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ECOSYSTEM CONSEQUENCES OF REGIONAL PINYON MORTALITY

By Michael J. Clifford

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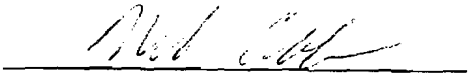

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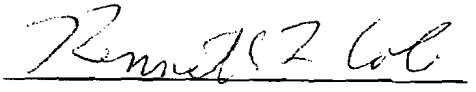
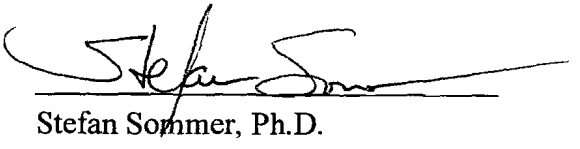


Kenneth L. Cole, Ph.D.



Stefan Sommer, Ph.D.

ABSTRACT

ECOSYSTEM CONSEQUENCES OF REGIONAL PINYON MORTALITY

Michael J. Clifford

Pinyon-juniper (*Pinus edulis – Juniperus* spp.) woodlands in the American Southwest have expanded in many areas since the late Nineteenth century. Woodland expansion has occurred in both, extent and increasing stand density. Expansion of woodlands has been attributed to reduced fire frequency, increased ungulate grazing, and changes in climate. These increases of woody vegetation have been shown to alter nutrient cycling, fire dynamics, and vegetation patterns across the landscape which can have large-scaled impacts on ecosystem function. Recently, pinyon-juniper woodlands and the American Southwest have experienced nearly a decade of continuous drought, beginning in the mid-1990s. In 2002 severe drought caused a bark beetle outbreak (*Ips confusus*) in pinyon pine, causing high levels of pinyon mortality throughout the region. This persistent drought has caused wide spread mortality among pinyons, and to a lesser extent, junipers.

Determining how stands have been altered by tree die-off, and determining which characteristics of a stand that promotes increased mortality are important for monitoring vegetation change under a changing regional and global climate. This will help managers in these semi-arid regions to understand how climatic events can alter the landscape. The spatio-temporal patterns and changes in vegetation have been

adequately detected among other woody vegetation types throughout the world. Documenting these woodland dynamics at stand to landscape-scales using field methods and remote sensing allows for an increased understanding of how climatic changes impact overstory vegetation under a global climate change-type drought.

Results from remotely sensed data and ground plots show increases of 32% in pinyon-juniper woodland canopy cover from 1936 until the drought of 2002 - 2003. During this brief period of drought and outbreaking bark beetles, canopy cover was reduced to levels lower than found in 1936. During times of woodland expansion, landscape heterogeneity was reduced, but was reduced even further after the mortality event in 2002 - 2003. Drought-induced tree mortality was largely unexplained by stand-level characteristics, while environmental factors appear to influence mortality more greatly, but relationships were still weak indicating severe drought induced stress occurred throughout the region.

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Preface

This thesis was written for publication, Chapter 1 will be submitted to Landscape Ecology, while Chapter 2 will be submitted to Forest Ecology and Management. For consistency, these chapters are written in the same format, but will contain redundancies in some areas pertaining to background information and ending conclusions. Each chapter will be modified individually for submission to the prospective journal.

Chapter 1: Ecological thresholds and climate change-type drought in a pinyon-juniper woodland

Abstract

We demonstrate how drought-induced mortality of pinyon pine (*Pinus edulis*) can act as an ecological threshold and reset canopy cover to levels prior to 68 years of woodland expansion. Since late Nineteenth Century settlement, woody vegetation has expanded throughout southwestern North America. This historic expansion has been linked to increased livestock grazing, fire reduction, and changes in climate. During 2002 and 2003, regional drought lead to a bark beetle (*Ips confusus*) outbreak, causing high levels of pinyon mortality. Using high resolution panchromatic aerial photographs from 1937 and 1997, and QuickBird satellite imagery from 2004 to measure the spatio-temporal dynamics of these woodlands over 68 years, the following questions were addressed: 1) What are the spatio-temporal dynamics of pinyon-juniper woodlands? And, 2) How has global climate change-type drought altered woodland cover? 3) What are the differences in the spatio-temporal dynamics of woodland expansion vs. woodland die-off? And 4) What was the impact of past fire on canopy cover? Results indicate approximately a 32% increase in canopy cover (from 9.5% to 14%) in canopy cover from 1936 to 1997. This was followed by a 55% decrease (from 14% to 6.4%) in total canopy cover from 1997 to 2004. Furthermore, areas of low canopy cover in 1936 were more likely to have increases in canopy cover later in the century, but canopy area patterns were not predictive in determining which areas experienced canopy loss from tree mortality. Expansion of woodlands also caused a decrease in landscape heterogeneity, as vegetation cover classes

decreased 37%, while drought mortality further decreased cover classes another 22%, for a total cover class loss of 52% from 1936 to 2004. During the 68 year time period of our study area, 6% of the landscape burned, but did not greatly reduce canopy cover, indicating bark beetle mortality plays a greater role in structuring woodland cover.

Introduction

Woody vegetation has increased in spatial extent and density throughout much of the western US since the late 19th century (Johnsen and Dalen 1990; Archer 1994; Yorks et al. 1994; Belsky 1996; Van Auken 2000; Brockway et al. 2002; Floyd et al. 2004; Lalibert et al. 2004; Anderson and Baker 2005; Shaw 2006; Strand et al. 2006; Weisberg et al. 2007; Bradley and Fleishman 2008; Strand et al. 2008). Expansion of woody vegetation in these semi-arid regions during the past century has predominantly been attributed to grazing (Archer 1994; Van Auken 2000; Harris et al. 2003), fire suppression (Van Auken 2000), and climatic change and atmospheric CO_2 enrichment during the past century (Archer 1994; but see Van Auken 2000 for review). Much of the current research has focused on the impact of expansion to biogeochemical cycles or above and below ground primary production (Knapp et al. 2008). Expansion of woody vegetation can increase carbon sequestration in these systems (Asner et al. 2003; Knapp et al. 2008; Strand et al. 2008).

Climatically driven disturbance events such as floods, hurricanes, fire, drought etc. are a major source of vegetation changes and drive large-scale patch dynamics (Turner 1989; Malmstrom and Raffa 2000; Asner et al. 2003), as well as changes at

the stand-level (Allen et al. 2006). These disturbance events, typically considered as part of the natural range of variability, can alter vegetation and shift ecotones (Allen and Breshears 1998; Dullinger et al. 2004; Floyd et al. 2004). In the semi-arid Southwest, drought has been a major disturbance, but also a driver of other disturbance events (Allen and Breshears 1998). Drought, for example, promotes longer fire seasons and larger and more severe fires (Schoennagel et al. 2004; Westerling et al. 2006) and increases the risk of bark beetle outbreaks (Allen and Breshears 1998; Breshears et al. 2005; Shaw et al. 2005; Muller et al. 2005; Gitlin et al. 2006; McDowell et al. 2008).

Global and regional climate models show a trend towards warming, greater aridity, and an increased frequency of extreme droughts in the Southwest (IPPC 2007; Seager et al. 2007). Vegetation will also change within the natural range of variability as climate changes. Past vegetation changed as climate changed and the distribution of future vegetation is predicted to change as climate warms (Dullinger et al. 2004). As a result, vegetation changes may occur more rapidly (Easterling et al. 2000). In 2002 and 2003 bark beetle outbreak in the Southwest was related to drought and increased temperatures (Breshears et al. 2005).

In northern Arizona, field surveys (e.g., Shaw et al. 2005), government land office surveys (e.g., Anderson and Baker 2005), and remote sensing studies (e.g., Kadmon and Harari-Kremer 1999; Lalibert et al. 2004; Weisberg et al. 2007) have documented spatio-temporal changes in pinyon-juniper woodlands over the past century (Johnson 1962; Ffolliott and Gottfried 2002; Breshears et al. 2005; Shaw et al. 2005). Remotely sensed data in particular has been of great value for documenting

such changes, because it facilitates the spatially continuous assessment of large geographic areas, not that of discrete points in the landscape only as is the case with field surveys.

In this study, we used remotely sensed data from 1936, 1997 and 2004, as well as historic fire data to determine the spatio-temporal dynamics of pinyon-juniper woodlands in northern Arizona. In this study we used a relatively long temporal scale of 68 years and focused on the role of drought and tree die-off and fire in pinyon-juniper woodland dynamics, addressing four questions: 1) What are the historical spatio-temporal dynamics of pinyon-juniper woodlands? 2) How has the recent global climate change-type drought altered woodland cover? 3) What are the differences in the spatio-temporal dynamics of woodland expansion vs. woodland die-off? And 4) What was the impact of past fire on canopy cover?

Methods

Study Area, Image Acquisition, and Pre-Processing

Our study area was located in northern Arizona, on the north side of the San Francisco Peaks in the pinyon-juniper woodlands (Floyd et al. 2009; Clifford et al. 2008). The study area is approximately 27 km north of Flagstaff, Arizona (**Figure 1**) and was 21,429 ha in size. The average elevation was 2029 m, but ranged from 1742 m to 2476 m.

Aerial photos from 1936 were obtained from the Museum of Northern Arizona as hardcopy images. These photos were scanned at 1200 dots per inch and compiled into a mosaic with 1 meter pixel resolution and georeferenced to a 1997

digital orthophoto quarter quadrangle (DOQQ) with a pixel size of 1 meter. From 2004, high resolution multispectral QuickBird images (DigitalGlobe Inc.) were acquired, mosaicked, and georeferenced to the 1997 DOQQ. The near-infrared band (NIR) of these images was pansharpened to 0.62 m. All images were georeferenced to overlay on each other and masked to of the areas shared, or overlapped, by each image. All image processing was performed in Erdas IMAGINE 9.1. **Figure 2** shows the generalized work flow used throughout the study.

Image Classifications

The 2004 QuickBird imagery was classified using a supervised classification technique (Carmel and Kadmon 1998). Since the QuickBird image was used to determine which trees were alive after the drought of 2002, the NIR band was used in the classification. This classification determined areas of live trees, no trees (i.e., intercanopy spaces) and dead trees. To perform the classification, more than 100 signatures were collected from each image. There was no attempt to differentially classify pinyons and junipers, as both have a similar absorption in the NIR (Stimson et al. 2005). The panchromatic images (e.g., 1936 and 1997) were classified using a supervised classification method similar to the QuickBird imagery, except appropriate gray-scale thresholds were used to differentiate tree cover from intercanopy spaces (Carmel and Kadmon 1998; Anderson and Cobb 2004).

Image Analyses and Plot-Level Vegetation Change

All data obtained from image classifications were in raster format and were scaled to 1 meter pixels. While in raster format, a pixel count of each classification class (canopy cover or intercanopy) was performed, giving canopy cover across the landscape for each image. An accuracy assessment of the classification was performed on the 2004 QuickBird image. This image was used as it was chronologically closest to the date of the drought and the use of this image allowed for both pre-drought canopy cover and post-drought canopy cover to be obtained and referenced to ground plot data. To perform the accuracy assessment, the image was masked to the extent of ground plots (n = 18) that were randomly established within the extent of the images. Each ground plot was a 200 meter long and 10 meter wide belt transect, comprised of twenty 10 m × 10 m plots, with every other plot being sampled for a total of 10 subplots per plot. The center of each subplot was recorded with a Trimble Geoxplorer with sub-meter accuracy. GPS points were post-processed and exported as a shapefile into ArcGIS 9.2 using Pathfinder Office 2.90. Each sub-plot was digitized, thus creating a 10 m × 10 m polygon of the subplot. Raster images were masked to the extent of these polygons. Canopy cover was obtained from classifications of the image within the shapefile of each plot and compared to canopy cover derived from field data collected at the ground plots. The shapefiles of these ground plots were also used to examine finer detailed vegetation dynamics, and data were collected from each image at the subsequent plot.

To determine which areas in 1936 where susceptible to woodland expansion, canopy cover from 1936 was regressed against cover of ground plots in 2002. To

further determine canopy cover expansion, the proportional increase of canopy cover from 1936 to 2002 was examined.

Landscape-Level Vegetation Change Analyses

To examine landscape-scale changes in vegetation over time, the classified images from 1936 and 2004, both total and live canopies, were rescaled to 1 ha resolution, which represented percent canopy cover for each 1 ha section of the landscape. This was done by summing the amount of 1 meter pixels in a 1 ha area and converting to a percentage. Grids of 1 ha were used to compensate for the misregistration errors and minor misclassifications and gave a useful representation of the landscape as many forest and woodlands are managed at the hectare-level, and also because ecological processes occur at many scales (Allen 2007).

Rescaled images were analyzed at each pixel on the landscape. The amount of canopy lost was calculated from the total canopy cover image and the live canopy cover image as:

$$a_i - b_i = x_i$$

Where a_i is total canopy cover of 2004 at pixel i, and b_i is live canopy cover of 2004 at pixel i, and $_{xi}$ is the amount of canopy lost at pixel i. Expansion of woodlands from 1936 to 2004 was calculated as:

$$j_i / a_i = y_i$$

Where j_i is canopy cover in 1936 at pixel i, a_i is total canopy cover of 2004 at pixel i, and y_i is the proportion of change at pixel i. The 1936 and 2004 images were then

regressed against each other to determine relationships between canopy cover at each hectare in time. Additionally, the proportion of canopy lost was calculated as:

$$a_i / b_i = z_i$$

Where a_i is total canopy cover of 2004 at pixel i, and b_i is live canopy cover of 2004 at pixel i, and z_i is the proportion of canopy lost at pixel i. The 1 ha resolution analyses helped determine if the plot-level vegetation analyses using canopy cover on individual transects was consistent over the landscape.

Landscape-Scale Vegetation Patterns

To determine the overall landscape patterns from 1936 to 2004, each of the rescaled 1 ha images (e.g., 1936 cover, 2004 total cover, and 2004 live cover), were classified into seven canopy cover classes (**Table 1**). These classes represent the amount of canopy cover present and landscape metrics were used to quantify landscape patterns using FRAGSTATS 3.3 (McGarigal et al. 2002). Patch number and the metric contagion were used to determine the canopy cover class dynamics of each image and determine how the landscape has changed over time, with respect to canopy heterogeneity and canopy type.

Role of Fire on Canopy Cover

Geographic information systems (GIS) data layers of past historic fire distribution on our study area were acquired from the Coconino National Forest. These historic fire layers were clipped to our study area and the amount of area covered by fire distribution in 1936 and 2004 were calculated. An area similar in size and elevation,

which was adjacent to the historically burned areas were separately clipped and canopy cover from 1936 and 2004 was calculated for these areas as well. This was performed to determine if non-burned areas differed in canopy cover from burned sites. The amount of canopy cover in 1936 and 2004 was calculated by performing a pixel count in the area of interest. Since canopy cover data obtained for each image are population data, the data was graphed and compared graphically to determine how fire impacted canopy cover from 1936 to 2004.

Results

Accuracy Assessment of QuickBird Imagery

We determined the accuracy of the supervised image classification on the 2004 QuickBird image, using ground data from plots. From these ground plots, live tree cover and total tree cover (n = 18) was used for the accuracy assessment on both classified images (live canopy cover and total canopy cover). QuickBird imagery accurately predicted variation in canopy cover of both total ($r^2 = 0.803$) and live canopies ($r^2 = 0.806$). There was a better relationship between ground plots and live canopy cover than total canopy cover, which was due to the NIR band that allowed live trees to stand out from the background/intercanopy more easily than dead trees, which were included in the total canopy cover classification.

Plot-Level Vegetation Change

Tree canopy cover increased 32% from 1936 to 1997 (from 9.5% cover to14.1% cover). Cover between 1997 and 2004 decreased 55% (from 14.1% cover to 6.4%

cover), which we attribute to massive drought induced mortality by the pinyon ips (**Figure 3**).

When canopy cover of ground plots was compared to cover obtained from the 1936 image, 94% of these areas increased in cover from 1936 to 2002, with ground plots being surveyed in 2002. Canopy cover increased from a mean of 6.4% in 1936 to a mean of 11.1% in 2002. During the 66 year period, sites where cover increased the most originally had lower canopy cover (e.g., 1 - 5%) and sites with high canopy cover (e.g., > 25%) in 1936 increased relatively little or remained stable (**Figure 4**). This was shown when examining the proportion of change from 1936 to 2002, sites with less cover increased more than sites with higher cover (Figure 4). To determine if these patterns were consistent when using remotely sensed data, the classified 2004 QuickBird image was used to determine vegetation change at 1 ha resolution. When total canopy cover from 2004 was compared to canopy cover from 1936, there was a greater increase of canopy cover in low canopy cover sites from 1936 and high cover sites from 1936 remained relatively stable (**Figure 5**). These same patterns emerged when the 2004 classification of live trees regressed against the 1936 classification, suggesting that high cover sites in 1936 were less susceptible to canopy cover reduction and tree mortality (Figure 5).

Vegetation Changes at the Landscape-Level

Vegetation changes at the landscape-level showed similar patterns of change found at the plot-level. When the proportion of canopy cover increase from 1936 to 2004 was compared to canopy cover in 1936 across the landscape, there was a similar pattern to

the plot-level analyses. Cover increased in areas of low cover in 1936, while areas of high cover in 1936 increased less or was relatively stable (**Table 3**).

There was a significant positive relationship between total canopy cover of 2004 and canopy cover lost due to tree mortality ($P < 0.001$, $r^2 = 0.873$). While the proportion of canopy lost was significantly related to percent canopy cover, the relationship was extremely poor ($r^2 = 0.006$). Therefore, areas of high pre-drought canopy cover experienced greater canopy loss, while areas of low canopy cover incurred less loss; these areas proportionally lost a similar amount of canopy as a result of drought induced mortality.

Landscape-Scale Vegetation Patterns

Landscape metrics were used to determine how the landscape patterns, such as canopy heterogeneity, canopy type, and canopy arrangement have changed. In 1936 the landscape contained a high percentage (55 %) of no canopy cover or low canopy cover (canopy cover class 1 and 2) (**Figure 6**). Cover of the remaining classes (3 – 7) was evenly distributed at approximately 10% of the landscape. This indicates in 1936 there were many openings and areas of low woodland cover. Between 1936 and 2004, woodland expansion shifted canopy cover classes, where approximately 35% of the landscape contained classes 1 and 2, showing woodland expansion occurred in grasslands and low density woodlands. The high cover areas also increased in frequency across the landscape, especially in cover classes 5 – 7 (e.g., > 10% canopy cover). The increased percentage of these cover classes indicate woodland expansion

also occurred in areas previously inhabited by pinyons and/or junipers leading to increased woodland density.

Drought mortality and the loss of canopy shifted the landscape dominance back to cover classes 3 and 4 (2.01% - 10%), essentially thinning woodlands. Cover classes 1 and 2 increased to approximately 45% of the landscape, while cover class 7 decreased to less than 1%. Canopy mortality removed nearly all of the high density and cover areas across the landscape and shifted dominance to mid-cover classes (see Table 1 for classification scheme).

The expansion of woodland canopy from 1936 to 2004 decreased landscape heterogeneity, while drought mortality further decreased heterogeneity (Table 1; Table 2). In 1936 the landscape was a mosaic of canopy cover classes with high cover class density (e.g., patch density) relative to the woodland structure in 2004. The number of cover classes decreased precipitously from 4435 to 2763 (37%), and finally to 2132 (52%) from 1936, 2004 total canopies, and 2004 live canopies, respectively (Table 3). Drought mortality caused a decrease in canopy cover class number by 22%.

Role of Fire on Canopy Cover

The earliest recorded fire in the study area was 1954 and 6% of the study area burned during the 68 years of our study time frame. Most fires occurred at high elevations, averaging 2180 m, while the average elevation of the entire study area was 2029 m. The average elevation of the adjacent areas to fire was 2166 m and the total area was 10% of the study area. Canopy cover from 1936 to 2004 did not decrease from

historic fires (Figure 7). In 1936 there was 10.9% canopy cover in areas that burned and there was 11.7% in adjacent areas. In 2004 there was 16.3% and 21.1% in burned areas and adjacent areas, respectively. Cover in adjacent areas in 1936 was similar to cover in burned areas, while in 2004 cover of adjacent areas was nearly 5% higher than in burned areas.

Discussion

Classifications and Accuracy Assessment

The accuracy of the supervised classification of 2004 QuickBird imagery was within the typical range of similar studies and methods used (e.g., Kadmon and Harari-Kremer 1999; Anderson and Cobb 2004; Clark et al. 2004). It is important to note the accuracy of remotely sensed data will never reach 100%, as there will always be some inherent error, whether it is in the imagery, processing (e.g., rectification, mosaicking, topographic correction, scanning distortion) or GPSing of the ground reference plots.

Vegetation Change

Canopy cover of pinyon-juniper woodlands increased by 32% between 1936 and 1997, increasing from 9.5% to 14%. The highest increases of woodland cover were in areas where canopy cover was low in 1936 (1 - 5% cover) and remained relatively stable in areas with high cover (> 25% cover). This indicates there was a threshold of canopy cover in these woodlands and there was relative stability in areas with high canopy cover, while in low-density sites there was increased expansion and density of

canopy cover. This canopy cover threshold has been found in other areas of pinyon-juniper woodland (Weisberg et al. 2007). The mechanisms of this expansion and increased density in northern Arizona are unknown, but in other study areas were woody vegetation expanded into grasslands or other vegetation types, increased grazing, fire suppression, and possibly climate change influenced woody expansion (Archer 1994; Jackson et al. 2002; see citations in the introduction). Differences in microsite and topography also influence the rate of woodland expansion (Weisberg et al. 2007).

From 1997 to 2004 canopy cover decreased 55%, from total cover of 14% to 6.4% cover. There was an overall loss of 30% canopy cover from 1936 to 2004, with canopy cover changing from 9.5% to 6.6%. In 61 years there was a 32% increase in canopy cover and in less than 7 years there was a 55% decreased in cover, and when based off of our ground data, mortality occurred during a two year period from 2002 and 2003.

During the 68 years of aerial photos and satellite images (1936 to 2004), there was a shift in canopy cover distribution. In 1936, canopy cover was relegated to clumps and large patches, and the overall landscape was generally more heterogeneous when compared to cover in 1997 and total cover in 2004 (Figure 1). Long periods without disturbance events can increase homogeneity of the landscape (Pickett 1980; Baker 1992), which impacts wildlife as well as future disturbance events (Turner 1989; Turner et al. 2001). In this study, the relatively long period without disturbance events has driven a pinyon-juniper-grassland vegetation-type to a pinyon-juniper dominated ecosystem (i.e., woodland expansion). Canopy cover after

the drought of 2002 was reduced to levels lower than those occurring in 1936 and also increased heterogeneity of the landscape. In many areas, reverting pinyon-juniper woodlands back to a pinyon-juniper-grassland with scattered junipers and young pinyons. Given that fire is a major disturbance event in pinyon-juniper woodlands and while fire return intervals have been estimated at 250 - 400 years (Floyd et al. 2004), these periodic extreme droughts may serve as an alternate source to reduce canopy cover and tree density and may play a more important role than fire in landscape and regional changes in woodland cover.

Landscape-level Vegetation Patterns

When the percent canopy cover across the entire landscape was rescaled to 1 ha there was a positive relationship between the proportion of expansion from 1936 to total cover in 2004. This relationship shows there was an expansion of pinyon-juniper woodlands across the landscape. These results are consistent with other observations, which have recorded an expansion of pinyons and/or junipers in the Southwest since Anglo-American settlement (Johnsen and Dalen 1990; Yorks et al. 1994; Brockway et al. 2002; Bradley and Fleishman 2008).

There was a positive relationship between the loss in canopy and total canopy cover from 2004, but a non-significant relationship between percent canopy mortality and total cover in 2004. These results are contrary to other studies, which indicate that increased canopy cover or stand density should increase mortality (Negron and Wilson 2003; Weisberg et al. 2007; Greenwood and Weisberg 2008). High cover

areas did contain more overall mortality than low cover areas, which has thinned the canopy of the pinyon-juniper woodlands in this landscape.

The expansion of pinyon-juniper woodlands from 1936 to 2004 decreased heterogeneity of the landscape. As woodlands expanded, the overall number of vegetation cover class patches within the landscape decreased by 37%, creating a more homogenous woodland canopy structure, but drought and bark beetle mortality thinned much of the overstory and patch number decreased further. The high canopy cover areas (e.g., canopy classes 6 – 7) that increased during expansion were lost during drought, decreasing high cover patches in both number and size (Table 1). These data show that, while overall canopy cover was reduced to levels lower than 1936, the reduction occurred mostly in high cover areas. These areas were reduced to more intermediate canopy cover levels. Canopy cover in the low cover areas was not reduced back to a complete grassland as those occurring in 1936, but to a grassland-savanna with intermediate to low canopy cover. Therefore, indicating that mortality caused by this climate change-type drought thinned, but did not remove woodland cover.

Role of Fire on Canopy Cover

Fire did not significantly reduce canopy cover on our study area. Fires influenced 6% of the landscape, mostly at higher elevations. In the 6% of the landscape that was affected by fire, overall canopy cover was 5% lower than in similar adjacent areas. Canopy cover of pinyon-juniper woodlands were less impacted by fire then by bark

beetle mortality. This indicates bark beetles may have a more important role in controlling canopy cover and density than fire, especially over large spatial extents.

Ecosystem Consequences of Tree Mortality

The rapidity of the reduction in canopy cover shows how quickly climatic disturbances and associated disturbances, such as bark beetles and fire can reduce vegetation cover and thus ecosystem processes. The other comparable natural disturbance mechanisms that can potentially cause such quick and severe ecosystem changes are high severity fires, hurricanes, tornadoes and avalanches (Kulakowski et al. 2006), but these disturbances are local and do not act at the regional scale like droughts (Finney et al. 2005). The loss of approximately half of the overstory has strong implications for carbon and nutrient cycling, animal habitat, understory vegetation responses, and future disturbance dynamics (Kulakowski et al. 2003). For instance, with a reduced canopy, grass cover has increased under dead pinyons (Clifford, unpublished data). Gaps left by dead trees are being converted into grasslands, which will alter fire dynamics in drought-affected pinyon-juniper woodlands (Kulakowski et al. 2003). Fire will carry differently with a reduction in the overstory, which currently consists of a mixture of dead and live trees. Projected fall of pinyon snags shows most dead trees will fall within 15 years of mortality (Clifford et al. 2008), further altering how fire will ultimately burn and carry in pinyon-juniper woodlands. Currently, the probability of fire crowning has been reduced, but with the increase in fuel-loads on the ground from dead branches and trees as well as increased

grass production, the probability that fire will carry can be expected to increase (Clifford et al. 2008).

With a current focus on global climate change, both scientifically and politically, sources of carbon sinks and fluxes are actively monitored (e.g., Breshears and Allen 2002; Asner et al. 2003; Strand et al. 2008). It has been shown that woody plant expansion has increased carbon sequestration (Asner et al. 2003; Strand et al. 2008). While expanding woodlands may fix carbon, there has not been research showing the changes of carbon during extensive tree die-offs (but see Breshears and Allen 2002). Drought during the 1950s in the Southwest caused mortality among expanding woody shrubs (Allen and Breshears 1998; Strand et al. 2008), thus showing there are documented die-backs of these woody vegetation-types. The implications for studying carbon sequestration and fluxes in expanding woody vegetation need to take into consideration global climate change-type droughts and die-off events.

The relatively long expansion of pinyon-juniper woodlands and quick reduction in canopy cover shows the importance of climatic extremes acting as disturbances, and the ensuing ecological threshold that ecosystems may experience in the future. The results of this study show this may be true even on shorter, historical vegetation changes and timelines, rather than longer millennial-scales typically associated with vegetation change (Betancourt et al. 1991; Anderson 1993; Jackson et al. 2002). With the current rate of increasing global temperatures, these die-off events will likely increase in frequency and vegetation changes will rapidly occur over large areas during relatively short time periods.

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Tables and Figures

Table 1. Percent cover of each cover class with standard error. When woodlands expanded, there was a loss of grasslands (e.g., canopy cover class 1) and an increase in canopy density and overall cover. As woodlands died, more area of the landscape became intermediate in tree cover.

Class metric		Average vegetation cover class size (ha)					
Cover class	Percent cover	1936		2004 live		2004 total	
1	0	36.33	± 17.23	25.85	± 12.38	26.60	± 13.66
2	0.01 - 2	2.71	± 0.12	6.40	± 1.64	4.22	± 0.34
3	2.01 - 5	2.25	± 0.08	7.46	± 1.14	3.74	± 0.33
4	5.01 - 10	2.37	± 0.09	10.76	± 3.78	5.96	± 1.03
5	10.01 - 20	3.28	± 0.23	11.34	± 5.18	10.78	± 2.70
6	20.01 - 30	2.31	± 0.14	2.58	± 0.34	8.48	± 2.18
7	> 30	8.39	± 1.98	2.59	± 0.66	10.77	± 4.55

Table 2. Landscape heterogeneity decreased with woodland expansion, and die-off as measured by amount of vegetation cover classes on the landscape.

Landscape metric	1936	2004 live	2004 total
Contagion	22.79	30.07	20.48
Amount of cover classes	4435	2132	2763
Cover class density (cover class/ha)	20.70	9.95	12.89

Table 3. Spatial and temporal relationships between years show canopy cover that was lost had strong relationship to total canopy cover in 2004. There was little relationship between canopy cover in 2004 and percent mortality of the canopy. The proportional change in canopy cover between 1936 and 2004 was consistent at the landscape-level and plot-level analyses (see Figure 4).

Predictor variable (x)	Independent variable (y)	*P*	r^2	Line equation
Total canopy cover 2004	Cover loss	<0.0001	0.873	y=1.7566 + 1.4586x
Total canopy cover 2004	Canopy cover 1936	<0.0001	0.336	$Y=a(1-e^{-bx})$
Total canopy cover 2004	Percent mortality	<0.001	0.0006	y=0.5881 +-0.0005x
Proportional change in cover	Canopy cover 1936	<0.0001	0.323	$y=11.156e^{-0.2901x}$

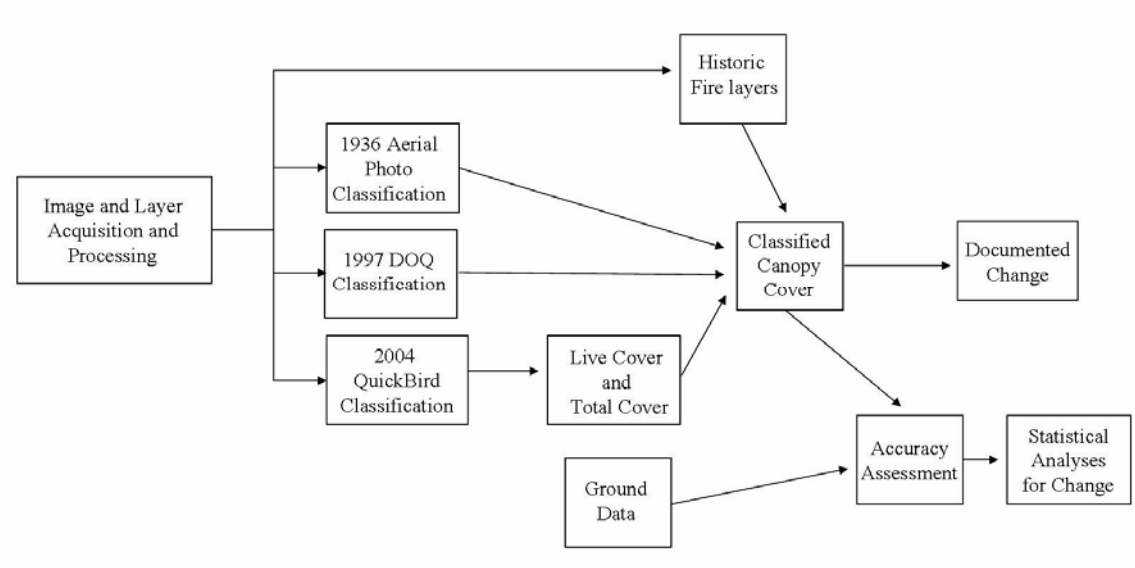

Figure 1. Study work flow framework, including GIS layer and mage acquisition, processing and analyses.

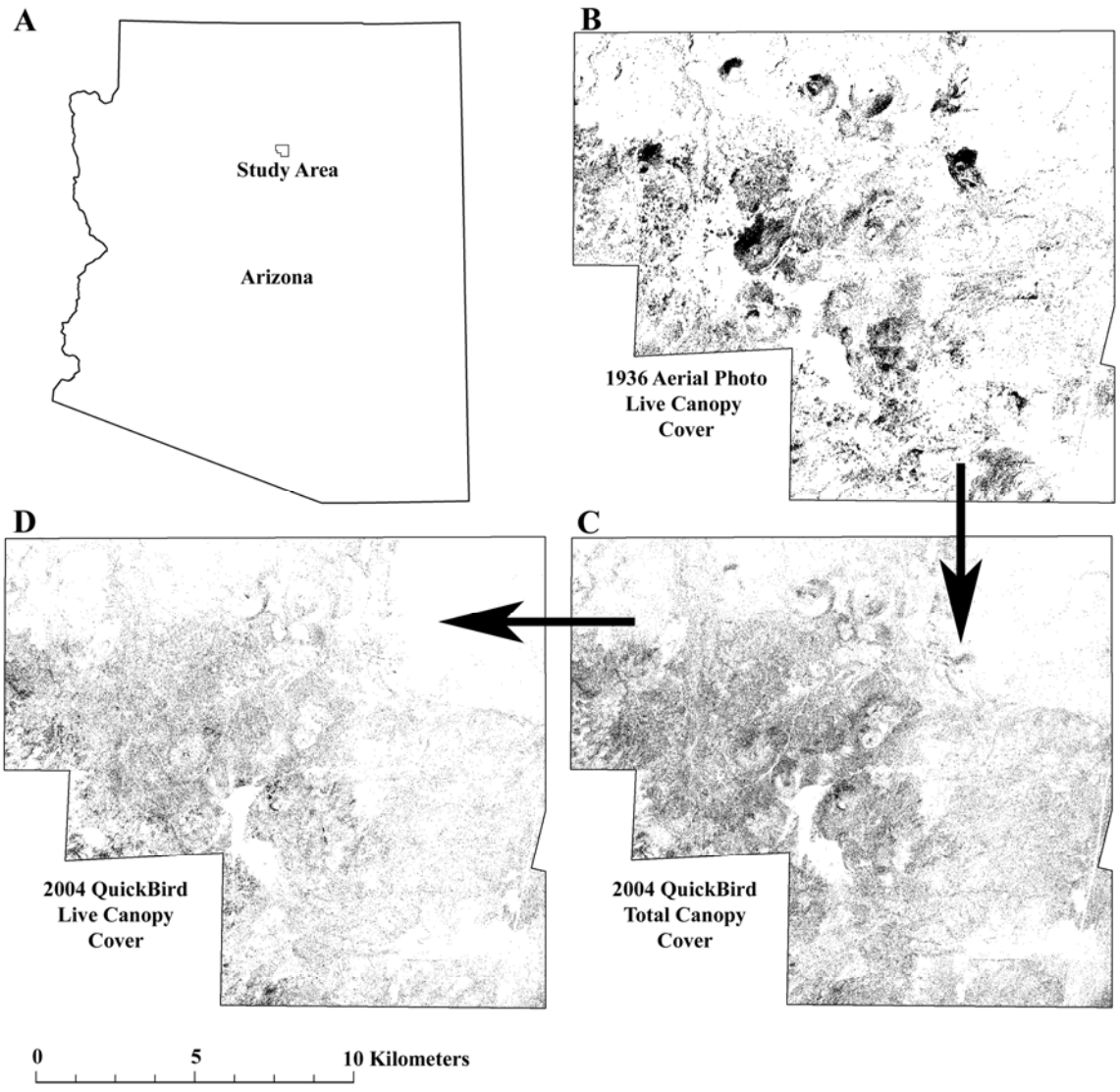

Figure 2. Change in canopy cover from 1936 to 2004. Canopy cover is represented by the dark areas, while intercanopy spaces are light areas. A) Location of study area in northern Arizona. B) 1936 classified aerial photo with 9.5% canopy cover. C) Total canopy cover from 2004 with 14% cover. D) 2004 live cover with 6.4% canopy cover.

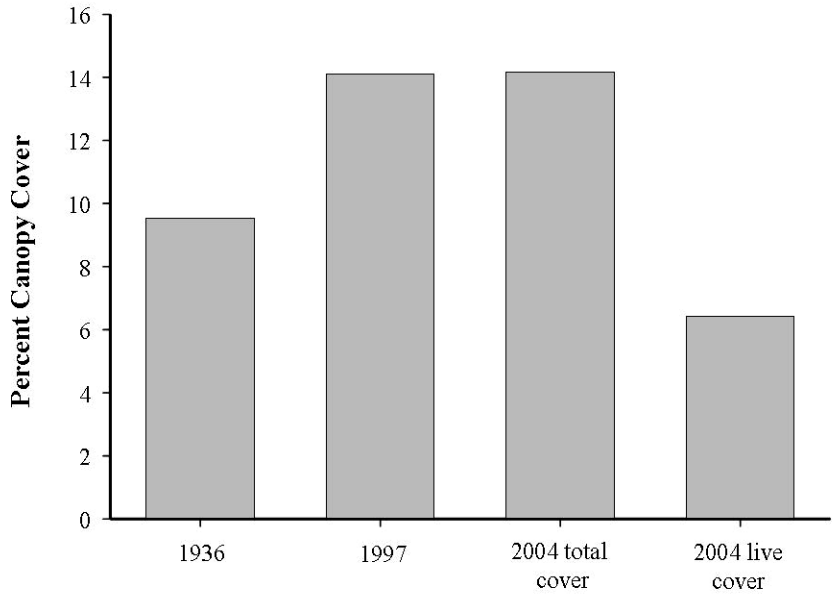

Figure 3. Change in canopy cover from 1936 to 2004 and impact of fire on canopy cover. Canopy cover increased from 1936 to 1997, and the increase can be seen in the 2004 total canopy cover image, as well, but drought and bark beetle outbreak occurred in 2002 (Breshears et al. 2005; Clifford et al. 2008).

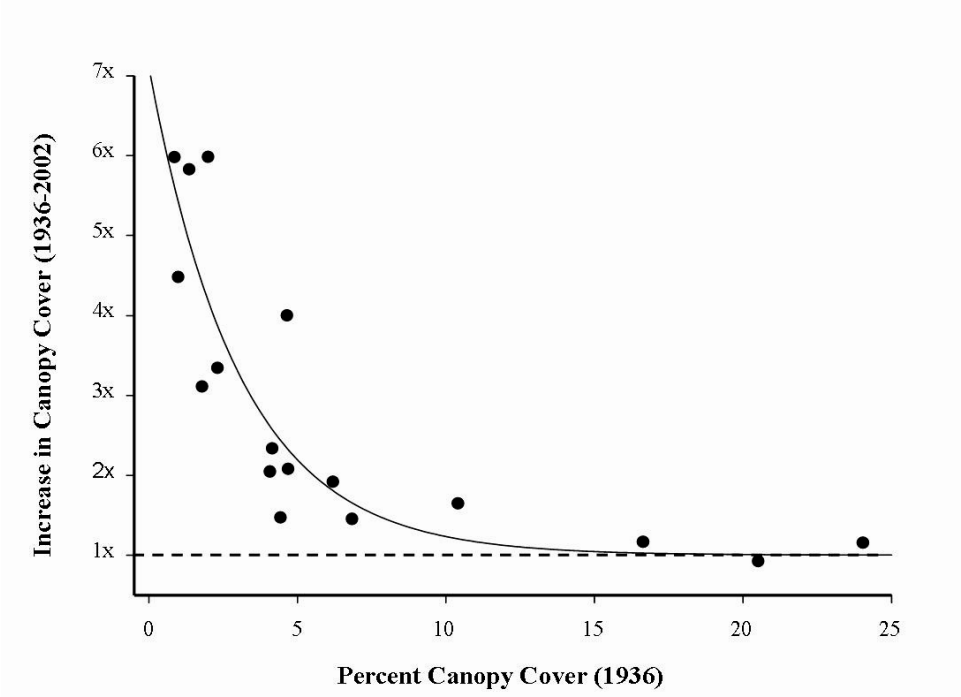

Figure 4. The proportional increase in canopy cover from 1936 to 2002 shows low cover sites in 1936 expanded at greater rates than high cover sites ($P < 0.001$, $r^2 = 0.755$). The dashed line represents where no change in canopy cover has occurred.

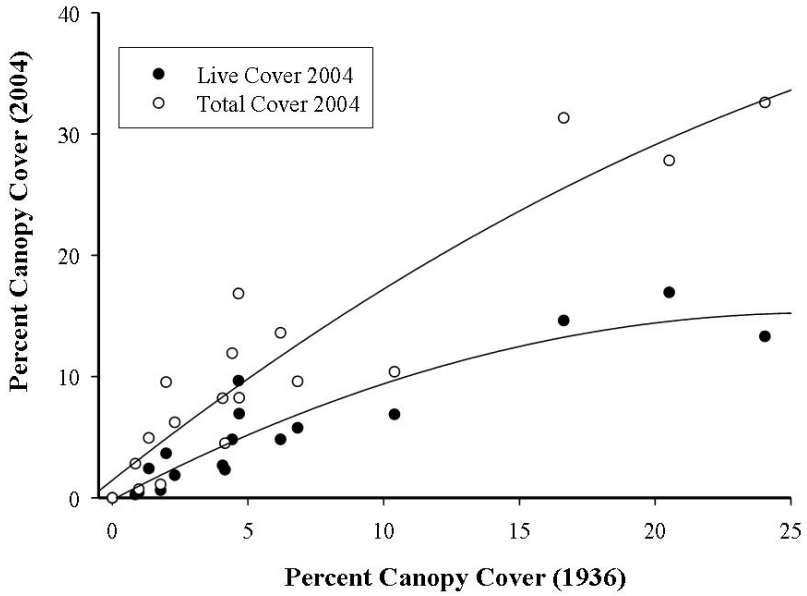

Figure 5. Relationship between total canopy cover (open circles) and live canopy cover (closed circles) in 2004 to canopy cover in 1936. Notice the curve for total cover has a greater slope in the low canopy cover, indicating these areas expanded more rapidly than high cover areas from 1936, which is consistent with ground data. Live cover of 2004 shows a similar pattern of a threshold near 15% cover.

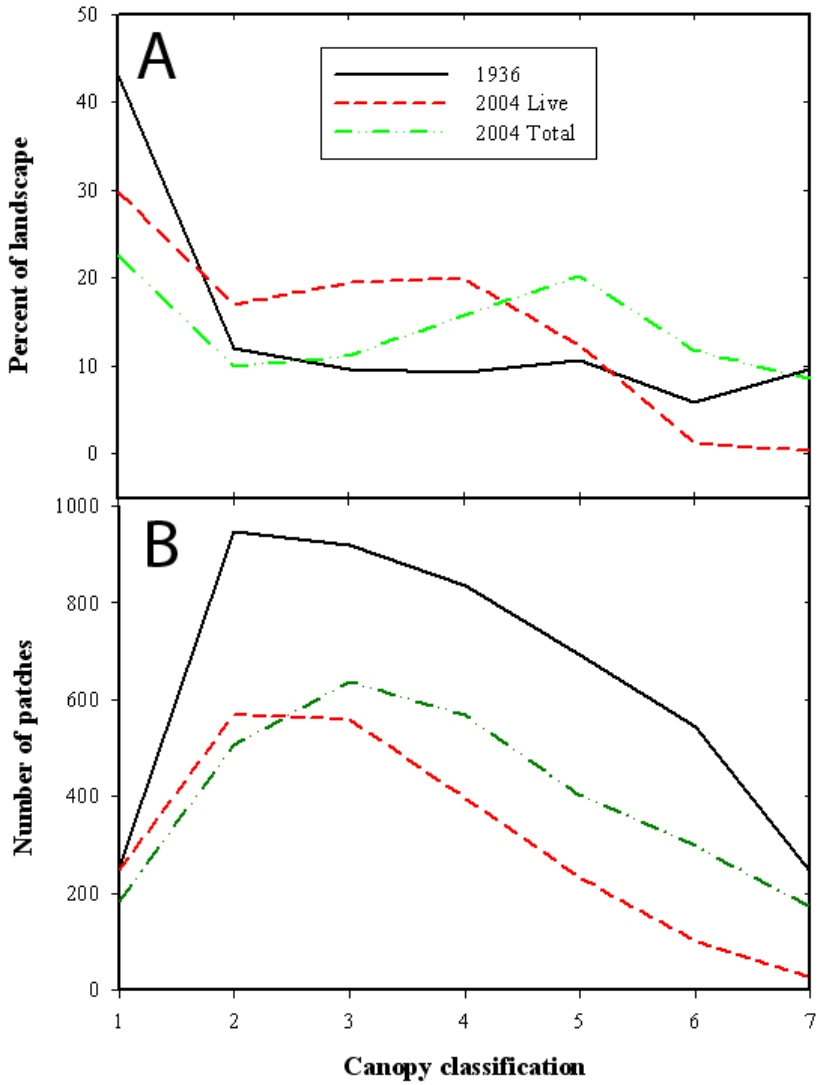

Figure 6. A) Dynamics of the landscape show decreasing cover in lower canopy cover classes (i.e., canopy classes 1 - 2), and increases in higher cover classes as woodlands expand from 1936 to 2004. When drought and bark beetles kill much of the overstory, the intermediate canopy cover classes increase, while the high cover classes decrease. B) When the woodlands expanded, there was a loss of patches across the landscape, especially in the low and intermediate cover classes (i.e., classes 1 - 5). Drought mortality further decreased patch numbers, especially in the high cover canopy classes (i.e., classes 6 - 7).

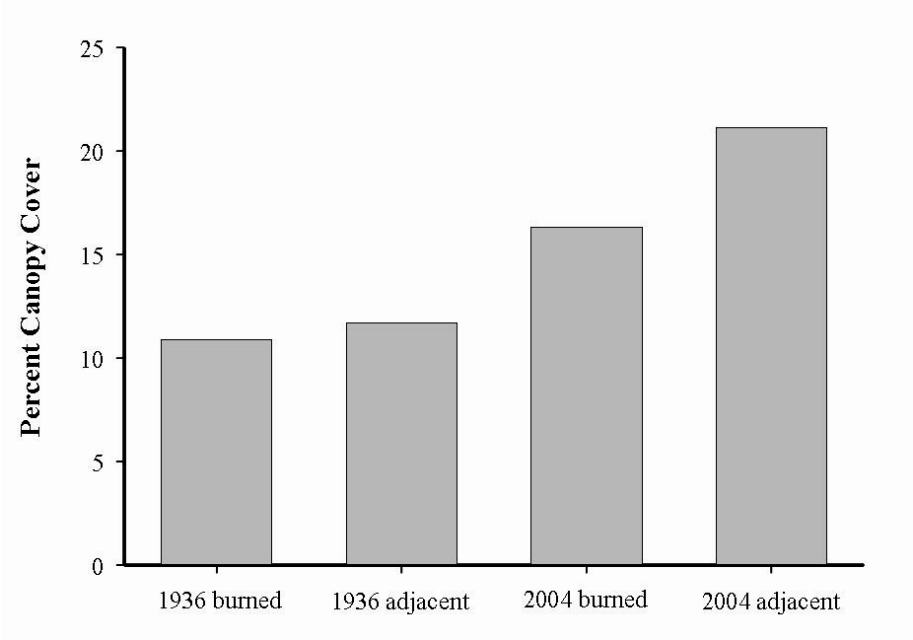

Figure 7. The impact of historic fire on canopy cover shows canopy cover increased in both burned and adjacent areas from 1936 to 2004. Adjacent areas had 5% more canopy cover than burned areas, indicating fire can influence canopy cover on the landscape. Fires covered 6% of the landscape, mostly at high elevations.

Chapter 2: Impacts of drought on stand structure in pinyon-juniper woodlands

Abstract

Extreme drought in 2002 caused high mortality among pinyon pines and to a lesser extent, junipers throughout the Southwest. This region-wide mortality event has been linked to global climate change and may serve as early indicator of future vegetation changes. Determining patterns of mortality from this event will help land managers create informed decisions about the impacts of drought in semi-arid woodlands and understand how changing climatic conditions will impact these stands. Traditional management paradigm suggests stands of high densities should be reduced to decrease tree stress during drought conditions and promote healthy forests and woodlands. This paradigm has not held under extreme drought, but other mechanisms of mortality that examine abiotic and spatial aspects of this recent mortality event have not been examined. In this study, plots were established throughout pinyon-juniper woodlands in Arizona and New Mexico to examined four questions 1) Was there differential mortality of pinyons and junipers? 2) Were there abiotic or stand level predictors of tree mortality? 3) Did tree size impact mortality? And 4) How has tree mortality altered stand structure? In New Mexico, there were several positive and negative predictors of pinyon mortality, including, elevation, shrub cover, bare ground cover, and stand density. Mortality was not predicted for junipers in New Mexico or either species in Arizona. Results from Arizona showed a positive relationship between pinyon and juniper mortality, indicating both biotic and abiotic factors influenced tree mortality. Both tree species in Arizona had relatively high

mortality among all age classes, but older pinyons were preferentially killed in New Mexico. Stands in both states were thinned, with pinyons being preferentially killed over juniper in both study sites. Post-drought recruitment of pinyons and junipers was lower than mortality levels during the same time period in Arizona.

Introduction

Disturbance events are ubiquitous in many ecosystems and are important drivers of ecosystem patterns and processes (Turner 1989). Most disturbances are minor ecosystem processes, but few of these events can be quite severe, causing major ecosystem changes and altering successional trajectories (Turner 1989; Tausch et al. 1993; Allen and Breshears 1998; Turner et al. 2001; Turner et al. 1993). In the semi-arid southwestern US, drought is a major disturbance event that can alter vegetation patterns (Allen and Breshears 1998; Mueller et al. 2005), forest and woodland dynamics and stand structure (Allen et al. 2006). Drought in the Southwest has recently occurred and persisted since the mid-1990s (Breshears et al. 2005; Shaw et al. 2005). These drought conditions cause stress to vegetation and subsequently, die-off among many vegetation types (Breshears et al. 2005; Gitlin et al. 2006). Severe drought conditions, coupled with increased temperatures during 2002 caused a major bark beetle (*Ips confusus*) outbreak among pinyon pine (*Pinus edulis*) in the pinyon-juniper woodlands (*Juniperus monosperma*) (Breshears et al. 2005).

Current climate models have predicted increasing future temperatures, both globally and regionally, as well as altered precipitation regimes. In the Southwest these models predict warmer temperatures coinciding with decreased precipitation

(Seager et al. 2007). These persistent drought conditions will further alter vegetation patterns as dominant overstory trees species continue to die-off and distributions shift. Temperature in the Southwest during the severe drought of 2002 was higher than the previous drought of similar magnitude in the 1950s (Breshears et al. 2005). Increasing temperature not only impacts plant distribution and physiology, but also affects insect physiology and phenology. Warmer temperatures have been shown to increase many herbivorous outbreak-type insect species (Swetnam and Lynch 1993; Logan et al. 2003; Bigler et al. 2006). These outbreaks occur as vegetation becomes stressed and host plant defense mechanisms are unable to deter herbivorous insect attacks. Warmer temperatures can also increase survival of insect populations leading to increased generation periods in a season (Logan et al. 2003).

In the Southwest, pinyon-juniper woodlands are a dominant vegetation type at the mid-elevations (1650 - 2300 meters). Prehistorically, these woodlands were important for native cultures as food sources and historically have been important for cattle and sheep grazing, as well as wildlife habitat. Persistent drought in the 1990s and early 2000s has caused high-levels of mortality among this vegetation type, especially in pinyon pine. Abiotic or stand-level predictors of dominant tree mortality have largely been unexplained, while stand density has been examined (Negron and Wilson 2003; Floyd et al. 2009), there are still conflicting results. Negron and Wilson (2003) found an increase in stand density index promoted mortality of pinyons in northern Arizona after the early drought event of 1996, but prior to the severe drought and majority of mortality occurring in 2002 and 2003. While Floyd et al. (2009) and Clifford et al. (2008) found increased pinyon mortality in stands of less dense

woodlands when examining pinyon-juniper woodlands at larger scales throughout the region post 2002 drought, but poor model fit indicate little or no pattern of mortality due to stand densities.

When bark beetle outbreak occurs in coniferous forests and woodlands, typically the larger, mature trees are attacked, while the small trees are ignored by these cambial feeders (Mueller et al. 2005). These trees are selected as they have enough cambial tissue to support the beetles though their life-cycles. The death of many large trees will alter the reproductive ability of the population, as well as potentially reduce smaller trees from competition, thus facilitating increased growth. With a large loss in overstory tree canopy of forested systems, early successional tree species may increase, while in the semi-arid pinyon-juniper woodland ecosystem, grasses and understory vegetation will respond positively to the reduction in overstory canopy (Clifford unpublished data; Allen 2007). As most bark beetle outbreaks are associated with environmental stress, the drought accompanying the outbreak of *I. confusus* in 2002 and 2003 which attacked pinyons, also killed many junipers. The mortality of junipers is assumed to be caused by drought stress and not bark beetles, as currently there have not been documented biotic mechanisms associated with the recent drought mortality of junipers.

There have been several studies documenting the impacts of drought and bark beetle mortality in pinyon-juniper woodlands since the drought of 1996 (Ogle et al. 2000; Negron and Wilson 2003; Breshears et al. 2005; Shaw et al. 2005; Mueller et al. 2005; Gitlin et al. 2006), but none of these studies have examined mortality and stand dynamics of pinyon-juniper woodlands at large spatial extents (> 100 km^2)

which are more relevant to management and whole ecosystems. In this study, three questions were addressed to explain causal effects of drought mortality at large-scales, these included: 1) Was there differential mortality of pinyons and junipers? 2) Were there abiotic or stand-level predictors of tree mortality? 3) Did tree size impact mortality? And 4) How has tree mortality altered stand structure?

Methods

Sites were established in north-central New Mexico in the Middle Rio Grande Basin (MRGB) and in northern Arizona on the north side of the San Francisco Peaks. The plot design was different between the study areas, but the same variables were collected and the study designs were complimentary. The Arizona study area was established from 1998 - 2001, prior to the drought, allowing us to capture the temporal components of tree mortality that occurred on this stand. The MRGB sites were established from 2005 - 2007, after the majority of pinyon mortality occurred, but the extent of this study area covered approximately 6-fold more area than the area of the Arizona study area. At each site, variables collected on each tree included: basal trunk diameter (BTD) also known as root collar diameter, crown height, crown width (by averaging two perpendicular measurements of the crown), and tree status (alive or dead). At the Arizona study area, dead pinyons were examined for signs of bark beetle attack, and since nearly all trees examined showed signs of attack, all dead pinyons encountered larger than 2 cm in BTD were assumed to have been killed by bark beetles. Dead pinyons, smaller than 2 cm, were assumed to be killed by drought or other insect herbivores, such as twig beetles, such as *Pityophthorus* spp.

and *Pityogenes* spp. (Shaw et al. 2005). All site locations were recorded with a Trimble GeoXplore3 global positioning system (GPS) unit. All GPS points were differentially corrected and post-processed using GPS Pathfinder Office 2.90 and exported as shapefiles in ArcGIS 9.2. The shapefiles of each site was used to determine the elevation, slope, and aspect which were obtained from 10 m resolution digital elevation models (DEMs). Aspect was analyzed by placing each site into one of the four-cardinal direction or a no direction category. Slope was analyzed by classifying slope steepness into four categories (0 - 5%, 5.1% - 15%, 15.1% - 25%, > 25%). Solar radiation (in Watt hours/m^2) was also derived from the DEM and used as a predictor of mortality.

San Francisco Peaks, Arizona

Fifty-seven sites were established from 1998 - 2001, located north of the San Francisco Peaks (Figure 1). Locations were randomly chosen from maps to ensure they encompassed the elevation range (1675 m to 2302 m) of the pinyon-juniper woodland, as well as a range of stand and canopy densities throughout the study area. At each location, a site, 10 meters wide and 200 meters long was established, and separated into 100 m^2 plots. Alternate 100 m^2 plots were sampled, and data was averaged from the ten, 100 m^2 plots sampled on each site. At each alternating 100 m^2 plot each tree was spatially mapped at during establishment of the site and entered into a geographic information system (GIS). Mapping of sites prior to 2002 allowed us to determine when trees died and which trees were recruited during this time. Sites were surveyed during the year of establishment and again in 2002, 2003 - 2004, and

2006 - 2007. The tree species occurring on sites included pinyon, one-seeded juniper, alligator juniper (*Juniperus deppeana*) and ponderosa pine (*Pinus ponderosa*).

Middle Rio Grande Basin, New Mexico

In pinyon-juniper woodlands throughout the MRGB, 75 sites were randomly selected and sampled during the summer of 2005 (n = 32), 2006 (n = 21), and 2007 (n = 22), within the 2674 km^2 MRGB study area in north-central New Mexico (Figure 1). Locations were chosen by the proximity to a road (> 50 m and < 1 km), and the proximity to other randomly selected sites (~ 5 km from another site). Three, 100 m^2 plots were established at each site. These plots were placed in a triangular formation, 75 meters apart. At each plot, four, 1 m^2 quadrats were randomly placed within the plot to measure cover of woodland understory (i.e., percentage of forb, grass, shrub, rock, litter, and bare ground). The purpose of these quadrats was to determine if certain understory variables could promote mortality or if there were differences in functional group cover between areas of low or high mortality. This study area was divided into three areas (north, central, and south) based on defining geographic features (e.g., mountain ranges). The elevation of sites ranged from 1673 m to 2293 m. The tree species included pinyon pine, one-seeded juniper, ponderosa pine, alligator juniper, and oak (*Quercus* spp). Sites in the MRGB received the same GPS treatment as the Arizona sites.

Statistical Analyses

To determine if there were abiotic predictors of mortality or stand attributes that could have caused higher mortality among woodland tree species, linear regressions and multiple regressions were performed to determine the relationship between pinyon and juniper mortality and abiotic or stand variables, these included: elevation, basal area, stand density, percent bare ground, and percent shrub cover, etc.. Likely external stressors were chosen that could potentially magnify effects of biotic (i.e., bark beetles) mechanisms of mortality or drought conditions. Slope and Aspect were analyzed by an ANOVA to determine how different slopes or aspect influenced pinyon or juniper mortality.

Spatial Data Analyses

Point data collected from a GPS were exported as shapefiles into ArcGIS 9.2. These point data were interpolated spatially for each study area (Arizona and New Mexico) with respect to percent pinyon and juniper mortality by using the krige function in ArcGIS 9.2. Regressions of percent mortality were used where appropriate to validate spatial interpolations. Each study area was examined for spatial clustering with respect to pinyon and juniper mortality by using Moran's I, a point pattern analyses performed in ArcGIS 9.2.

Results

Factors of Regional Tree Mortality

At the Arizona study area mean mortality of pinyon and junipers was 53% and 9%, respectively, while the New Mexico study area total mean pinyon mortality was 41%

and juniper mortality of 3% (**Table 1**). Combined tree mortality (pinyon and juniper) were 41.4% and 23.7% in Arizona and New Mexico, respectively.

There were few significant predictors when examining tree mortality in the New Mexico study area, the two variables that had the strongest relationships to pinyon mortality were stand density and cover of bare ground. Tree density significantly predicted tree mortality ($P < 0.003$; $r^2 = 0.127$), but the relationship was negative with mortality decreasing as density increased (**Table 2**). Bare ground showed a significant positive relationship to pinyon mortality ($P < 0.005$; $r^2 = 0.194$). Aside from bare ground, shrub cover was the only other positive predictor of pinyon mortality. When juniper mortality in New Mexico was examined using the same predictors used for pinyons (e.g., Table 2), there were no significant predictors of mortality.

Several abiotic factors, such as elevation, slope and aspect, were examined as predictors of mortality. Elevation did not predict mortality in the Arizona study area, but did predict pinyon mortality in the New Mexico study area, with pinyon mortality increasing at lower elevations (Table 2). In Arizona, north aspects had the lowest pinyon mortality at 37.7%, while western aspects had the highest pinyon mortality at 70.2% (**Table 3**). Other aspects had similar mortality at approximately 50%. These patterns of low mortality on the north aspect and high mortality on the west aspect are similar for juniper, but are not significantly different between aspects. Similar patterns of pinyon mortality were found in New Mexico, lowest mortality was on north aspects at 34.7% and high mortality on west aspects 48.3%. Juniper mortality in New Mexico was similar among all aspects. When slope was examined in Arizona,

mortality increased with slope, while in New Mexico flat slopes and steep slopes had similar mortality (**Table 4**). In both study areas, juniper mortality was highest on the steepest slopes, but not significantly higher.

Affects of Mortality on Stand Structure

Tree mortality in pinyon-juniper woodlands has reduced the population numbers of pinyons, and to a lesser extent, junipers. If all dead trees surveyed were considered to be alive prior to 2002, the ratio between living pinyons and junipers was 1.90:1 and 2.13:1 in Arizona and New Mexico, respectively. In 2007 the ratio of live pinyons to junipers was 0.94:1 and 1.03:1 in Arizona and New Mexico, respectively. Tree mortality was similar among age classes, based on BTD, in Arizona (Figure 2). But in New Mexico, older pinyons had significantly higher mortality than young pinyons ($P < 0.001$). Juniper in both states showed similar patterns. The overall stand structure changed in both study areas due to tree mortality, with an overall reduction in trees of all age classes. In the Arizona sites there were major losses in all ages of pinyons. A similar pattern was shown in the New Mexico sites, except mortality varied more among tree size. Smaller pinyons ad lower mortality, while larger pinyons had much higher mortality.

In Arizona 49 sites were surveyed for pinyon and juniper recruitment from 2006 to 2007. These showed 6 new pinyon seedlings (i.e., 0.80% recruitment) recruitment and 5 new juniper seedlings (i.e., 0.65% recruitment). During this same sampling period 68 pinyons (i.e., 9.3% mortality) and 9 junipers (i.e., 1.1% mortality) died. At the New Mexico sites, determining which seedling began growing after the

drought was not possible as we sampled this area post-drought, but we included all pinyons and junipers < 1.5 cm as a seedling (n = 666 and n = 181, respectively). Because mortality differed markedly in each region, seedlings were examined at the regional scale to determine if there was a difference in the amount of seedlings. There was not a significant difference between amount of seedlings in each region (i.e., north, central, and south) despite the large difference in pinyon mortality.

Temporal patterns of mortality could be determined in the Arizona study area, but not the New Mexico study area. Prior to the drought of 2002, mortality was low for both pinyon and juniper, approximately 3% (Figure 3). During the drought of 2002 and 2003 pinyon mortality increased tremendously to a mean of 53%, while juniper mortality increased to 9.9%. After the drought, mortality was still higher than pre-drought levels for pinyon (e.g., 9.3%), but not juniper (1.3%).

Spatial Distribution and Patterns of Mortality

In each study area, there were gradients of pinyon mortality (Figure 4; Figure 5). In Arizona, pinyons that were killed prior to 2002 (before the major tree die-off), were significantly clustered spatially (Moran's I; $P < 0.01$), while pinyon mortality occurring after 2002 was not spatially clustered. This spatial clustering of early pinyon mortality indicates local areas of high mortality. This was evident near Sunset Crater National Monument, a known area of high physiological stress for pinyons (Cobb et al. 1997). As the drought progressed, interpolations showed there was widespread pinyon mortality. Pinyon and juniper mortality were high and low in similar areas. These overlapping areas indicate or abiotic stressors contributed to

mortality and were magnified by biotic mechanisms causing pinyon mortality. The relationship between percent pinyon mortality and percent juniper mortality was significant ($P < 0.001$; $r^2 = 0.497$; Figure 6), thus validating the interpolations and further indicating an abiotic mortality factor during tree mortality.

Landscape-level patterns of mortality of pinyons in New Mexico show a latitudinal gradient from high mortality (e.g., 50%) northeastern area of the study site to low mortality (e.g., 3.7%) in the southern areas of the study site (Figure 5). Juniper mortality in this study area did not follow the same pattern. Low mortality was found in the northern region and higher mortality was found in the southern region, but mortality throughout the entire study area was low.

Discussion

Factors of Regional Tree Mortality

Higher average pinyon mortality was found in the New Mexico sites while higher average juniper mortality was found in Arizona sites. Factors of mortality where not consistent across study sites, but pinyon mortality was predictable in New Mexico. When the percentage of bare ground increased or there was a decrease in tree density mortality increased. Bare ground can increase soil temperature and decreasing soil moisture, thus increasing stress and susceptibility of bark beetle attack and mortality (Mueller et al. 2005). Increased shrub cover was also significantly related to pinyon mortality and has contributed to interspecies competition between pinyons and shrubs (Sthultz et al. 2007) or this could be a response of shrubs to the pinyon mortality

(Clifford, unpublished data). Since the sites in New Mexico were surveyed beginning in 2002 we are unaware of the shrub cover prior to the drought.

Increasing tree densities are considered stressful for trees and may increase the probability of high-intensity crown fires. Therefore, many pinyon-juniper woodlands have been managed for reduced density (Aro 1971; Johnsen and Dalen 1990). Negron and Wilson (2003) found increased pinyon mortality on sites with a higher stand density index in northern Arizona. Their study occurred after the 1996 drought, but before the major tree die-off event of 2002 and encompassed a spatially small and homogenous extent. When mortality across a much larger scale was examined, involving heterogeneous pinyon-juniper woodland, data indicated a negative relationship between stand density and pinyon mortality in New Mexico or no relationship in Arizona. These data show the importance for managing woodlands for extreme climatic events and not only for normal climatic scenarios and traditional paradigms may not be relevant for management. These data also indicate that high density stands should not always be considered unhealthy and densities may be high because nutrient and water availability in those areas is higher.

The abiotic factors such as slope and aspect influenced the patterns of mortality in both study areas. Aspect had similar influences on mortality in each area. The more mesic aspects had lower mortality than xeric aspects. While patterns of mortality on different slopes did not hold in both study areas. These differences may be explained by different soil types of the study areas. The Arizona study area is found mostly on cinder flows and the New Mexico study area covers many different

soil-types due to the large study are extent (see Table 2). This difference soil-type may convolute patterns of mortality (Greenberg and Weisberg 2008).

In both study areas, there has been a shift towards fewer large trees and more small trees in the population. While there may be more small trees, post-drought recruitment of both pinyons and junipers has been low compared to mortality of each species. If the ratio of recruitment to mortality continues at the current rate, pinyon-juniper woodlands will begin a conversion to juniper-grasslands and eventually to a low density savannah or grassland. The mortality of large pinyons will also affect the ability of the stand to reproduce, further impacting future cohorts of pinyon pine.

Spatial Distribution and Patterns of Mortality

The spatial distribution of both pinyon and juniper mortality was similar in Arizona. This similarity and the relationship of pinyon and juniper mortality indicate that there are abiotic possesses promoting mortality of both tree species in this area. The higher mortality among pinyons than junipers can be attributed to the outbreaking bark beetle population, along with physiological differences in water uptake and storage (Breshears et al. 1997). This relationship of mortality shows the importance of abiotic stresses in disturbances as well as the how biotic mechanisms can severely alter a stressed ecosystem.

The spatial distribution of pinyon and juniper mortality in New Mexico did not show similar patterns as the mortality in Arizona. Pinyon mortality occurred in a latitudinal gradient, with high mortality in the northeast, but decreased precipitously southward in the study area. Juniper mortality did not show these patterns, but with

overall juniper mortality at 3% in the MRGB, levels of juniper mortality may be too low to detect discernable patterns at the spatial scale and resolution of this study, and may need to be examined at a finer grain to detect local patterns of mortality (Urban 2005).

In both study areas, there were gradients of mortality among both tree species. These gradients indicate some areas are more susceptible to bark beetles than other areas. When patterns of pinyon and juniper mortality are similar for an area, such as the Arizona study area, there is evidence of abiotic factors influencing mortality, thus bark beetles may be a magnifying factor of environmental stresses involved in pinyon mortality. In the MRGB, where gradients and patterns of mortality among pinyons and junipers are different, there may be evidence for an overall less stressful environment, indicated by the low overall mortality in junipers and the relatively high density of stands (16.0 trees × 100 m^{-2} in MRGB compared to 7.2 trees × 100 m^{-2} in Arizona). The high density stands may indicate a better growing environment for pinyons and junipers.

Conclusions

Neither pinyons nor junipers in our study areas had many strong predictors of mortality, but did show several spatial patterns of mortality. In most cases, areas of high mortality did not differ from areas of low mortality (Clifford et al. 2008). The many weak statistically significant abiotic variables that predict or indicate pinyon mortality show the complexity of this recent drought and bark beetle outbreak. The spatial patterns of mortality alluded to at least a local underlying factor in mortality

and indicate the die-off event was not only from stressed trees that were attacked by bark beetles, but there were abiotic factors of mortality as shown by interpolations of pinyon and juniper mortality and the relationship of pinyon and juniper mortality in the Arizona study area. This spatial pattern was not consistent in New Mexico, but could be attributed to low overall juniper mortality in this area or the spatial resolution of this study.

The overall consequences to the ecosystem of this die-off event will likely last for centuries. Pinyon-juniper woodlands are an extensive, regional vegetation-type that incurred severe mortality throughout many areas of the Southwest (Shaw et al. 2005). In northern Arizona and north-central New Mexico, pinyon mortality averaged 53.7% and 41.2%, while juniper mortality was 9.5% and 3.3% in Arizona and New Mexico, respectively. The loss of nearly half of the codominate pinyons has structurally altered the stand.

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Tables and Figure

Table 1. Basic site descriptives of each study area.

Study area	Extent of study area (km^2)	Average elevation (m)	Stand density (trees/$100m^2$)	Basal area ($m^2/100m^2$)	Pinyon mortality (%)	Juniper mortality (%)
Arizona	528	2033	7.2	0.343	53.7	9.5
New Mexico	2674	2017	16.0	0.321	41.2	3.3

Table 2. Predictors and factors of percent pinyon mortality in New Mexico and Arizona. Reduced tree density and non-cover variables were positively related to pinyon mortality in New Mexico. There were no significant predictive variables for pinyons in Arizona or for junipers in either Arizona or New Mexico.

New Mexico				
Dependent Variable	*P*	F Value	r	r^2
Stand density (Total trees)	0.002	10.607	-0.356	0.127
Basal area	0.591	0.292	-0.063	0.001
Elevation	0.018	5.881	-0.273	0.075
Number pinyons	0.039	4.402	-0.238	0.057
Number junipers	0.774	0.034	-0.034	0.001
Juniper basal area	0.709	0.141	0.044	0.002
Pinyon basal area	0.101	2.752	0.191	0.036
Canopy cover	0.467	0.534	0.085	0.007
Percent juniper mortality	0.378	0.786	-0.105	0.011
Shrub cover	0.002	10.267	0.351	0.123
Bare ground	0.003	9.877	0.441	0.194
Rock cover	0.044	4.297	-0.308	0.095
Forb cover	0.867	0.028	0.026	0.001
Solar radiation	0.033	4.719	-0.248	0.062
Arizona				
Dependent Variable	*P*	F Value	r	r^2
Stand density (Total trees)	0.306	1.068	0.138	0.019
Basal area	0.751	0.101	0.043	0.002
Elevation	0.664	0.191	0.060	0.004
Number pinyons	0.911	0.013	0.150	0.000
Number junipers	0.173	1.905	-0.183	0.033
Juniper basal area	0.646	0.214	0.062	0.004
Pinyon basal area	0.370	0.816	0.121	0.015
Canopy cover	0.419	0.663	0.109	0.012
Solar radiation	0.081	3.167	-0.237	0.056

Table 3. Influence of aspect on mortality of pinyon and juniper in each study area. New Mexico sites were not established on aspects without direction. Sample size for each slope category is shown in parentheses, pinyon and juniper have the same sample size, so only pinyon is shown.

	Aspect				
Study Area	No Direction	North	East	South	West
Arizona					
Pinyon	49.7 (11)	37.6 * (14)	56.6 (14)	58.5 (9)	70.2 * (9)
Juniper	12.1	7.9	11.9	7.4	21.9
New Mexico					
Pinyon	NA	34.7 (25)	36.0 (12)	48.3 (25)	45.0 (13)
Juniper	NA	2.1	1.7	2.4	8.7

* Variables are significantly different ($P < 0.05$)

Table 4. The influence of slope on pinyon and juniper mortality in Arizona and New Mexico. There were no sites in New Mexico located on slopes > 25%. Sample size for each slope category is shown in parentheses, pinyon and juniper have the same sample size, so only pinyon is shown.

	Slope			
Study Area	0 - 5%	5.1% - 15%	15.1% - 25%	> 25%
Arizona				
Pinyon	42.0 * (26)	53.7 (15)	53.2 (7)	81.8 * (9)
Juniper	8.1	12.7	9.6	22.4
New Mexico				
Pinyon	54.7 (40)	19.7 * (30)	62.2 (5)	NA
Juniper	1.7	5.3	4.3	NA

* Variables are significantly different ($P < 0.05$)

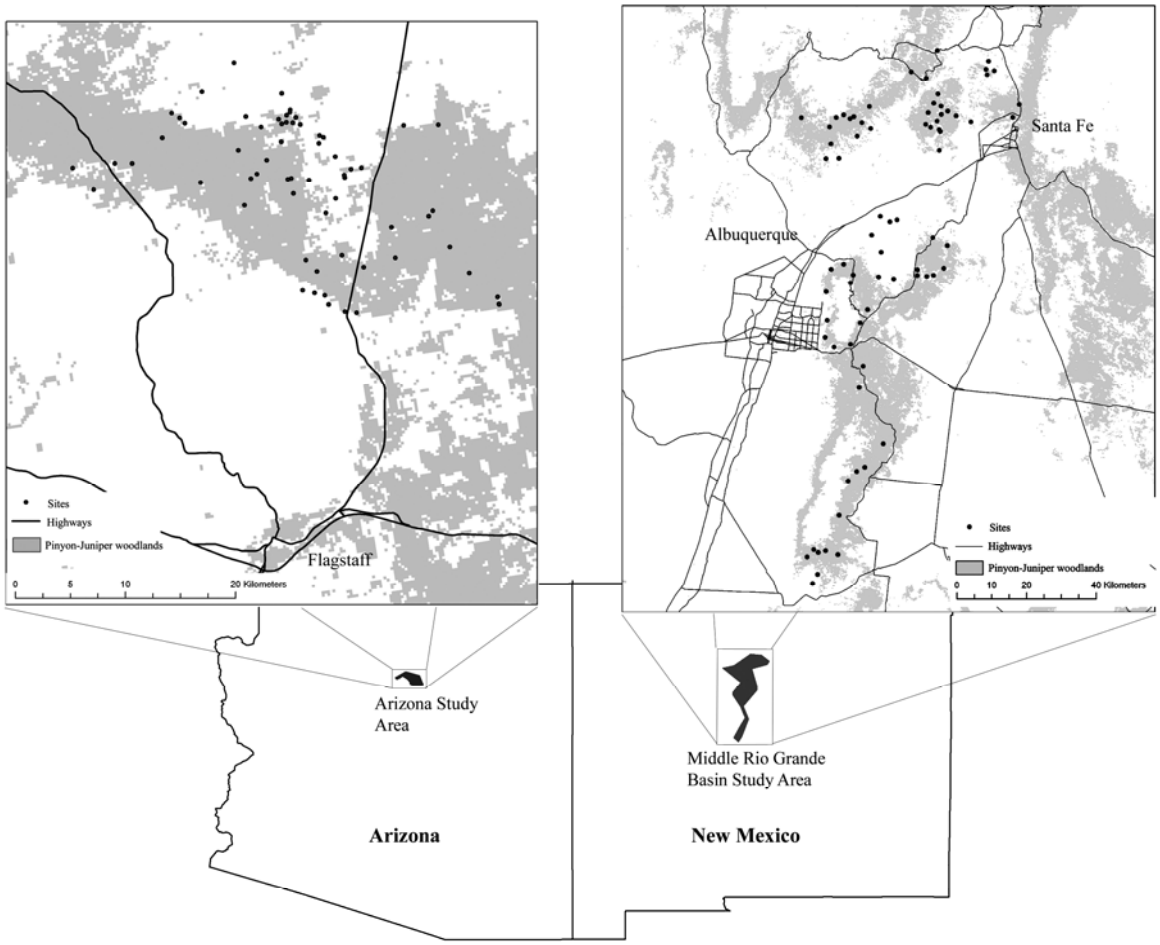

Figure 1. Map of the study areas and location of each study area relative to the other. Close-up maps of the plot locations for each study area. The shaded areas are the extent of pinyon-juniper woodlands (Southwest ReGAP data layer) and major highways are shown.

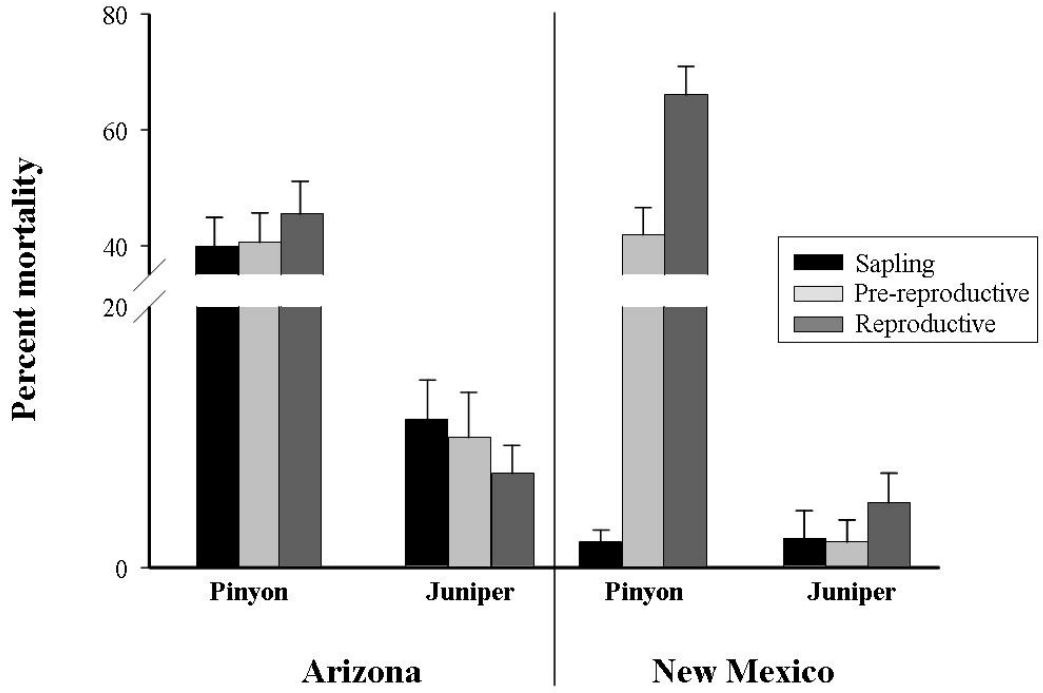

Figure 2. Mortality among tree classes was not significantly different pinyon or juniper in Arizona and for juniper in New Mexico. Older pinyons in New Mexico showed significantly higher mortality than younger pinyons. (Age classes are used from Floyd et al. (in press)).

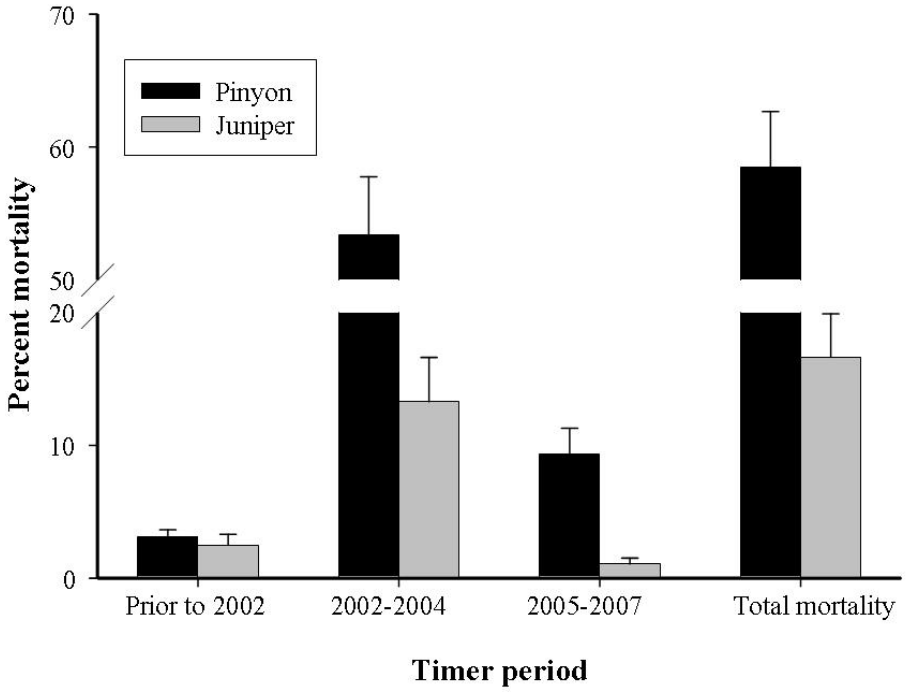

Figure 3. Temporal dynamics of pinyon and juniper mortality in northern Arizona show mortality during drought period of 2002 - 2003 was much higher than pre- and post-drought time periods. Post-drought mortality of pinyon remained higher than pre-drought levels, but juniper mortality was similar in pre- and post-drought time periods.

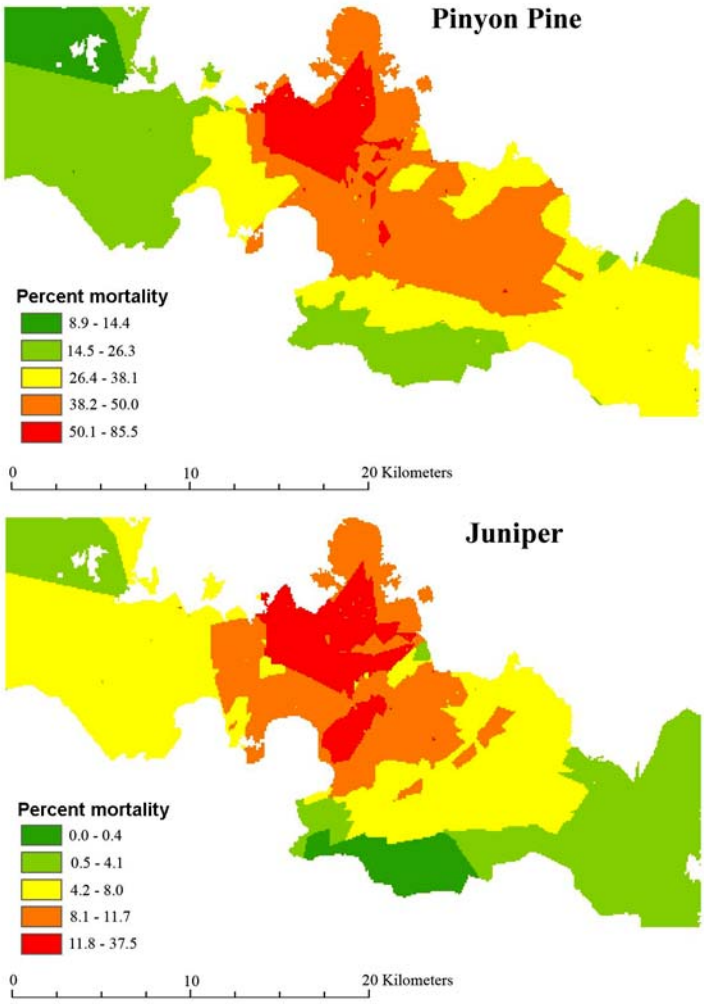

Figure 4. Interpolation maps of pinyon (top) and juniper (bottom) mortality in the pinyon-juniper woodlands in the Arizona study area. The spatial similarities among patterns of high and low mortality among both species indicate environmental and abiotic factors drive mortality. Note the scale of mortality is different among each species.

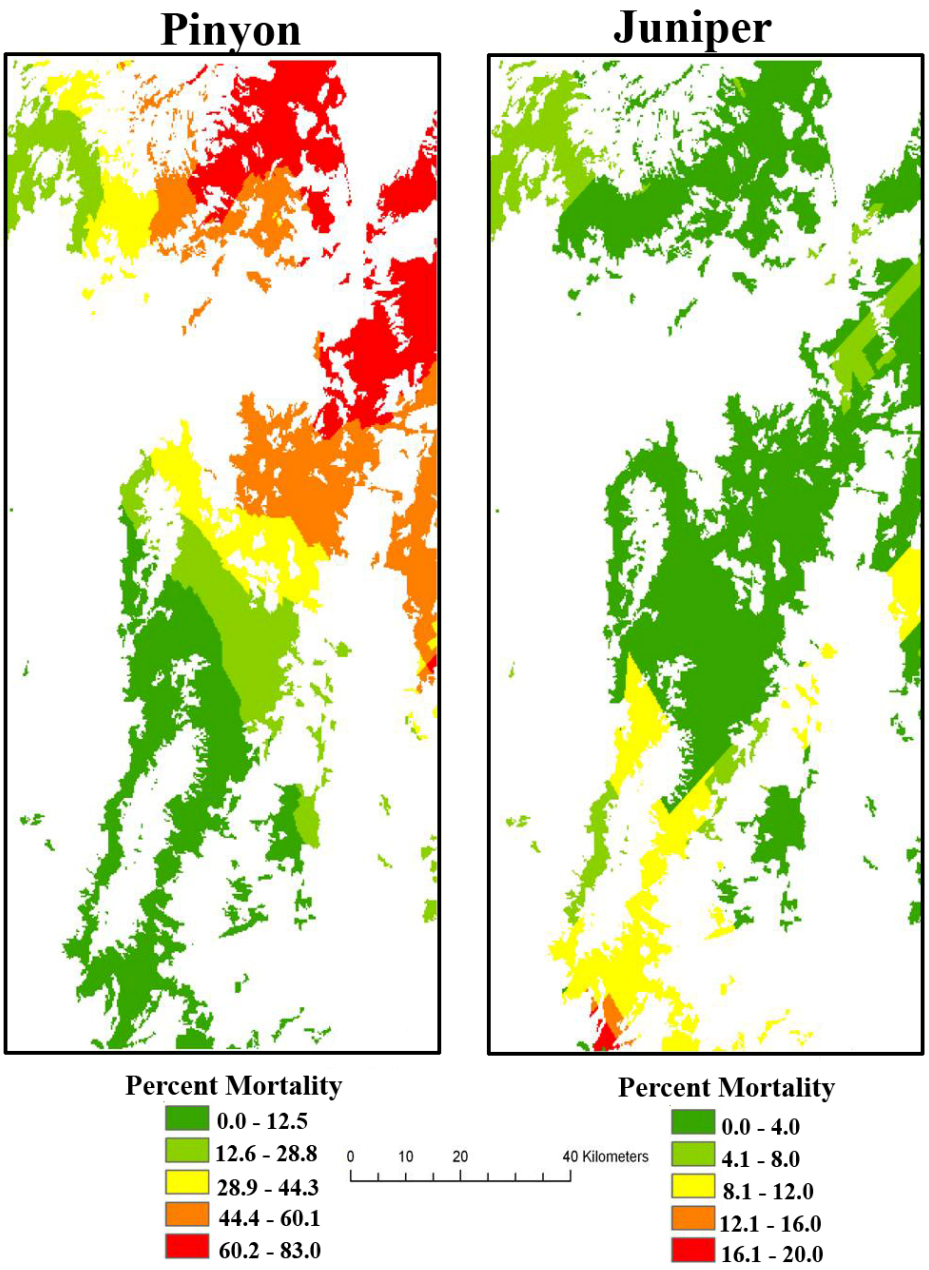

Figure 5. Interpolation map of pinyon mortality (left) and juniper mortality (right) in the pinyon-juniper woodlands of the MRGB. Map is masked to the extent of the pinyon-juniper woodland GAP data layer.

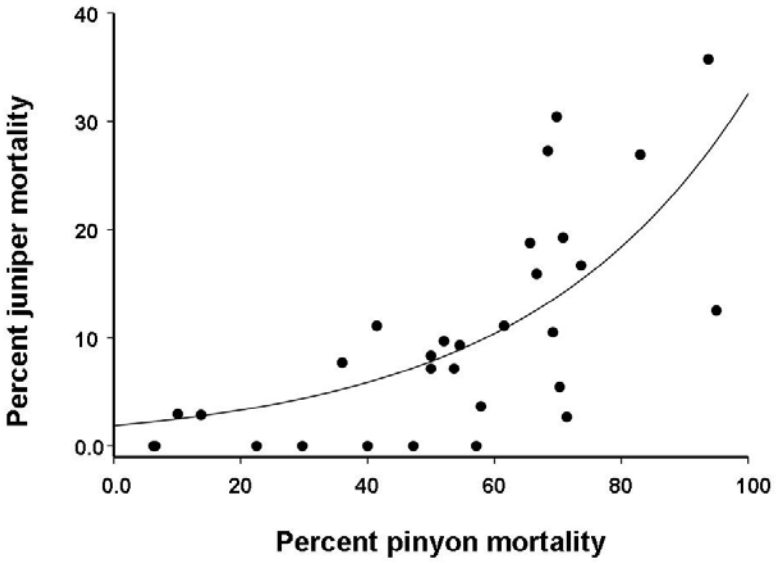

Figure 6. Relationship between percent pinyon mortality and percent juniper mortality in Arizona ($P < 0.001$; $r^2 = 0.497$). This relationship validates the spatial interpolations and shows there are environmental interactions associated with tree mortality.