on mid-slopes and ridge tops. That mesic valley bottoms are more productive than mid-slopes that receive high amounts of incident solar radiation partially explains the trend of higher fuel loads on lower topographic positions (Hariston and Grigal 1991). Valley bottoms also receive sedimentation from upland areas during rainstorms, and fuels often settle in lower topographic positions by rolling down hill from middle and upper topographic positions. Steeper slopes exhibit lower amounts of fuel because it washes downward to flatter areas.

Differences in fuel loads and environmental site conditions among the fuel types in our study indicated that fires could burn with greater variability across the three landscapes. The incidence of fine fuels that promote fire ignition and spread (grasses, litter and 100 hr fuels) were consistently higher across all fuel load types in DMTNC than the other two sites, suggesting that fire spread would be faster at this site. These factors would contribute to a regime of frequent, low intensity fire. BIBE and MCPA had higher CWD and live fuel components that were probably the result of fuel buildup after fire exclusion. The higher fuel loads in BIBE and DMTNC suggest that these reserves may be more prone to higher intensity burns that are uncharacteristic of historical fire regimes.

Different fuel load types might burn differently during extreme climatic anomalies. Fire ignition and spread are closely tied to the presence of fine fuels (McClaran and VanDevender 1995). El Nino/La Niña cycles could have a greater impact on forests in DMTNC due to the higher coverage of fine fuels in this site. In BIBE and MCPA higher amounts of live, standing dead and down fuels are available and could burn under extremely dry conditions that occur less frequently, at a much higher intensity.

Topographic variability and site productivity drivers for fuel distribution patterns in our three study areas indicates that upper elevations, valley bottoms, and lower topographic positions may be primary foci for fuels reductions treatments. Because of their lower fuel loads, lowlands and mid-to-upper topographic positions may be less critical areas for fuel treatments.

## Snag Dynamics

This study represents the first research quantifying the abundance and distribution patterns of snags in the Southwest for forest types other than ponderosa pine and fir. No other information exists that describes the snag incidence for pinyon pine, juniper, or mixed pine-oak forests, snag inventories by Ganey and Vojta (2004) in the Coconino and Kaibab National forests in Arizona included a few pinyon pine and juniper snags.

The density of live trees, snags and dead and down fuels was high in MCPA compared to other sites in Mexico such as the Sierra San Pedro Martir (SSPM) in Baja California where fire exclusion has not occurred (Stephens et al. 2003, Stephens and Gil 2005). However, snag densities on our sites were similar to those reported for other fire-suppressed areas in western North America (Barbour et al. 2002, Ganey 1999, Ganey and Vojta 2004). Snag densities for fuel load types 1-3 were similar to those found in the SSPM, and ranged from 2-8 snags ha<sup>-1</sup>. Snags in our sites were generally much smaller in diameter than in SSPM and other parts of the Southwest (Ganey 1999, Ganey and Vojta 2004). Fuel load types 4 and 5 had high snag densities relative to SSPM, perhaps a result of heavy tree mortality during the 1950s drought (Taylor, unpublished data) or from western pine beetle infestations (Everitt et al. 1997a, Everitt et al. 1997b) in west Texas.

Differences in the dominant vegetation types and in the management history of our study area probably explain the larger diameters of trees in other sites relative to ours. All prior research on snag dynamics in the Southwest and northern Mexico has been conducted in Jeffrey and ponderosa pine dominated forests that generally have larger trees. Forests in our sites were mainly composed of smaller pinyon pines, oaks, and junipers. Fuel load types 4 and 5 contained larger trees including ponderosa pine, southwestern white pine, Douglas fir, Coahuila fir, and Arizona cypress; snags in these forests were more similar in size to those described in previous studies.

Snags and large woody debris have high ecosystem value because they provide nesting, foraging, and resting habitat for many wildlife species in North America (Balda 1975, Harmon et al. 1986, Bull et al. 1997, Rabe et al. 1998, Laudenslayer 2002). In the southwestern United States, a rich assemblage of birds is largely dependent on snags for nesting sites (Balda 1975, Cunningham et al. 1980). Ponderosa pine cavity nesters such as the Mexican spotted owl (*Strix occidentalis lucida* Nelson), northern flicker (*Colapetes auratus* Linnaeus) and hairy woodpecker (*Picoides villosus* Linnaeus) are known to inhabit our sites, highlighting the importance of maintaining snags in ponderosa-sw white pine forests types, especially if fire is reintroduced to these landscapes. Pinyon, juniper, and oak dominated woodlands host several other species of cavity nesting birds including the Gray Vireo (*Vireo vicinior* Coues), the tufted titmouse (*Baelophus bicolor* Linnaeus), the whiskered screech owl (*Otus trichopsis* Wagler), and the Strickland's woodpecker (*Picoides stricklandi* Malherbe). The dearth of knowledge about the cavity nesting requirements of these species underscores the need for future studies that investigate cavity use and tree species preferences of birds in pinyon-juniper woodlands of the Southwest and Mexico.

## Chapter 7 Predictive Mapping of Vegetation and Fuels

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### Landscape Metrics

A set of 15 raster-based topographic, landform, and solar radiation variables were derived using the National Elevation Dataset (NED) for the study area at a 30 x 30 m spatial resolution (USGS 2005) (Table 2). We filtered each of these grids using a 3 x 3 pixel window assigning the mean value to the center pixel to reduce fine scale noise in the dataset, and to improve prediction accuracy following Gregory and Ohmann (2002). Raster values for the landscape metrics were assigned to each plot by intersecting the spatial location of the sample plot with each landscape data layer using ArcMap 9.2 software (ESRI 2005).

### Landsat 7 ETM+

A cloudless Landsat enhanced Thematic Mapper (ETM+) image from June 18, 2002 was used to develop 11 data layers from bands 1-5 and 7. A June scene from 2002 was chosen because it coincided with the timing of field sampling of vegetation and fuels, and because the scene was taken during the fore summer drought just prior to the onset of the summer monsoon. All tree species in the study area are evergreen, and therefore a scene taken outside of the growing season was chosen to reduce the noise in the spectral signature from perennial grasses and herbaceous vegetation. We transformed the image into tasseled cap brightness, greenness, and wetness indices (Kauth and Thomas 1976), computed a soil adjusted vegetation index (SAVI Gilabert 2002) and an Enhanced Vegetation Index (EVI; Huete et al. 2000). Tasseled cap transformations were made based on their good performance in mapping vegetation cover (Jensen 1996). SAVI and EVI were chosen over the more widespread Normalized Difference Vegetation Index (NDVI) because of the poor performance of NDVI in arid regions (Gilabert 1988). The raster grids were filtered in the same manner as the landscape grids, and ETM+ derived values were assigned to vegetation and fuel plots by intersecting the spatial location of the plots with each of the successive satellite data layers.

### Vegetation and Fuels Mapping

We used classification trees (Breiman et al. 1984) to predict the vegetation and fuel load types obtained from our cluster analysis results across the landscapes of DMTNC, BIBE, and MCPA. Vegetation and fuel decision trees were constructed for each site using the rpart package in the statistical language R (R Development Core Team, 2005). Landscape and spectral characteristics for each plot were used as predictor variables; vegetation or fuel type served as response variables. The fit of the vegetation and fuel models was evaluated by examining the cost complexity parameter which measures how well the explanatory landscape metric and Landsat ETM+ derived variables separate the data. Percent misclassification and the Kappa statistic (Congalton and Green 1999) were used to assess classification accuracy. Predictive vegetation and fuel maps were developed using Decision Tree in ENVI remote sensing software, version 4.2 (Research Systems Inc. 2005). Field validation of the models is being undertaken to assess map accuracies ( $n = 200$ ); validation points have been stratified in the field by vegetation type and fuel type.

**Table 13: Landscape metrics and spectral data used for predicting vegetation and fuel types using classification and regression trees.**

Variable code	Definition
<b>Landscape Metrics</b>	
Elevation	Elevation (m), from 30-m digital elevation model (DEM)
N Aspect	Cosine transformation of aspect (degrees) (Beers et al. 1966) 1.0 (southwest) to -1.0 northeast
S Aspect	Sine transformation of aspect (degrees) (Beers et al. 1966) 1.0 (southwest) to -1.0 northeast
Slope	Slope (degrees), from 30-m DEM
PRR	Cumulative potential relative radiation based on hourly solar position, topography and topographic shading (Pierce et al. 2005)
Topopos 150	topographic position, calculated as the difference between a cell's elevation and the mean elevation of cells within a 150 m radius
Topopos 450	topographic position, calculated as the difference between a cell's elevation and the mean elevation of cells within a 450 m radius
Topo configuration	Topographic configuration ranging from concave to convex calculated using the spatial analyst function in ArcMap 9.1
topo configuration	topographic configuration ranging from concave to convex calculated using the spatial analyst function in ArcMap 9.1
Landform	landform type derived from Terrain Analysis System software (Lindsay 2005) based on Pennock et al. (1987). Landform types include (1) convergent footslope, (2) divergent footslope, (3) convergent shoulder, (4) divergent shoulder, (5) convergent backslope, (6) divergent backslope, and (7) level
sediment transport	sediment transport capacity index = $(As / 22.13)^{0.6} \times (\sin S / 0.0896)^{1.3}$ where As = the specific catchment area and S = the local slope
wetness index	wetness index (Beven and Kirkby 1979) derived from Terrain Analysis System software (Lindsay 2005) defined as $WI = \ln(As/\tan S)$ .
network index	minimum wetness index value encountered along a flowpath (Lane et al. 2004). This value defines when a cell with a zero or negative saturation deficit is connected to the drainage network
flow direction	flow direction from ArcHydro extension in ArcMap 9.1 and 30-m DEM
flow accumulation	flow accumulation from ArcHydro extension in ArcMap 9.1 and 30-m DEM
<b>Landsat ETM+</b>	
Band 1	Band 1 (blue)
Band 2	Band 2 (green)
Band 3	Band 3 (red)
Band 4	Band 4 (near-infrared)
Band 5	Band 5 (mid-infrared)
Band 7	Band 7 (mid-infrared)
Brightness	Soil brightness index from tasseled cap transformation
Greenness	Green vegetation index from tasseled cap transformation
Wetness	Wetness index from tasseled cap transformation
EVI	Enhanced Vegetation Index (Huete et al. 2000)
SAVI	Soil Adjusted Vegetation Index (Gilabert et al. 2002)

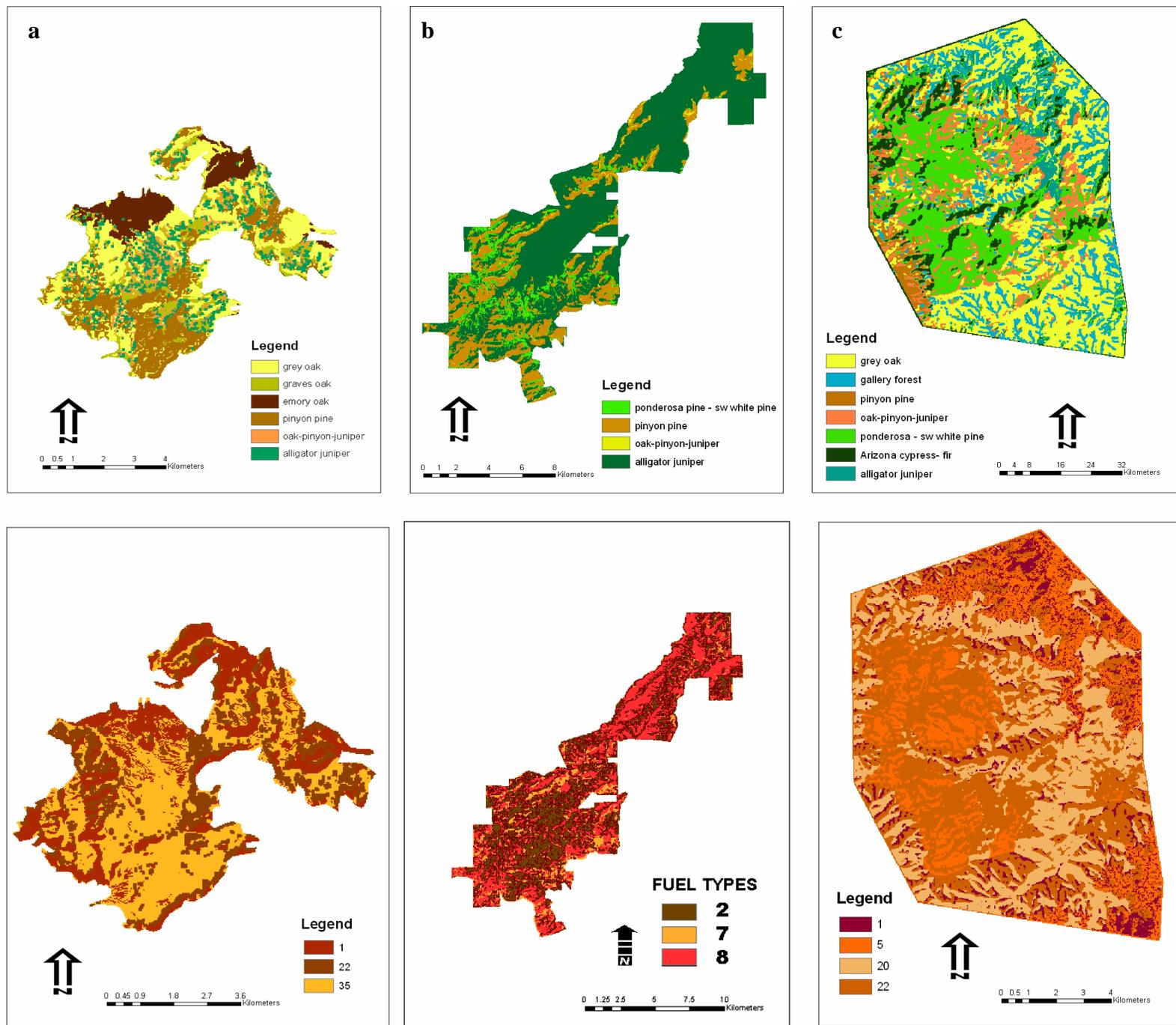
## Results

The environmental and spectral predictors of the distributions of vegetation and fuels differed by study area (Tables 13 and 14). Individual models were built for each site using independent CART algorithms in light of the differentiation of the spectral and environmental characteristics across sites (Figs. 14).

Vegetation and fuel load types were differentiated by spectral signature (MRPP  $P < 0.0001$ ) (Tables 12 and 13). The overall misclassification accuracy for the vegetation maps was 29.9% for DMTNC, 26.4% for BIBE, and 31% for MCPA (Table 15). The classification accuracy was lower for vegetation types containing species that occurred in multiple vegetation types (i.e. gray oak, alligator juniper, and oak-pinyon-juniper vegetation types). *Quercus grisea*, *Juniperus deppeana*, and *Pinus cembroides* dominated specific vegetation types, but were also found across a range of different sites. The overlap in species composition most likely caused spectral confusion that resulted in the misclassification of these vegetation types. Other vegetation types such as gallery forest also contained a wide mixture of species, making correct classification more difficult.

Classification accuracy for the fuels maps was greater than for the vegetation maps: 28.8% for DMTNC, 17.0% for BIBE, and 13.3% for MCPA. Spectral confusion was also a probable cause for any misclassification of fuel load types since there was some overlap in fuel load characteristics and tree species composition. The kappa statistic for the vegetation map was 0.50 for DMTNC, 0.69 for BIBE and 0.66 for MCPA. Kappa values were similar for the fuels maps with values of 0.52 for DMTNC, 0.54 for BIBE, and 0.61 for MCPA.

Figure 14: Vegetation (top) and fuels (bottom) maps for a) BIBE, b) DMTNC, and c) MCPA



**Table 13: Mean spectral characteristics of the vegetation types for each of the three study sites. Spectral characteristics differ significantly by vegetation type according to MRPP ( $P < 0.001$ ).**

	gray oak	gallery forest	graves oak	emory oak	pinyon pine	oak-pinyon-juniper	ponderosa pine - sw white pine	Arizona cypress - fir	alligator juniper
band 1	122	113	102	123	94	89	47	48	93
band 2	129	125	112	126	102	99	67	65	102
band 3	127	131	108	120	94	89	66	58	92
band 4	177	180	173	176	152	158	162	148	147
band 5	166	149	175	171	129	123	85	78	127
band 6L	135	133	153	167	101	112	59	54	139
band 6H	133	135	148	163	101	114	61	55	140
band 7	129	130	125	130	99	93	63	61	101
EVI	0.200	0.471	-0.068	0.200	0.158	0.419	-0.371	-0.377	0.571
SAVI	-0.096	-0.203	0.193	0.017	0.117	0.235	0.395	0.381	0.087
brightness	0.078	0.121	-0.076	0.200	-0.167	0.200	-0.426	-0.503	-0.060
greenness	0.020	-0.045	0.319	0.088	0.201	0.257	0.560	0.540	0.136
wetness	-0.110	-0.214	-0.201	-0.184	0.157	0.149	0.372	0.429	0.049

**Table 14: Mean spectral characteristics of the fuel types for each of the three study sites. Spectral characteristics differ significantly by vegetation type according to MRPP ( $P < 0.001$ ).**

	1	2	3	4	5
band 1	111	105	75	85	93
band 2	117	119	90	95	102
band 3	111	121	89	84	94
band 4	158	176	164	149	162
band 5	137	138	111	115	157
band 6L	130	127	92	125	98
band 6h	131	129	93	127	94
band 7	115	120	87	91	104
SAVI	121	114	146	154	161
EVI	-0.051	-0.104	0.148	0.204	0.265
greenness	0.435	0.353	0.069	0.584	-0.209
wetness	-0.011	0.039	0.310	0.211	0.411
brightness	-0.026	-0.103	0.176	0.124	0.154

## Discussion

Results from this study provide spatially explicit decision support tools for mitigating fire hazards and restoring forest vegetation to less fire prone conditions. The integrated classification tree approach proved to be an effective tool for quantifying vegetation and fuel abundance and distribution patterns and for predicting their occurrence across the landscapes of DMTNC, BIBE and MCPA. We believe that the methods we used are potentially useful in other regions of the Southwest where detailed forest vegetation and fuels inventory data are lacking.

Our misclassification rates of approximately 30% are typical of other vegetation mapping projects based on decision tree classifiers. Other misclassification rates for predicting vegetation and fuels using classification tree and biophysical gradient models ranged from 67 % to 89% (Franklin 1998 and 2002, Miller et al. 2003, Brown de Colstoun et al. 2003, Falkowski et al. 2005). Our misclassification rates reflect the difficulty of reducing ecological processes like species distribution and fuel accumulation patterns to numerical models. Moreover, we mapped distinct vegetation assemblages at the species level; the majority of vegetation mapping studies assigned vegetation types to much broader categories such as oak, grassland, or conifer cover types. Spectral confusion from the overlap in species composition among vegetation types probably accounts for much of the error rates in our study, since species like juniper and pinyon pine were distributed throughout multiple vegetation types.

The vegetation and fuels maps we produced have broad application for informing forest, wildlife, and fire management. Vegetation maps can be used to identify potential habitats for wildlife and our vegetation maps are already being used by managers in conjunction with Montezuma quail and Mexican spotted owl projects in DMTNC. Because the vegetation maps are linked to forest stand structures and diversity patterns, managers can use them to identify locations on the landscape that are of particular interest for biodiversity management. For example, managers often strive to maintain a diversity forest stand structures across a landscape to provide a range of successional habitats for wildlife. Our results suggest that the greatest biological diversity in this region is found in mesic valley bottoms that comprise a relatively small proportion of the landscape. Our vegetation maps can be used to identify these biodiversity hotspots.

The fuels maps produced as a result of our study can be used to identify locations on the landscape where fuel loads are out of synchrony with inherent disturbance regimes and may require active management to restore them to more sustainable conditions. In large protected areas such as DMTNC, BIBE, and MCPA, it is impossible to treat the entire landscape to reduce the risk of catastrophic, high intensity fire. Spatially explicit maps linked to detailed fuel survey data can help identify locations on the landscape that will benefit from fuels treatments.

## Chapter 8 Conclusions and Management Implications

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Frequent low intensity fire was a dominant force shaping the Sky Island forests of the Chihuahuan Desert Borderlands. The disruption of this inherent fire regime in the mid 20<sup>th</sup> century altered forest stand structure and fuel loads. Knowledge derived from our research can guide managers toward developing strategies for restoring forests to sustainable stand structures and fuel loads. Our understanding of the interplay between vegetation patterns and topography will help managers to identify key locations to concentrate their biodiversity protection and restoration efforts.

We identified two major differences between our study area and other portions of the Southwest with respect to historical fire seasonality and the onset and causes of fire exclusion. Fires occurred earlier in the growing season in the Chihuahuan Desert Borderlands than in much of Arizona and New Mexico, a pattern that corresponds to fire seasonality and lightning ignition patterns in Mexico (Mosiño Alemán and Garcia 1974, Turman and Edgar 1982, SEMARNAP 2000). From a management perspective this implies that prescribed burns that take place in March will most closely resemble historical fire seasonality. It is well known that fire seasonality has a strong effect on vegetation response (Agee 1993), but managers are often restricted to burning outside historical burning seasons. March fires may appeal to managers wanting to implement prescribed fires during times of the year other than summer when high temperatures and low fuel moisture can cause escaped fires to burn uncontrollably.

Fire exclusion in our study region occurred 50-80 years later than the majority of previously studied sites in Arizona and New Mexico, suggesting that forests in the Chihuahuan Desert Borderlands may have stand structures that are less-altered from fire exclusion than in other regions. Therefore, it may be possible to more easily restore these forests using natural and prescribed fire and thinning. Little is known about vegetation response to fire in the Chihuahuan Desert Borderlands. Only through experimentation will managers be able to determine the effectiveness of restoration treatments designed to restore ecosystem structure and function.

Another important component of this study that distinguishes it from much prior fire history research is that we analyzed historical natural variability of fire in pinyon pine forests. While the historical frequency of fire in pinyon-juniper woodlands remains a point of debate (i.e. Floyd et al. 2000 and 2004, Baker and Shinneman 2004), our data represent only the second fire scar-based fire history study in this forest type. Moreover, our results imply that fires were historically frequent in these woodlands, with changes in forest stand structure occurring in response to the disruption of this disturbance process. Based on our findings, prescribed fire is indeed a valid management tool for this forest type, although the fire regime characteristics of pinyon-juniper woodlands may vary considerably from region to region.

Climate change forecasts for northern Mexico are consistent in suggesting that future conditions will be warmer and drier (Magaña et al. 1997, Villers-Ruiz and Trejo-Vázquez 1998). This may cause further elevational migrations of vegetation and increase the frequency and severity of extreme drought years. Such changes in climate may cause shifts in the types of vegetation most prone to fire. Forests with heavy live, dead and down fuel loads may become especially vulnerable to high intensity fires. Gallery, ponderosa pine-sw white pine, and Arizona cypress-fir vegetation types that have high

tree densities and basal areas may be especially susceptible to fire under drier conditions. The higher susceptibility of these forest types is alarming considering that they contain the greatest  $\alpha$ - and  $\beta$ - diversity on the landscape. Upper elevations and valley bottoms with high live and dead fuel accumulations may also be more susceptible to high intensity fire in under future climate change scenarios.

While reference conditions that characterize historical fire regimes and forest stand structures reflect a “snapshot” in time, such information can be used by managers to understand the range of historical natural variability of fire regime parameters (size, seasonality, and frequency), fire-climate relationships, and recent changes in forest stand structure and fuel loads resulting from the cessation of frequent, low intensity fire. Historical information such as the fire regime statistics presented in this report should be interpreted in the context of modern management goals and shifting mosaics of forest stand structures and species composition under a changing climatic regime. A historical mean fire return interval of 8-10 years does not necessarily mean that managers should emulate this pattern. Sustainable forest stand structures may be better maintained using prescribed fire on a more or less frequent basis (e.g. Fulé et al. 2003). Just as Baker and Ehle (2001) argued that historical fires did not burn the entire landscape, our data suggest that fires in our study area were predominantly small in size. The small size of historical fires in this region suggests that similarly small-sized prescribed fires might be an effective method of mimicking natural fire regimes.

Results from our study highlight the close relationship between landscape structure and the distribution of fuels, vegetation, and tree diversity in DMTNC, BIBE, and MCPA. Tree distribution patterns follow similar environmental gradients reported for other Southwestern mountain ranges, although diversity distribution patterns in our sites differ. The Sky Island evergreen woodlands of the Sierra Madres are thought to be biodiversity hotspots in an otherwise desert landscape (deBano et al. 2004, Gotfried et al. 2004). Until now, little data exist to quantify biodiversity in these ecosystems (but see Whittaker and Niering 1975, Poulos in press). This study fills knowledge gaps about the species richness and distribution of biodiversity in a little-studied part of the Sierra Madre Oriental. Biodiversity is highest in mesic vegetation types that comprise a relatively small proportion of the landscape. The vegetation maps produced in this study highlight the spatial distribution of species richness on the landscape. Managers can use both the diversity distribution patterns and the vegetation maps to target these communities for conservation.

The fuel distribution patterns in our study area suggest upper elevations and valley bottoms have the highest fuel loads in DMTNC, BIBE, and MCPA. Identifying locations in need of fuel reduction treatments that are also accessible is often difficult. Our analysis of the distribution of fuels and our spatially explicit fuel prediction maps are potentially valuable decision support tools to assist managers in understanding how fuels are distributed across the landscape, and where fuel treatments using fire and fire surrogates are most needed or could be most effective.

Information about the composition and distribution of snags across the landscapes of DMTNC, BIBE, and MCPA provide new information about wildlife habitat in pinyon-juniper and pine-oak woodlands. This information has broad application for a variety of forest management activities including fire science and wildlife management. Since this work represents the first research that quantifies the occurrence of snags in pinyon-

juniper and pine-oak woodlands, our results may be applicable to snag management in other locations with similar forest types.

## Conclusion

Results from this study provide managers with key information that can be used to guide forest, fuels, and wildlife management at the local-, landscape-, and regional-scales. Our work highlights the interactions among fire, climate, topography, and vegetation and fuel distribution patterns. Furthermore, we discovered that our study sites are different in some key respects to other regions of the Southwest. Our results indicate that fire and vegetation patterns in the Chihuahuan Desert Borderlands are more similar to parts of Mexico than to other parts of the Southwest. This work presents new information about forest ecosystem dynamics in several understudied forest types. Pinyon-juniper and pine-oak woodlands cover over 325,000 square kilometers in North America, from southern Idaho in the United States (42° N Lat.) to southern Puebla, Mexico (18° N Lat.) (Aldon and Springfield 1973, Barger and Ffolliot 1972). However, little is known about the relationships between fire, climate and vegetation dynamics in these forest types. Future studies that investigate these relationships in other similar forested areas would provide valuable points of comparison for our work.

## Deliverables

### Technology Transfer

Information about the role of climate in shaping fire regimes and shaping vegetation patterns is being shared with land managers and independent stakeholders in the following ways: 1) submission of manuscripts to peer-reviewed scientific journals; 2) dissemination of research findings to fire and resource management staff in Big Bend National Park, The Nature Conservancy, The Chihuahuan Desert Fire Ecology Group and Maderas del Carmen; 3) paper and poster presentations at regional conferences and symposia; and 4) incorporation of key findings in interpretative displays being developed at the environmental education center in DMTNC and the visitors center at BIBE

This study has also enhanced the recently established "sister parks" relationship between BIBE and MCPA. Sharing research initiatives has helped strengthen the relationship between the two parks. Information from this study will also be shared with a Mexican protected area adjacent to BIBE and MCPA (Santa Elena Canyon Protected Area), since this preserve shares many of the same ecological and anthropogenically-induced processes.

Our deliverables for this project exceed those outlined in our initial proposal. Electronic versions of all reports and articles will be submitted to the Joint Fire Science Program Office.

Final reports (completed by the end of March 2007) to the participating agencies will include:

- Data, maps, and reports will be shared with the Chihuahuan Desert Network. Data include: raw .xls data files, and fire history data, fire-climate interaction data, and vegetation stand structure and diversity data in the form of digital figure and table

- files that they can incorporate into subsequent presentations of their own. ArcGis shapefiles of the vegetation and fuel maps will be shared with each management agency so that they may use them for future planning and management.
- Raw data files will be provided following acceptance of corresponding manuscripts submitted to peer-reviewed journals.

#### Publications

- Poulos, H. M. In press. The Physiological Ecology of Pine-Oak Forests of the American Southwest. *Journal of Sustainable forestry*.
- Poulos, H. M., A. E. Camp, R. G. Gatewood, and L. Loomis. In press. A hierarchical approach for scaling forest inventory and fuels data from local to landscape scales in the Davis Mountains, Texas, USA. *Forest Ecology and Management*.
- H. M. Poulos, U. Manage Goodale, and G.P. Berlyn. In review. Drought response of two Mexican oak species, *Quercus laceyi* and *Quercus sideroxyla* (Fagaceae) in relation to elevational position. *American Journal of Botany*.
- Poulos, H. M., and G. P. Berlyn. In review. Phenotypic plasticity in needle morphology and water status of *Pinus cembroides* across an elevational gradient in the Davis Mountains of west Texas, USA. *Journal of the Torrey Botanical Society*.
- Poulos, H. M, A. E. Camp, J. Villanueva-Díaz, J. Serano, and R. G. Gatewood. In preparation. Fire regimes and fire-climate interactions of the Chihuahuan Desert Borderlands. To be submitted to *Ecological Applications*.
- Poulos, H. M., and A. E. Camp. In preparation. Environmental controls on dominance and diversity of woody plant species in the Chihuahuan Desert Borderlands. To be submitted to *Plant Ecology*.
- Poulos, H. M., and A. E. Camp. In preparation. Environmental controls on the distribution of coarse woody debris in relict montane conifer forests of west Texas, USA and northern Coahuila, Mexico. To be submitted to *Forest Ecology and Management*.
- Poulos, H. M. In preparation. Snag dynamics of pinyon-juniper woodlands and pine-oak forests of the Sky Islands of the Chihuahuan Desert Borderlands. To be submitted to *Forest Ecology and Management*
- Poulos, H. M., A. E. Camp, and R. G. Gatewood. In preparation. Predictive mapping of vegetation and fuel loads in the Chihuahuan Desert Borderlands using a classification and regression tree approach. To be submitted to *Remote Sensing of Environment*
- Villanueva-Díaz, J., J. Serano, and H. M. Poulos. A 300-year record of climate for the Davis Mountains, Texas, USA. To be submitted to *The Tree Ring Bulletin*
- Poulos, H. M., and G. P. Berlyn In preparation. Leaf anatomical variability and photosynthetic capacity of three species of *Quercus* (Fagaceae). To be submitted to *Tree Physiology*.
- Poulos, H. M., and G. P. Berlyn. In preparation. Drought tolerance and physiological response of *Quercus sideroxyla* to biostimulants. To be submitted to *Journal of Sustainable Forestry*.

### Symposium Proceedings

- Poulos, H.M., and A.E. Camp. 2005. Vegetation-environment relations of the Chisos Mountains, Big Bend National Park, Texas. Biodiversity and Management of the Madrean Archipelago II, May 11-15<sup>th</sup>, 2004.

### Research Presentations

#### Finished

- Mills, H.R., and A.E. Camp. 2004. *Habitat Modeling and Vegetation dynamics of the Chisos Mountains, Big Bend National Park*. Association of American Geographers, Annual Meeting, Philadelphia, Pennsylvania.

#### Upcoming

- Poulos, H. M. October 2006. *Predictive mapping of vegetation and fuel loads in the Chihuahuan Desert Borderlands using a classification and regression tree approach*. Northeastern Mensurationists Annual Meeting, Grey Towers, Pennsylvania.
- Final presentations to the managers of each of the three study sites, scheduled for spring 2007.

### Interactive Displays

- Interactive displays about fire-climate-vegetation interactions are under construction in BIBE and DMTNC, and will be finished by December 2006.

### Dissertation Theses

- H. M. Poulos. Landscape- and regional-scale controls on fire regimes and vegetation abundance, distribution and diversity patterns in the Chihuahuan Desert Borderlands. To be completed in January 2007.

### Additional Deliverables

- Vegetation and fuels maps were developed for each study site using field data, landscape metrics derived from digital elevation models, and spectral information from a Landsat ETM+ scene. The maps were produced using a flexible, non-parametric classification and regression tree approach.
- Ground truthing of the vegetation and fuels maps is underway, and will be completed by December 2006.
- In addition to our analysis of vegetation-environment relationships, we also performed a comprehensive analysis how tree species diversity is distributed across the landscape, and determined which environmental factors promote the highest tree species diversity.
- We also performed a suite of greenhouse experiments on water relations to provide a mechanistic explanation for the species distribution patterns we quantified in this study.

### Lessons Learned

Conducting this research across multiple study areas allowed us to quantify and compare fire, climate, and vegetation interactions from local to regional scales. Selecting

three mountain ranges that differed in elevational gradient and size allowed us to determine how the species composition, diversity and distribution patterns varied according to the size, location, and topography of the individual mountain ranges.

Working with three distinct management agencies in sites that straddle an international boundary posed several logistical difficulties. The closing of the border crossing in BIBE after September 11<sup>th</sup>, 2001 increased our trips between BIBE and MCPA from one to 16 hours. Getting the necessary permits to work in Mexico also proved challenging, taking more than a year from the submission of the paperwork to actually obtaining documents. The new Homeland Security Act made moving across the border with our tree core and fire-scar samples difficult, even with the appropriate permits for collection and transport of these materials.

The director of the dendrochronology lab at the Instituto Nacional de Investigación Forestal y Agropecuaria (INIFAP), José Villanueva-Díaz, and professors at the Universidad Nacional Autónoma Antonio Narro (UNAAN), sponsored us toward obtaining permits for working in MCPA and processed the tree core and fire-scar samples we were unable to transport back to the United States. This has been a mutually beneficial relationship.

In addition to the originally proposed study, we used our dataset to develop vegetation and fuels maps for each site using ancillary data derived from digital elevation models and satellite imagery. This product is currently being used for forest and fire management applications, and the vegetation maps have helped identify potential Mexican spotted owl and Montezuma quail habitat in the Davis Mountains. The fuels maps are being used to identify areas at risk for high intensity fire. The maps have also assisted managers in identifying locations for potential fuels treatments.

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