

# Tree mortality from a short-duration freezing event and global-change-type drought in a Southwestern piñon-juniper woodland, USA

This study documents tree mortality in Big Bend National Park in Texas in response to the most acute one-year drought on record, which occurred following a five-day winter freeze. I estimated changes in forest stand structure and species composition due to freezing and drought in the Chisos Mountains of Big Bend National Park using permanent monitoring plot data. The drought killed over half (63%) of the sampled trees over the entire elevation gradient. Significant mortality occurred in trees up to 20 cm diameter ( $P < 0.05$ ). *Pinus cembroides* Zucc. experienced the highest seedling and tree mortality ( $P < 0.0001$ ) (55% of piñon pines died), and over five times as many standing dead pines were observed in 2012 than in 2009. *Juniperus deppeana* vonSteudal and *Quercus emoryi* Leibmann also experienced significant declines in tree density ( $P < 0.02$ ) (30.9% and 20.7%, respectively). Subsequent droughts under climate change will likely cause even greater damage to trees that survived this record drought, especially if such events follow freezes. The results from this study highlight the vulnerability of trees in the Southwest to climatic change and that future shifts in forest structure can have large-scale community consequences.

## 1 **1. Introduction**

2           Recent widespread tree mortality has been documented across the globe in response to  
3 increasingly warmer and drier climatic conditions (Allen and Breshears, 1998; Breashears et al.,  
4 2009; vanMantgem et al., 2009; Allen et al., 2010). Global-change-type droughts, which are  
5 severe droughts coupled with elevated summer temperatures, have resulted in landscape- and  
6 regional-scale shifts in forest stand structure and species composition (Breshears et al., 2005;  
7 Shaw et al., 2005). While multi-year droughts have been widely identified as agents of tree  
8 mortality (Guarin and Taylor, 2005; vanMantgem et al., 2009; Ganey and Vojta, 2011), short-  
9 duration acute droughts of one to two years in duration can also be responsible for extensive tree  
10 death (Breshears et al., 2005; Hogg et al., 2008).

11           Acute drought events that follow short-duration winter freezes can be especially  
12 damaging to plant tissue. Tree death can occur under severe drought after just a single, short-  
13 duration freezing event (Willson and Jackson, 2006). Rapid changes in temperature present a  
14 unique challenge to trees because cold snaps can cause air bubbles and sap ice to form which can  
15 result in stem breakage and hinder water transport (Scholander et al., 1961; Hammel, 1967;  
16 Sucoff, 1969; Zimmermann, 1983).

17           A five-day freeze occurred in February 2011 in Big Bend National Park, which was  
18 followed by the most severe one-year drought on record in Texas in the spring and summer of  
19 2011 (Neilson-Gammon, 2011) (Figure 1). West Texas was particularly affected by the drought  
20 (National Drought Mitigation Center, 2011), and the Chisos Basin of Big Bend National Park  
21 received just 10.9 cm of precipitation in 2011 (one fifth its historical average of 49.2 cm)  
22 (WRCC, 2012). Together, the freeze and drought events were likely responsible for widespread  
23 tree mortality between 2011 and 2012 in this region.

24 As part of a permanent forest monitoring study in The Chisos Mountains (CM) of Big  
25 Bend National Park, I monitored tree mortality in a Southwestern piñon-juniper forest between  
26 2009 and 2012. This interval overlapped the five-day February freezing event and global-change-  
27 type acute drought that occurred in 2011, providing the unique opportunity to document a  
28 coupled freezing- and drought-induced tree mortality event. While piñon-juniper tree mortality in  
29 response to severe drought has been documented in several sites in the southwestern United  
30 States, few studies have examined the combined effects of short-duration freezing and acute  
31 drought events on piñon-juniper woodland stand structure and species composition. Moreover,  
32 this research highlights tree mortality patterns across a post-Pleistocene relictual mountain range  
33 (i.e. Sky Island) that differs dramatically from other previously studied piñon-juniper forests in  
34 terms of species composition and climatic setting.

35 In this paper, I quantify tree mortality by estimating changes in forest stand structure and  
36 species composition across the forested area of the elevation gradient in Big Bend National Park.  
37 I measured changes in live and standing dead tree density, basal area, and species composition in  
38 CM as a whole and at low, middle and high elevations individually. This information provides an  
39 assessment of the combined effects of freezing and acute drought stress in Sky Island forests that  
40 are surrounded by lowland desert and whose distributions are already greatly restricted by  
41 contemporary climatic conditions.

## 42 **2. Materials and methods**

### 43 *2.1 Study area*

44 The Chisos Mountains are a small rhyolitic mountain range located entirely within Big  
45 Bend National Park. Current forests are Pleistocene relicts, and their distributions are the product

46 of species migrations from lowlands to uplands during early Holocene warming (VanDevender  
47 and Spaulding, 1979). The CM rise to 2300 m asl. They are bound at lower elevations by deserts  
48 dominated by shrub and succulent desert flora, where tree establishment and growth is inhibited  
49 due to high temperatures and moisture-limited conditions. The CM represent an ecological  
50 transition zone because of their position at the eastern edge of the Basin and Range Province and  
51 they share biological affinities with flora of the Rocky Mountains and the Sierra Madre Ranges  
52 (Muldavin, 2002). Soils are a mixture of mollisols and entisols. They are composed of  
53 moderately deep gravelly loam, which is well drained and non-calcareous (Carter, 1928). Runoff  
54 is moderate to rapid. Available water capacity is low.

55 Forests (above 1600 m asl) in CM are composed of piñon-juniper-oak, pine-oak, and  
56 mixed conifer woodlands. Piñon-juniper woodland is the dominant forest type which is  
57 comprised of Mexican piñon pine (*Pinus cembroides* Zuccarini), alligator juniper (*Juniperus*  
58 *deppeana* vonSteudal), gray oak (*Quercus grisea* Liebmann), Emory oak (*Q. emoryi* Leibmann),  
59 and weeping juniper (*J. flaccida* vonSchlechtendal) (Poulos and Camp, 2010). Lower elevations  
60 also contain small populations of one seed juniper (*J. monosperma* Englemann) and red berry  
61 juniper (*J. pinchotii* Sudworth) and oak shrublands that are dominated by *Q. pungens* Leibmann.  
62 Arizona pine (*P. arizonica* Englemann), Douglas fir (*Pseudotsuga menziesii* Mirbel), and Arizona  
63 cypress (*Cupressus arizonica* Greene) also have restricted populations in Boot Canyon in CM.  
64 Taxonomy follows Powell (1998).

65 The modern climate is arid, characterized by cool winters and warm summers.  
66 Precipitation is distributed bi-modally in late summer and winter with the majority of  
67 precipitation falling during summer storms as part of the North American Monsoon System.  
68 Mean annual precipitation for the Chisos Basin is 49.7 cm (range 10 – 135 cm). Mean January  
69 precipitation is 1.5 cm (range 0-2.5 cm) and is 8.0 cm (range 0.2 – 20.5 cm) in July. Mean

70 monthly minimum temperatures are 1.8 °C in January and 17.0 °C in July. Maximum  
71 temperatures are 14.1 °C in January and 29.1 °C in July.

## 72 *2.2 Field sampling*

73 Thirty-six plots were established at low, middle, and high elevations (12 at each elevation)  
74 in the CM in June 2009 and I resampled them during the growing season in June 2012 after the  
75 drought. Low elevation plots were randomly placed in Green Gulch within 100 m of the edge of  
76 tree cover in CM. Middle elevation plots were randomly distributed across the Chisos Basin.  
77 High elevation plots were randomly distributed along the Southeast Rim. Plots were located so  
78 that they did not intersect trails, power lines, or archeological or cultural resources. The Southeast  
79 Rim was chosen for the high elevation sampling area because it had not previously burned in  
80 prescribed fires or wildfires. Trees > 5 cm diameter at breast height (dbh) were measured using  
81 10 m radius (0.03 ha) fixed area plots. Seedlings (individuals < 5 cm dbh) were tallied by species  
82 in nested 5 m radius plots. Plot boundaries for both the tree and seedling plots were determined  
83 using a two-way ultrasonic rangefinder (Cptcam Inc., Shenzhen, China). The center point of each  
84 plot was marked with rebar and its location was recorded with a gps. Each tree was tagged with a  
85 uniquely numbered brass tree tag in 2009. I recorded the species, dbh, condition (live or standing  
86 dead), distance from the plot center and azimuth from north of each individual. Distance and  
87 azimuth measurements greatly assisted in relocating plot center. In 2012, plots were revisited and  
88 all trees from the 2009 inventory were resampled. Tree condition (live, recent snag, snag broken  
89 above dbh, snag broken below dbh, or clean snag) was noted. Trees lacking leaves or needles,  
90 with brittle and/or missing branches were classified as recent snags in the 2012 sampling interval.  
91 All recent snags were also checked for evidence of bark beetle infestation including presence of  
92 pitch tubes and beetle galleries.

### 93 2.3 Statistical analysis

94 I quantified differences in forest stand structure in 2009 and 2012 using linear mixed  
95 effects models to account for the repeated measures sampling design. I used the R Statistical  
96 Language ([R Development Core Team 2012](#)) and the lme4 ([Bates, Mächler et al. 2012](#)) and  
97 lmerTest ([Kuznetsova, Brockhoff et al. 2013](#)) packages to perform linear mixed effects analyses  
98 of the temporal shifts in forest structure and species composition from the freeze and drought  
99 events. Timestep was designated as a fixed effect. Random effects were considered for the  
100 intercept, the sample plot, and the interaction of sample plot and timestep. The residuals of each  
101 model were inspected for deviations from homoscedasticity, and only models containing  
102 residuals without obvious deviations from normality were kept in the analysis. The final structure  
103 of the fixed-effects for each model was selected by sequentially dropping non-significant terms  
104 from the full model, by measuring changes in the significance of conditional F-tests for each term  
105 ([Pinheiro and Bates 2000](#)). The intra-class correlation was also estimated for each model in order  
106 to assess the amount of variance in the response variable that can be attributed to the random  
107 effects in a model. The models describing the data most adequately were then selected using the  
108 Akaike Information Criterion (AIC) ([Akaike 1974](#)). The significance of individual sites and site-  
109 year combinations was assessed after final model selection via the F statistic using the lmerTest  
110 package.

111 I used plots as the repeated sampling unit and the sampling year as the treatment  
112 representing pre- and post-drought sampling intervals. I compared tree basal area, live seedling  
113 and tree density by species, and differences in forest size structure for the two sampling years. I  
114 also used mixed effects models to investigate how the drought affected tree populations across  
115 the elevation gradient by evaluating changes in tree density and species composition in response

116 to the drought. I evaluated the trend in tree mortality by size by performing a regression analysis  
117 comparing the percentage mortality at 1.0 cm size-class intervals.

### 118 3. Results

119 The 2011 freeze and drought killed over half (62.9%) of the trees in the sample plots in  
120 CM. The event triggered significant mortality of both seedlings and trees up to 20 cm dbh ( $P <$   
121 0.05) (Figure 2). Live tree densities decreased by approximately 100 trees ha<sup>-1</sup>. Seedlings and  
122 smaller trees were preferentially affected by the drought, while larger trees generally survived  
123 (Figures 2C-D and 3) ( $R^2 = 0.62$ ;  $F = 13.1$ ;  $P = 0.0016$ ). Over half (59.9%) of the seedlings in the  
124 monitoring plots died between 2009 ( $1059 \pm 49.8$  ha<sup>-1</sup>) and 2012 ( $428.8 \pm 34.7$  ha<sup>-1</sup>) ( $P = 0.002$ ).  
125 However, basal area also decreased significantly from  $12.38 \pm 1.75$  m<sup>2</sup> ha<sup>-1</sup> in 2009 to  $8.47.6 \pm$   
126  $1.84$  m<sup>2</sup> ha<sup>-1</sup> ( $P = 0.001$ ) in 2012 indicating that some larger tree mortality also occurred. None of  
127 the adult trees that died over the sampling interval showed evidence of bark beetle infestation.

128 The freeze and drought resulted in divergent tree mortality patterns among species. Piñon  
129 pine experienced the highest seedling and tree mortality ( $P < 0.0001$ ), and over five times as  
130 many standing dead piñon pines were observed in 2012 as in 2009 (54.5% of the piñon pines  
131 died). Alligator juniper and Emory oak trees also experienced significant declines in live tree  
132 abundance ( $P < 0.02$ ) (20.7% and 30.9% change in tree density, respectively), and alligator  
133 juniper, one seeded juniper, and Emory oak similarly experienced significant seedling mortality  
134 ( $P < 0.05$ ).

135 Tree mortality occurred across the entire CM elevation gradient (Table 1). Overall tree  
136 mortality was significant across all elevations ( $P < 0.05$ ), and mortality increased with increasing  
137 elevation (Figure 4). Piñon pine experienced significantly greater tree mortality at low elevations

138 ( $P = 0.007$ ), but otherwise tree mortality by species did not vary significantly over the elevation  
139 gradient in response to the freeze and drought.

#### 140 **4. Discussion**

141 Landscape-scale tree mortality occurred in the Chisos Mountains in response to the five-  
142 day February freeze and subsequent global-change-type drought in 2011. The effects of this event  
143 spanned the entire mountain range and affected multiple tree species. The tree mortality that  
144 occurred in response to this short-duration freezing event and one-year drought is striking  
145 because relatively few trees in CM succumbed to the longer decadal drought of the 1990s in this  
146 region (Poulos, personal observation).

147 While the individual effects of the drought and freezing event could not be distinguished  
148 from the present study, both freezing- and drought-induced xylem cavitation likely contributed to  
149 the CM tree mortality patterns due to air bubble formation from frozen sap at low temperatures  
150 (Pittermann et al., 2005; Sperry, 2011) and to the entry of air bubbles into the xylem conduits  
151 across the pit membrane under extremely negative water potentials during the drought  
152 (Zimmerman, 1983; Sperry and Tyree, 1990). Pittermann et al. (2005) demonstrated  
153 experimentally that conifers exposed to freeze-thaw events occurring in concert with drought  
154 stress had high cavitation vulnerability relative to conifers experiencing drought alone. Schaberg  
155 et al. (2008) also demonstrated that spring warming following winter freeze caused root damage  
156 that resulted in almost 100% seedling mortality in greenhouse experiments on Alaskan yellow  
157 cedar. While some have suggested that multiple freeze-thaw cycles are necessary to cause  
158 extensive damage to xylem vessels in conifers (Sperry and Sullivan, 1992; Sperry et al., 1994),  
159 Willson and Jackson (2006) demonstrated that even conifers with small tracheid diameters like



160 junipers could experience xylem embolism from just a single freeze-thaw cycle when under  
161 drought stress. While the drought may have been responsible for most of the tree mortality  
162 observed between 2009 and 2012, the visible branch splitting and bark heaving on many CM  
163 trees after the freeze (Poulos, personal observation) indicated that low temperatures during the  
164 winter of 2011 could have also contributed to tree death.

#### 165 *4.2 Preferential mortality of small trees*

166 With increasing tree size, mortality rate commonly decreases (Lorimer et al., 2001; Palahi  
167 et al., 2003). The pattern of higher mortality of smaller trees in CM was consistent with the recent  
168 die off event of *Pinus edulis* between 2002 and 2004 Arizona, New Mexico, Colorado, and Utah,  
169 although Mueller et al. (2005) observed the opposite pattern during the 1996 and 2002 acute  
170 droughts in piñon-juniper woodlands of northern Arizona. My results in the CM were consistent  
171 with the trend of high seedling and sapling mortality under drought relative to larger trees that,  
172 with their deeper root systems and larger carbon stores, were able to survive those same drought  
173 events (Mendel et al., 1997; Mueller et al., 2005; Lopez and Kursar, 2007, Ganey and Vojta  
174 2011). The lack of evidence of bark beetle infestation in trees that died over the sampling interval  
175 also suggests that the high mortality of small-diameter trees was not related to insect attack.

#### 176 *4.3 Differential tree mortality by species*

177 Although Mexican piñon pine is a site generalist in west Texas (Poulos and Berlyn, 2007),  
178 the increased mortality of piñon pine relative to other tree species was consistent with the patterns  
179 of recent mass tree mortality in the Southwest in 1996 and 2002 where piñon pine was more

180 severely affected by drought than juniper (Mueller et al., 2005; Breshears et al., 2009). Junipers  
181 are typically more drought tolerant than pines in the American Southwest (Breshears et al., 2009;  
182 McDowell et al., 2008 but see Bowker et al., 2012). So while junipers in CM did experience  
183 significant mortality from the 2011 drought, they were probably less affected than the piñon pines  
184 because of their higher drought hardiness.

185         Emory oak was also significantly affected by the drought, and large stands of this species  
186 were completely killed in CM. Although, Southwestern oaks can survive over two months of  
187 severe moisture stress under experimental conditions (Poulos et al., 2007; Ehleringer and Phillips,  
188 1996), little is known about the mechanisms of oak drought and freezing tolerance in the  
189 American Southwest (but see Neilson and Wullstein, 1985; Davis et al., 1999). Oaks in this  
190 region likely display considerable variation in drought and freezing tolerance, but their large  
191 tracheid diameters may have led to greater freeze-induced cavitation vulnerability relative to  
192 other tree species (Davis et al., 1999). Emory oaks experienced lower mortality than piñon pines  
193 and junipers in this study, yet, there remains a need for more information about the range of  
194 variability in oak drought tolerance mechanisms in the Southwest as they represent a major  
195 component of Madrean Sky Island systems.

#### 196 *4.4 Shifts in forest stand structure and species composition*

197         Although the mortality event will undoubtedly provide new nesting sites for cavity-  
198 nesting birds in CM, the higher mortality of smaller trees, the loss of over half of the piñon pines  
199 in my monitoring plots, and the death of piñon pine and entire stands of Emory oak across all  
200 elevations could result in major shifts in forest stand structure and species composition. Since  
201 2011, CM has moved out of the drought and is experiencing normal temperature and precipitation

202 levels. The return to normal climatic conditions could have a positive effect on surviving trees by  
203 releasing them from competition for moisture and bolstering their survival potential in  
204 subsequent droughts (Bowker et al., 2012) since water use efficiency in piñon-juniper woodlands  
205 can be associated with stand density (Lajtha and Getz, 1993). Nonetheless, surviving trees in CM  
206 may have experienced permanent losses in xylem conductivity in 2011, which could result in  
207 delayed tree mortality (i.e. Bigler et al., 2007) or predispose them to succumb to future acute  
208 droughts, especially if these events are coupled with winter freezes. While many piñon pines  
209 survived the 2011 drought, future global-change-type droughts could shift CM species towards  
210 dominance by junipers and more drought-tolerant oaks.

#### 211 *4.5 Mortality patterns across the elevation gradient*

212           The pattern of increased tree mortality with increasing elevation was surprising and  
213 contradictory to other prior landscape-scale accounts of drought-induced tree mortality (Allen  
214 and Breshears, 1998; Gitlin et al., 2006; McDowell et al., 2009) and canopy dieback (D. Schwilk,  
215 2013, unpublished data). The increased tree mortality at higher elevations in CM is probably  
216 related to the southerly exposure of the high elevation plots that were located on mesas of the  
217 southeast rim at the edge of high elevation forest cover and the exacerbation of the drought  
218 effects by the February freeze-thaw cycle. While high elevations of CM are cooler and more  
219 humid than low elevations, the South Rim is exposed to high incident solar radiation due to its  
220 southerly aspect, as well as high winds and temperature fluctuations because it forms the southern  
221 edge of forest cover where the rim drops from 1981 m asl down to the desert floor. Higher  
222 elevations also probably experienced the lowest temperatures during the short-duration freeze  
223 event in 2011, although cold air drainage also contributes to low temperatures at low elevations  
224 (Schwilk, unpublished data). This may have stimulated greater damage to high elevation trees

225 through freezing-induced xylem cavitation in high elevation trees which may have led to higher  
226 mortality during the course of the drought.

## 227 Conclusion

228 The results from this study demonstrate the impact of freeze-thaw events followed by  
229 drought on Sky Island forest stand structure and species composition. Future acute drought events  
230 are likely to occur with greater frequency as global mean temperatures rise in the coming  
231 decades, and the climate becomes more unpredictable (Jentsch et al., 2007). Subsequent droughts  
232 are likely to cause even greater damage to trees that survived this record drought in Texas,  
233 especially if future drought events are coupled with severe freezes. Although I documented  
234 significant rapid tree mortality in CM over the study period, lagged tree mortality is likely.  
235 Delayed mortality has been observed elsewhere in response to severe drought (Pederson 1998;  
236 Bigler et al., 2007), since damage to water transport tissue can occur over multiple years (Tyree  
237 and Sperry, 1988; Hanson and Weltzin, 2000) and because tissue damage can also predispose  
238 trees to subsequent mortality from beetle infestations (Allen and Breshears, 1998). The dramatic  
239 tree die off in CM in response to just one year of abnormal climatic conditions highlights the  
240 need for long-term forest monitoring and studies that predict the effects of future climatic  
241 extremes on Sky Island forests of the American Southwest.

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## **Table 1** (on next page)

mortality by elevation

Table 1: Changes in live tree density ( $\text{ha}^{-1}$ ) between 2009 and 2011 in the Chisos Mountains of Big Bend National Park, Texas. Values are reported as means ( + S. E.).

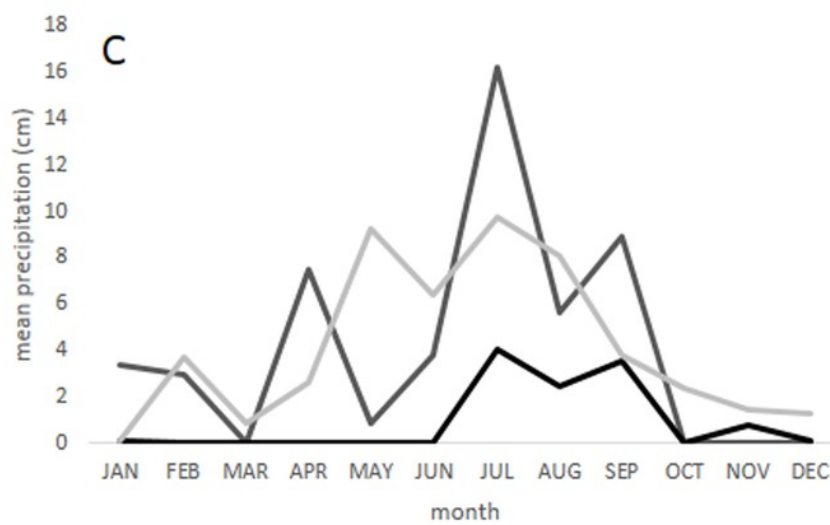
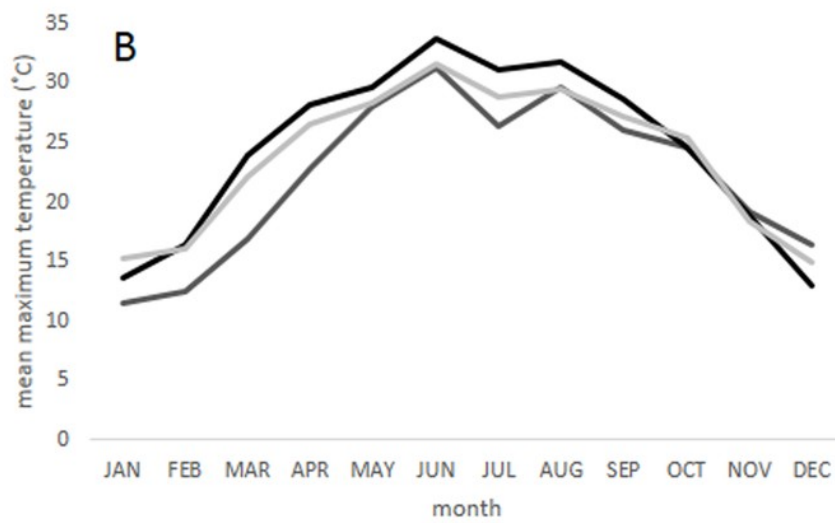
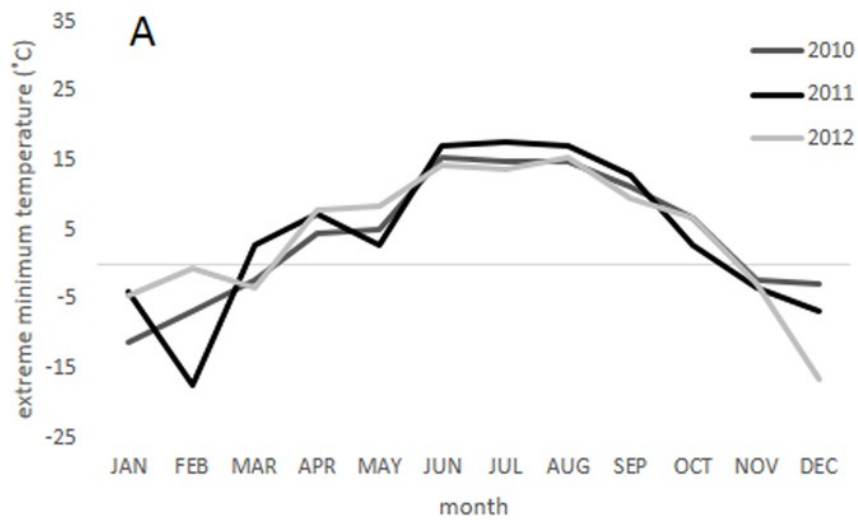
Table 1: Changes in live tree density ( $\text{ha}^{-1}$ ) between 2009 and 2011 in the Chisos Mountains of Big Bend National Park, Texas. Values are reported as means ( $\pm$  S. E.).

elevation	live trees pre drought	live trees post-drought	change in live tree density
Low	236.6 $\pm$ 41.6	146.7 $\pm$ 29.1	127.3 $\pm$ 23.9
middle	605.1 $\pm$ 100.0	483.3 $\pm$ 100.0	132.7 $\pm$ 60.8
High	748.4 $\pm$ 142.0	502.0 $\pm$ 150.0	296.2 $\pm$ 82.3

# Figure 1

## Chisos Climate

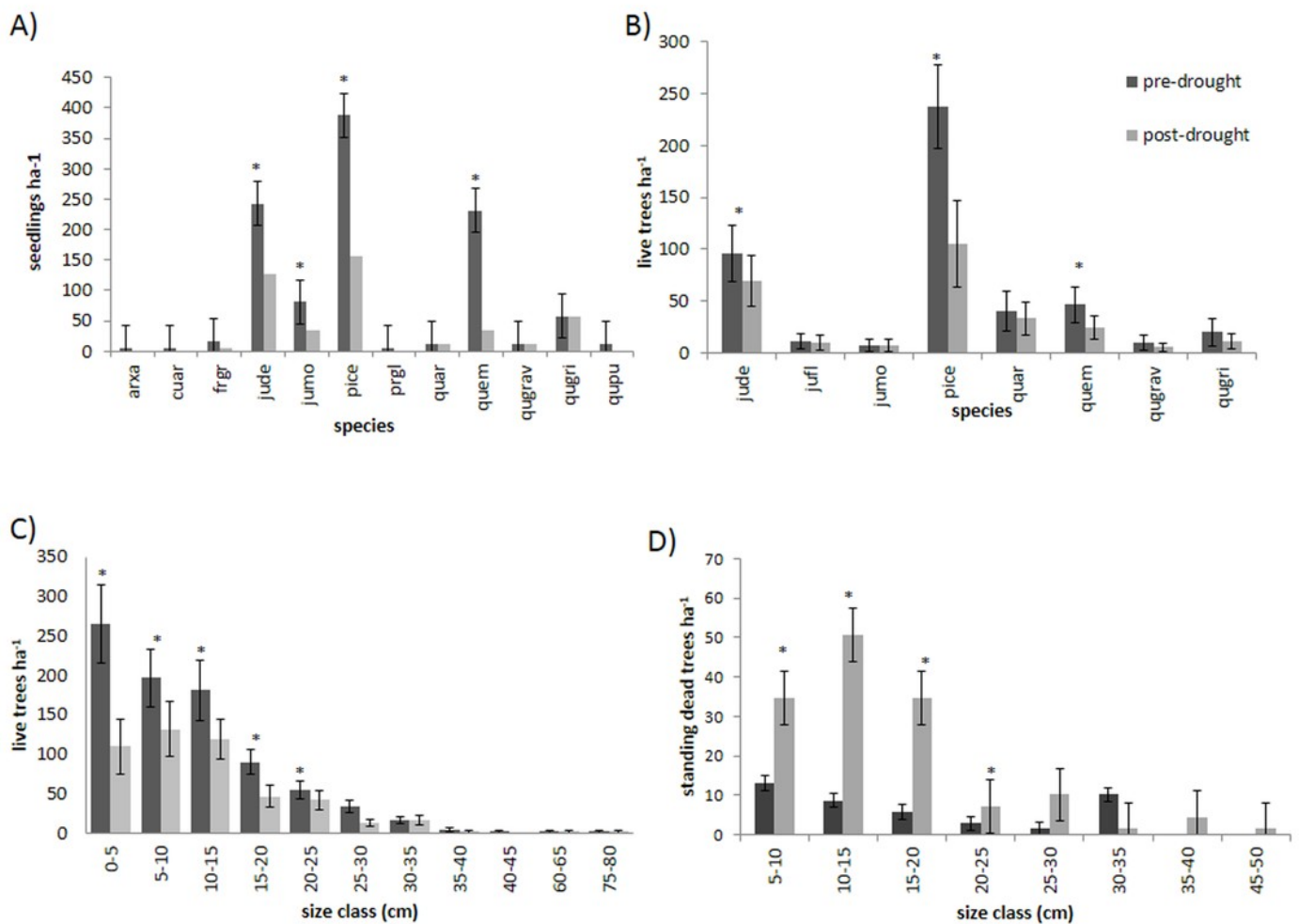
Climatic conditions from 2010-2012 in the Chisos Basin of Big Bend National Park, Texas (WRCC 2013) including A) monthly extreme low temperatures, B) mean monthly maximum temperatures, and C) mean monthly precipitation. The weather station is located within 0.25 km of the middle elevation sample sites in this study.



# Figure 2

## Stand Structure Change

Changes in forest stand structure due to drought and freezing in 2011 in the Chisos Mountains, Big Bend National Park, Texas. Mean values ( + S. E.) prior to the drought (2009) and after the drought (2011) are shown for A) seedlings by species, B) live trees (> 5 cm dbh) by species, C) live trees in 5 cm diameter classes, and D) standing dead trees. Significant changes between sampling intervals ( $P < 0.05$ ) are indicated with an (\*).

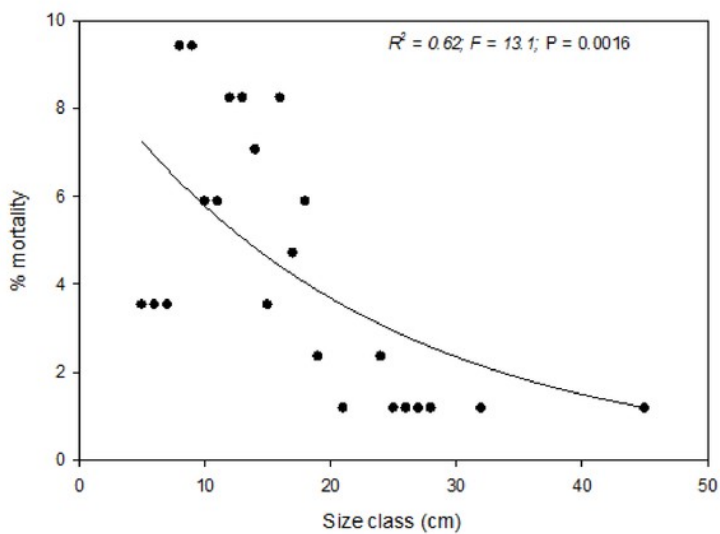




# Figure 3

size mortality regression

Regression of tree dbh (cm) as a predictor of percentage tree mortality. Percentage mortality was significantly ( $P = 0.0016$ ) correlated with tree size ( $y = 9.9538e^{-0.062x}$ ). Smaller trees suffered 2 to 5 times higher mortality than larger trees.



# Figure 4

mortality by elevation

Changes in mean ( + SE) live tree density ( $\text{ha}^{-1}$ ) at low, middle, and high elevations of the Chisos Mountains, Texas. Significant changes between sampling intervals ( $P < 0.05$ ) are indicated with an (\*).

