INFLUENCE OF DISTURBANCE, SOILS, AND SOCIO-ECONOMIC CONSTRAINTS ON RESTORATION IN BRUSH ENCROACHED, SEMI-ARID TEXAS RANGELANDS

A Dissertation

by

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ABSTRACT

Reducing brush has been a major goal of rangeland restoration because of increases in densities of native woody plants in recent decades. However, land managers attempting restoration are faced with diverse socio-economic and biophysical variables that interact to drive community shifts following interventions in encroached rangelands. I implemented a complementary set of studies to explore the drivers of vegetation response to disturbance. I designed an experiment to determine the relative resilience, or capacity to absorb disturbance without switching to an alternate state, of the woody dominated state in a brush encroached rangeland across soils with different textures: fine clays to coarse sands. I exposed plots on each soil type to one of three brush removal treatments: untreated control, hand-cutting with herbicide application, and roller-chopping. In addition, I mapped grass basal areas to determine differences in mortality, recruitment, and species turnover resulting from brush removal treatments.

Despite widespread application in the study region, the two brush reduction methods assessed in this study were not ubiquitously effective at overcoming the resilience of the woody plant dominated state. On sandy soils woody plants quickly regained pretreatment levels of dominance. However, on clay soils, grass remained dominant for the duration of the study, suggesting that, both cut-herbicide and mechanical treatments overcame shrubland resilience. This finding provides a baseline for prioritizing restoration strategies, allowing managers to target underlying conditions that are more conducive to restoration with lower levels of intervention. In addition, I

found perennial grass mortality to be higher in mechanically treated plots on all soil types than in chemically treated plots, suggesting cut-herbicide might be favorable to mechanical brush control for avoiding undesirable herbaceous community compositional shifts. Finally, recognizing the potential for social barriers to prevent adoption of ecologically effective interventions, I analyzed the effect of prescribed burning regulations and liability standards on prescribed fire use on private lands. Limited liability standards coupled with strict regulatory requirements increased the use of prescribed fire. This information can be used to formulate prescribed burning legislation that will promote the safe and effective use of prescribed fire for restoration and conservation of fire-dependent ecosystems.

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CHAPTER I

INTRODUCTION

An increase in the density of native woody plants, woody brush encroachment, has been occurring worldwide in recent decades (Smit 2004, Van Auken 2009). As a result, reducing brush densities has been a major goal of rangeland restoration in these systems because of the loss of grass production associated with high levels of encroachment (Smit 2004). However, land managers attempting restoration are faced with diverse socio-economic and biophysical variables that interact to drive community change in encroached semi-arid rangelands. Additionally, legal and economic constraints limit management options available to managers for dealing with bush removal and grassland restoration (Lambin et al. 2001, Fisher et al. 2008). However, even with unrestricted access to all management options, information regarding community response to different restoration techniques at a scale useful to practitioners is lacking. While community responses to brush removal and management have been studied in many systems, response variability across sites with different topoedaphic characteristics and land-use histories is not fully understood. Biotic and abiotic influences, as well as regulatory and liability limitations need to be assessed to develop effective management plans specific to the ecosystem characteristics of particular sites. In addition anthropogenic influences and climate change have altered disturbance regimes (Folke et al. 2004). In some rangelands, overgrazing leads to chronic low levels of fuel that decrease fire frequency (Asner et al. 2003). More infrequent fires allow for an increase in shrub density as fire sensitive seedlings exploit increasing fire-free periods to develop into more fire resistant adults. Ongoing disturbance regime alterations will continue to complicate vegetation community structure and composition responses to management for brush reduction. Often a simple return to historic disturbance regimes will not be sufficient to restore rangelands from degraded shrub-encroached states, rather, a more extreme intervention could be required.

Multi-scale assessments are needed to understand and predict vegetation shifts occurring as a result of management activities and changes in climatic extremes and human behavior. Community response to disturbance and management examined at the system level can miss variation resulting from more fine-scale distributed drivers (Zimmermann et al. 2010). Specialized responses to management occur as a result of differences in soil types, species-specific plant traits, density dependent relationships among species, legacy effects resulting from different historical land-use patterns, and the specific type and magnitude of intervention employed (Archer and Stokes 2000, Roberts 2004, Taylor et al. 2012). Therefore, developing targeted management plans that are based on an ecological understanding of the system requires knowledge of system response to management at scales ranging from the single plant level, where biotic drivers influence important demographic variables that drive population dynamics and community structural and compositional change, all the way to the state level, where policy influences management decisions. Interactions among these drivers that influence plant community trajectories at different scales necessitate a focus on multiple spatial and temporal scales that encompass variability in a suite of ecological and social characteristics (Bestelmeyer et al. 2006). In addition, disturbance regime alteration resulting from global climate change and human actions affects plant community dynamics. Therefore, additional knowledge relative to the potential for novel disturbance regimes, such as flash drought, to drive changes in plant community structure and functioning is pertinent to the development of effective management interventions that meet land-management objectives.

During the course of my doctoral research program, I implemented a complementary set of studies, thoroughly detailed in the following chapters of my dissertation, to explore the biophysical and socio-economic drivers of vegetation response to disturbance.

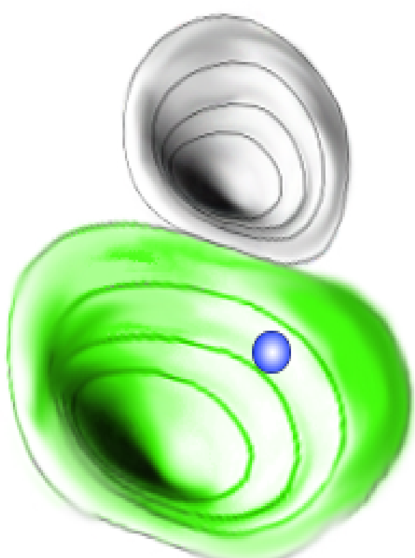
CHAPTER II

SEMIARID SHRUBLAND RESILIENCE VARIES ACROSS SOIL TYPES: IMPLICATIONS FOR OPERATIONALIZING RESILIENCE IN ECOLOGICAL RESTORATION

Introduction

In ecosystems with multiple stable states, restoration from one state to an alternative, more desirable state requires overcoming the resilience, or capacity to remain in the current domain of attraction (Fig. 1), of the existing state (Carpenter and Cottingham 1997, Gunderson and Holling 2002). Yet, operationalizing the resilience concept in restoration ecology has proven difficult (Nyström et al. 2008). This is because effective restoration requires practitioners to know the amount of disturbance or management intervention necessary to trigger a shift to an alternate ecosystem state (Standish et al. 2014). Therefore, the utility of the resilience concept has been limited in practice by the inability for scientists to quantify thresholds associated with management actions and ecosystem transformability (Suding et al. 2004, Briske et al. 2006). However, quantifying thresholds is a major challenge. Thresholds are not static and can shift as a function of the interplay among complex ecological relationships operating across various spatial and temporal scales, many of which are not readily apparent to the observer (Peters et al. 2004, Bestelmeyer 2006). For these reasons, resilience continues to be viewed as a vague concept that is difficult to apply in developing restoration strategies (Bennett et al. 2005, Groffman et al. 2006, Suding and Hobbs 2009).

An alternative to quantifying ecological thresholds is to ascertain the relative resilience of an ecological state across identifiable and measurable ecosystem properties (Scheffer and van Nes 2007, Lindenmayer et al. 2008, Slocum and Mendelssohn 2008). Coupled with determinations of the location and extent of the identifiable ecosystem property across the landscape, practitioners can use studies that determine differences in the relative resilience of an ecological state across an environmental gradient to map the resilience of an ecosystem. They can then use spatial resilience in restoration to target conditions contributing to lower resilience of undesirable states, with the knowledge



Ecosystems can have alternate stable states resulting from multiple sets of self-perpetuating feedbacks (Gunderson and Holling 2002). These are represented in the figure as two basins (gray and green). System dynamics of the alternate states are different and maintain the state unless sufficiently large perturbation causes a state transition Schroder et al. 2005). The current location of the system in state space is represented by the blue ball in the figure. A perturbation to the state variables in the system can push the system into a new basin of attraction or simply move the system toward a threshold (where the two basins meet). If the threshold is not crossed, system dynamics will attract the ball back toward the center of the basin it currently occupies. Ecological resilience, or ecosystem resilience is the amount of perturbation an ecosystem can withstand before a transition to an alternative state (Holling 1973). It is represented as the width of the basin (Peterson et al. 1998).

Fig. 1 Conceptual diagram of ecological resilience in a system with two alternate stable configurations.

that management interventions are more likely to be successful in those areas (Wallington et al. 2005). In rangelands with alternative grassland and woody vegetation states, for example, knowledge of the topoedaphic conditions that decrease the ability for invaded woody patches to absorb disturbance and retain shrubland dominance would provide a basis for prioritizing restoration efforts. Broad-scale application of mechanical and chemical treatments for restoring grass dominance to degraded woody-encroached rangelands is extremely cost-prohibitive (Taylor et al. 2011, Twidwell et al. 2013b). Therefore, operationalizing resilience to provide a basis for prioritizing intervention efforts and increasing the efficiency of restoration is a critical need.

The objective of this study is to investigate differences in the relative resilience of a woody-dominated degraded rangeland state across soil conditions in order to provide information regarding the spatial resilience of the rangeland for practitioners to use in prioritizing restoration efforts. To test for differences in resilience, I expose a semiarid south Texas shrubland that occurs across multiple soil types, representing a range in texture from coarse sand to fine clay, to commonly employed mechanical and chemical brush control methods. Brush control methods are used in this area with the intent of exceeding the ability of the shrubland state to absorb disturbance and transform it to a grassland state with a distinct set of organizing structures and functions. Brush control with mechanical and chemical treatments can therefore provide a means of identifying relative differences in the resilience of this semiarid shrubland across soil conditions. I then use fire as a follow-up to mechanical and chemical treatments to assess whether communities closer to a tipping point, following initial perturbation with chemical and mechanical brush control, can be moved into a new basin of attraction with less intervention effort. Rangeland resilience depends on vegetation structure (Walker and Steffen 1993, Carpenter et al. 2001, Anderies et al. 2002). Therefore, woody plant cover can be used to measure the resilience of rangelands to perturbation (Walker et al. 1997, Carpenter et al. 2001). A rapid return to pre-disturbance structural configurations following brush control is evidence that the resilience of the pre-disturbance state has not been overcome and that self-perpetuating processes and structures of the alternative, desired state have not been established (Allen et al. 2005). Alternatively, a long-term shift away from pre-perturbation conditions suggests that the resilience of the prior state has been overcome and the system has shifted into a new basin of attraction (Folke et al. 2004, Allen et al. 2005, Slocum and Mendelssohn 2008). I can therefore infer differences in resilience across soils of different textures based on whether the shrubland rapidly returns to its pre-disturbance structure or remains in the new grass-dominated basin of attraction. This experimental approach provides a foundation for scientific studies to inform restoration practitioners of the relative resilience of ecosystem states across different underlying environmental conditions, without necessitating that threshold dynamics be quantified for the resilience concept to be usefully applied.

Methods

Study Area

This research was conducted at the Chaparrosa Ranch (29 lat,-100 long), in Zavala county in southwest Texas. The site is subtropical with hot summers and mild winters. Average annual rainfall is 560 mm, bimodally distributed with peaks in spring and fall (Jacoby and Meadors 1983). The system is comprised of two alternative stable states, a *Prosopis-Acacia* shrubland and a grassland state. The dominant species present in the shrubland state include mesquite (*Prosopis glandulosa* Torr), blackbrush acacia (*Acacia rigidula* Benth), guayacan (*Guaiacum angustifolium* Englem.), twisted acacia [*Acacia schaffneri* (S. Watson) F.J. Herm.], and whitebrush [*Aloysia gratissima* (Gillies & Hook.) Troncoso]. All of the dominant shrub species have the capacity to vegetatively resprout in response to disturbance. The grassland state is comprised primarily of warm season perennial tufted bunchgrasses. The dominant species are curly mesquite [*Hilaria belangeri* (Steud.) Nash], hairy grama (*Bouteloua hirsuta* Lag.), three-awn (*Aristida purpurea* Nutt.), and tanglehead [*Heteropogon contortus* (L.) P. Beauv. ex Roem. & Schult.] . The plots were established in *Prosopis-Acacia* shrublands on three different soil types in three different pastures. The three pastures included in the study experienced differing land-use histories representative of different historical land-uses in the region. Historical land uses in the pastures include brush management with herbicide application in one pasture which was also periodically grazed, high intensity low duration grazing in another, and no reported brush management with periodic moderate grazing in the third (Mattox 2013). All three pastures were subject to periodic moderate grazing during the course of the study.

Experimental Design

To test for differences in resilience across soils, I selected three soil types common in the study area which represented a range of soil textures from fine clays to coarse sands: Antosa-Bobillo sand association (ABC), Webb fine sandy loam soils (WEB), and Chacon clay loam soils (CKB; Soil Survey Staff 2013). Three pastures were identified that included each of the three soil types. Brush control methods were randomly assigned to 40m X 25m plots within each pasture-soil combination. This resulted in a randomized complete block design with three brush removal treatments (control, cut herbicide, and mechanical) replicated twice in each pasture-soil combination (Fig. 2). In control treatments no brush removal occurred. In cut-herbicide treatments, I cut all woody brush at the base of the plant and sprayed a 15% Remedy (Dow AgroSciences LLC, Indianapolis, IN) herbicide/diesel mixture on the stumps of the cut trees and shrubs. I used roller-chopping with a Pasture Aerator (Lawson Mfg. Inc., now RanchWorx, Palm Harbor, FL) in mechanically treated plots. Roller-chopping uses a cylindrical drum equipped with blades towed behind a tractor to cut and crush woody vegetation at the soil surface (Fulbright et al. 1991, Blanco et al. 2005). I chose roller-chopping rather than other mechanical methods for this study because, together with the cut-herbicide and control treatments, it provided a gradient of soil disturbance with high disturbance from roller-chopping, low soil disturbance from cut- herbicide treatment, and no soil disturbance in control plots. Both the roller-chopping and cut herbicide methods are commonly used for woody brush control and removal in the study region (Welch 2000).

Two years following mechanical and chemical applications, prescribed fires were conducted to attempt to move the system into a new basin of attraction after being pushed closer to the tipping point separating shrubland and grassland states. The *a priori* thought process was that mechanical and chemical treatments may be insufficient, by themselves, to surpass the resilience of the degraded shrubland state and move into an alternative grassland state, and fire may provide the additional push needed to meet this

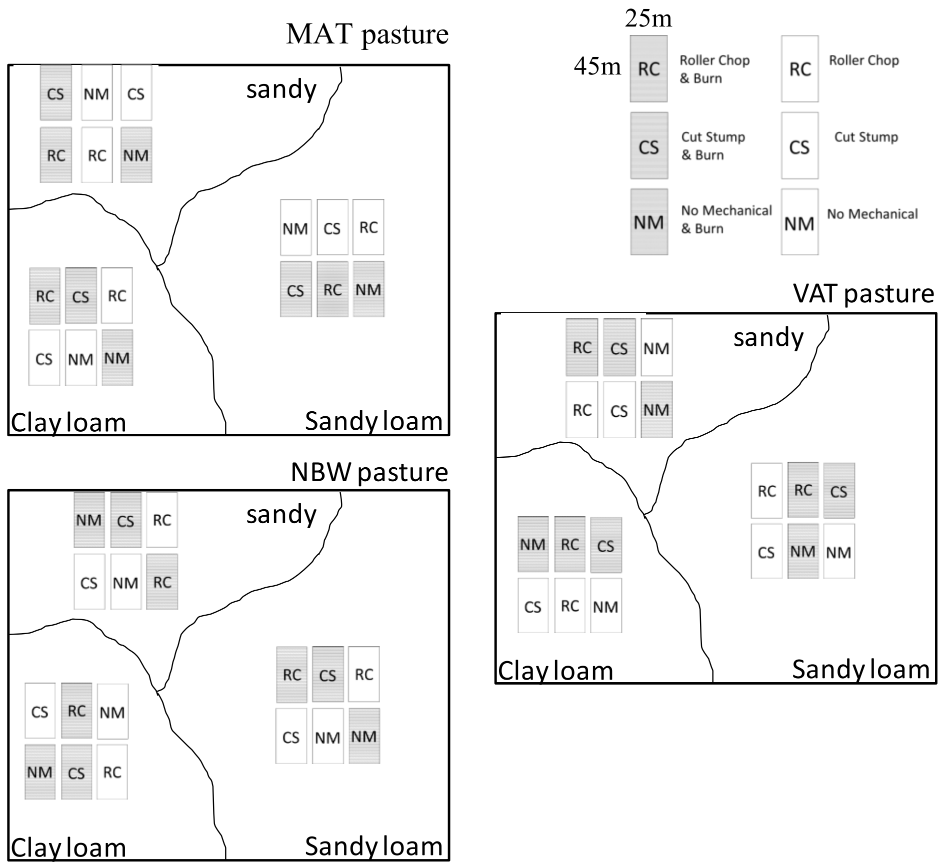


Fig 2. Conceptual diagram of experimental design.

restoration goal. Within each pasture-soil block, a prescribed fire treatment was randomly assigned to one of each brush removal treatment plots (control, cut herbicide, and mechanical). The treatments in 2013, replicated and balanced across site and soil groups, are therefore: 1. control (no brush removal and unburned), 2. burned (no brush removal), 3. cut herbicide (unburned), 4. cut herbicide and burned, 5. mechanical brush removal (unburned), and 6. mechanical brush removal and burned. Each plot was separated from the others by vegetated buffers.

I collected data on woody plant cover by species and height class (<0.5m, 0.5-1.5m,>1.5m), number of stems and individuals of each woody plant species, total herbaceous cover, total litter cover, and percent bare ground once each year during peak perennial grass production from 2010 (pretreatment) until 2013. Cover of woody plants, herbaceous, litter, and bareground was visually estimated for each quarter of the plot and averaged across quarters to determine total plot cover of each class. Five ($1m^2$) subplots were established within each plot to measure fuels using a fixed area method (Mueller-DomBois and Ellenberg 1974) to determine fuel structure and fine-fuel loading prior to burning. I visually estimated the percent cover and measured the depth of cured grass fuel. In addition, I visually estimated cover and depth of fine and coarse woody debris within each subplot for each fuel class (Fosberg and Schroeder 1971). I collected fuel data every 6 weeks during the period between application of brush removal treatments in fall 2010 and setting the fires in February 2013. I clipped, dried and weighed all fuels in twenty $1m^2$ quadrats outside of the experimental plots in order to calibrate visual biomass estimates for each subplot with harvested biomass measurements. Fire temperatures at each subplot were recorded during the prescribed burn using ceramic tile pyrometers painted with 10 temperature-indicating lacquers (OMEGALAQ® Liquid Temperature Lacquers; Omega, Inc.) that melted from 79˚C to 640˚C. Percent scorch was visually estimated immediately following the burns for each subplot.

Data Analysis

Repeated measures analysis of variance (ANOVA) and nonmetric multidimensional scaling (NMDS) were used to test whether shrubland states moved into an alternative basin of attraction after implementing brush control treatments or, alternatively, quickly recovered following treatment and therefore remained in the same state. ANOVA tested for differences in total woody plant cover, woody plant cover for each of the three height classes, percent herbaceous cover, and percent bare ground among soil types in a randomized complete block design. Soil type (ABC, WEB, and CKB) and pasture were the blocking variables. With pasture modeled as a random effect because the pastures represent a random sample of all potential land-use histories. Mechanical treatment (control, cut herbicide, and mechanical) was a fixed effect applied at the plot level. I compared all response variables among soils and treatments by estimating least square means with a Tukey's adjustment for post-hoc multiple comparisons for each year of the study (Maxwell 1980, Toothaker 1993). Nonmetric multidimensional scaling (NMDS) with Bray-Curtis distances was used to explore differences in the woody plant community composition among soils and brush removal treatments. I used site scores for the first two axes of the NMDS for each plot as dependent variables in repeated measures multivariate ANOVA to determine if the treatments resulted in significantly different woody plant communities, I also included soils and the interaction of brush removal treatments by soil types as terms in the analysis to explore treatment differences across soil types. Additionally, I calculated the relative abundance of the 8 most dominant woody plant species for each treatment on each soil type in 2010 and 2013. I used ANOVA to determine if changes in relative abundance between 2010 and 2013 differed among treatments and soils.

Tukey's post-hoc analysis on pretreatment (2010) data was used to assess whether shrubland woody plant cover differed across soil types prior to initiating treatments. I used NMDS for year 2010, prior to treatments to determine if shrubland community composition differed at the start of the experiment. I tested for changes in relative abundance of the most dominant woody plant species with ANOVA for randomized complete block design using the change in relative abundance from pretreatment to final sampling period as the dependent variable.

To test whether communities close to a tipping point following initial perturbation could be moved into a new basin of attraction, I conducted repeated measures ANOVA for years 2012 and 2013 with a similar model that included fire (burned, unburned) and its interaction with brush clearing treatment as fixed dependent variables. I set $\alpha=0.05$ to determine significance in all analyses. I evaluated the difference in percent scorch, temperature, and fuel loading among treatments on different soils with ANOVA for complete randomized block design with treatment as a fixed effect and soil and pasture as blocking variables. Visual cover estimates and height measurements of live herbaceous and 1hr, 10h, and 100h dead fuels from the 20 calibration quadrats were regressed against plot biomass determined through oven-drying and weighing all clipped materials. The fitted regression was used to estimate fuel loading for all subplots. All analyses were performed using the R statistical computing package (R Development Core Team 2010).

Results

Differences in Resilience Across Soil Types

Total woody cover differed among brush removal treatments, across soils, and across years (Table 1). Total woody plant cover did not differ among treatments prior to treatment (Tukey's HSD: control v. cut herbicide, p=0.093; control v. mechanical, p=0.0.267, cut herbicide v. mechanical, p=0.439). It did, however, differ among soils in 2010. Pretreatment cover was higher on clay CKB soils than sandy ABC soils or sandy loam WEB soils, which did not differ from each other. (Tukey's HSD: CKB v. ABC, $p<0.001$; WEB v. ABC p=0.872; WEB V. CKB p=0.002). Total woody cover decreased in the first year following brush control treatment on all soils. However, it began to recover to pretreatment levels in the second year following treatment on coarser sandy ABC and WEB soils, indicating that the community did not enter a new basin of attraction (Fig. 3). By 2013, woody brush cover on cut herbicide plots had returned to within 10% of controls on ABC and WEB soils. Similarly, cover in mechanically treated plots was within 10% of controls for ABC soils and had had completely recovered in WEB soils. In contrast, total woody brush cover on both mechanical and cut herbicide plots on the heavier clay CKB soils remained lower than cover on control plots by ~35% for the duration of the study (Fig. 3).

There was a significant treatment*year interaction for all three woody brush height classes (Table 1). In the large height class, cut herbicide and mechanical treatments dropped to 0% cover following treatment and remained low for the duration of the study. There was a slight increase in cover of the large height class in 2013 on mechanical plots as some shrubs had already attained this class on all soils but cover still remained substantially below that of controls 3 years following clearing on all soil types (Fig. 3). There was no significant difference in medium woody cover for ether cut-herbicide or mechanically treated plots on ABC soils in the final year of sampling (Tukey's HSD for 2013: cut-herbicide v. control, p=0.22, mechanical v. control, 0.18) or for the mechanical treatment on WEB soils (Tukey's HSD 2013: mechanical v. control, p=0.26). There was a significant but very small (<5%) difference between control and cut-herbicide plots on WEB soils in the final year of sampling (Tukey's HSD 2013: cut-herbicide v. control, p=0.043). On CKB soils, however, medium woody cover remained ~25% lower in cut herbicide and mechanically treated plots in the final year of the study (Fig. 3). Although not significantly different among treatments, small woody cover showed a consistent trend across soils, with an initial increase 2 years following treatment then a decline toward pretreatment levels for both brush removal treatments as the initial regrowth following treatment reached the medium height class (Fig.3).

There was a significant year*treatment interaction for both herbaceous cover and bare-ground (Table 2). Herbaceous cover was higher and percent bare ground lower in mechanically and chemically treated plots on all soil types in 2012 and 2013, after an initial drop in herbaceous vegetation and increase in bare-ground in the sampling period directly following treatment. This trend of high herbaceous cover and low percent bare ground was particularly pronounced on CKB soils. On CKB soils, multiple comparisons showed significant differences in 2012 and 2013 among treated plots and control plots in

Table 1. Differences in woody plant cover response among brush removal treatments on different soil types.

	Sum Sq	Mean Sq	NumDF	Den DF	F-value	p
Percent Woody Cover						
Treatment	3536.60	1768.30	2	15.68	22.24	<0.001
Soil	2065.40	1032.68	2	17.22	10.67	<0.001
Pasture	972.30	486.13	2	17.28	7.11	0.01
Year	3545.70	1772.87	2	49.19	21.91	<0.001
Treatment:Soil	826.00	206.50	4	17.38	2.55	0.08
Treatment:Year	3554.60	888.66	4	49.26	12.12	<0.001
Percent Woody Cover (<0.5m)						
Treatment	19.62	9.81	2	67.88	2.18	0.12
Soil	170.80	85.40	2	77.16	19.90	<0.001
Pasture	6.32	3.16	2	77.65	0.78	0.46
Year	363.89	181.95	2	68.34	45.21	<0.001
Treatment:Soil	14.04	3.51	4	76.70	0.87	0.48
Treatment:Year	252.02	63.00	4	56.45	18.47	<0.001
Percent Woody Cover (0.5-1.5m)						
Treatment	1878.14	939.07	2	15.65	23.47	<0.001
Soil	1310.02	655.01	2	17.41	16.45	<0.001
Pasture	376.04	188.02	2	17.48	6.01	0.01
Year	1283.00	641.50	2	47.32	15.79	<0.001
Treatment:Soil	875.81	218.95	4	17.51	5.39	0.01
Treatment:Year	1472.08	368.02	4	117.23	11.41	<0.001
Percent Woody Cover (>1.5m)						
Treatment	714.32	357.16	2	16.15	23.09	<0.001
Soil	95.37	47.68	2	17.43	2.04	0.16
Pasture	164.64	82.32	2	17.47	5.47	0.01
Year	578.79	289.40	2	49.61	18.43	<0.001
Treatment:Soil	64.18	16.04	4	17.64	1.02	0.42
Treatment:Year	756.49	189.12	4	30.01	12.23	<0.001

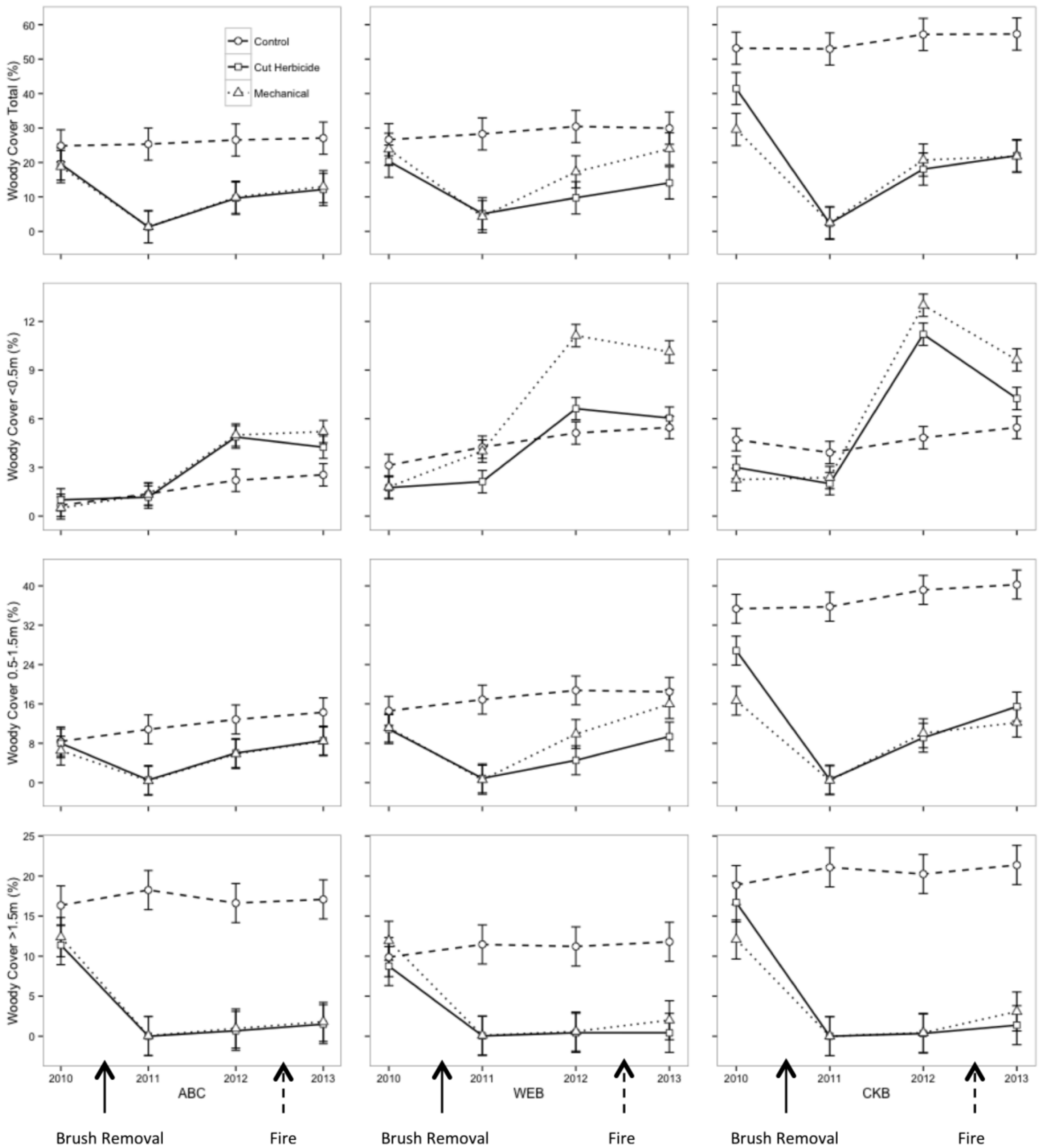


Fig 3. Changes in total woody brush cover and woody brush cover in each of three height classes: <0.5m, 0.5-1.5m, and >1.5m (mean ±SE) in response to different methods of brush removal (untreated control, cut stump, and roller chopping) on three different soil types: sandy(ABC), sandy loam (WEB), and clay loam (CKB). Year 2010 represents pretreatment conditions. Solid arrows indicate initiation of brush removal treatments and dashed arrows indicate initiation of prescribed fires.
herbaceous cover (Tukey's HSD: control v. cut herbicide, p=0.018; control v.

mechanical, p=0.017) and percent bare ground (Tukey's HSD: control v. cut herbicide, p=0.018; control v. mechanical, p=0.043) (Fig. 4). Multiple comparisons revealed no differences among treatments on other soil types, suggesting grass dominance had only been restored following brush reduction on CKB soils (Fig. 4).

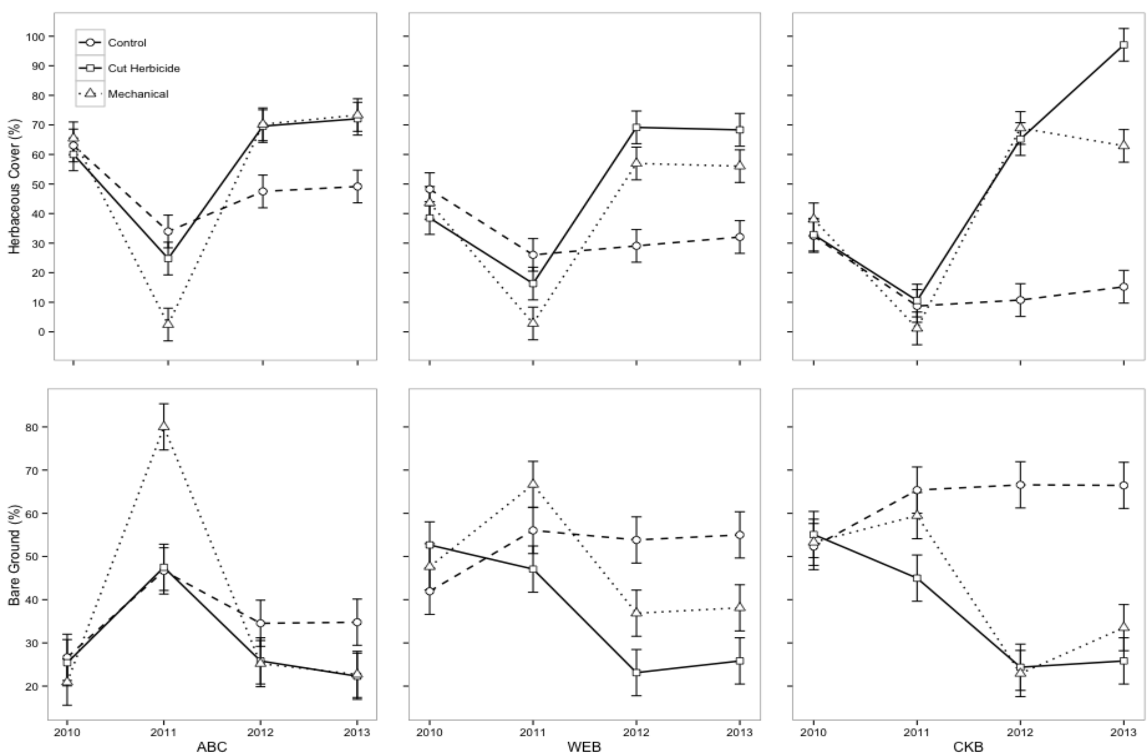


Fig. 4 Changes in herbaceous cover and bare ground (mean ±SE) in response to different methods of brush removal (untreated control, cut stump, and roller chopping) on three different soil types: sandy(ABC), sandy loam (WEB), and clay loam (CKB). Year 2010 represents pretreatment conditions.

Table 2. Differences in herbaceous plant cover and bare ground response among brush removal treatments on different soil types.

	Sum Sq	Mean Sq	NumDF	Den DF	F-value	p
Percent Bare Ground						
Mechanical	824.90	412.40	2	65.67	2.31	0.111
Soil	3681.90	1840.90	2	73.61	9.71	<0.001
Pasture	647.90	323.90	2	74.01	2.38	0.101
Year	14740.80	7370.40	2	65.97	41.25	<0.001
Treatment:Soil	952.50	238.10	2	73.56	1.33	0.273
Treatment:Year	14081.00	3520.30	4	33.02	20.99	<0.001
Percent Herbaceous Cover						
Mechanical	1074.50	537.24	2	15.04	3.38	0.061
Soil	1206.60	603.31	2	11.52	4.27	0.042
Pasture	1350.70	675.37	2	13.53	4.56	0.033
Year	5373.70	2686.87	2	50.01	16.73	<0.001
Treatment:Soil	1142.30	285.57	4	5.77	1.78	0.264
Treatment:Year	6198.10	1549.5	4	29.50	12.73	<0.001

Perturbation With Fire

Fire following mechanical or chemical treatments did not cause shrubland states to move into an alternative state across any soil type. Shrubland states recovered rapidly to pre-treatment structure and did not significantly differ in total percent woody cover, percent woody cover of any height class, herbaceous cover, or percent bare ground (Tables 3&4). Additionally, multiple comparisons revealed no differences between burned or unburned plots on any soil for any treatment other than CKB control plots, which had significantly higher woody cover in the burned plots initially (Fig. 5). However, while fuel loads differed among treatments ($F=3.25$, $p=0.049$), fuels were lower than the recommended 2240 to 3360 kg/ha needed for effective prescribed burning

Table 3. Differences in woody plant cover response to different brush removal treatments and prescribed fire on different soil types.

	Sum Sq	Mean Sq	NumDF	Den DF	F-value	p
Percent Woody Cover						
Trt	1095.2	547.6	2.0	15.9	20.5	<0.001
Fire	2.9	2.9	1.0	71.2	0.1	0.743
Soil	467.2	233.6	2.0	17.0	10.2	0.001
Year	158.3	158.3	1.0	71.2	5.9	0.018
Trt:Soil	315.5	78.9	4.0	17.2	2.9	0.051
Trt:Fire	70.5	35.2	2.0	71.2	1.3	0.277
Fire:Soil	140.6	70.3	2.0	71.2	2.6	0.080
Percent Woody Cover (<0.5m)						
Trt	106.1	53.0	2.0	17.3	11.8	<0.001
Soil	184.0	92.0	2.0	19.7	17.6	<0.001
Fire	0.0	0.0	1.0	72.3	0.0	0.972
Year	21.6	21.6	1.0	72.3	5.0	0.028
Trt:Soil	32.4	8.1	4.0	19.8	1.9	0.153
Trt:Fire	23.7	11.8	2.0	72.3	2.8	0.070
Soil:Fire	12.7	6.4	2.0	72.3	1.5	0.235
Percent Woody Cover (0.5-1.5m)						
Trt	594.7	297.4	2.0	15.8	18.0	<0.001
Soil	239.6	119.8	2.0	17.1	10.5	0.001
Fire	0.7	0.7	1.0	71.2	0.0	0.837
Year	239.3	239.3	1.0	71.2	14.4	<0.001
Trt:Soil	365.3	91.3	4.0	17.3	5.5	0.005
Trt:Fire	54.5	27.3	2.0	71.2	1.6	0.202
Soil:Fire	19.1	9.5	2.0	71.2	0.6	0.567
Percent Woody Cover (>1.5m)						
Trt	487.3	243.7	2.0	15.7	39.3	<0.001
Soil	16.3	8.1	2.0	17.0	1.4	0.270
Fire	5.9	5.9	1.0	71.1	1.0	0.332
Year	26.8	26.8	1.0	71.1	4.3	0.041
Trt:Soil	30.6	7.6	4.0	17.3	1.2	0.332
Trt:Fire	14.5	7.2	2.0	71.1	1.2	0.316
Soil:Fire	27.5	13.7	2.0	71.1	2.2	0.116

Table 4. Differences in herbaceous plant cover and bare ground responses to different brush removal treatments and prescribed fire on different soil types.

	Sum Sq	Mean Sq	NumDF	Den DF	F-value	p
Percent Bare Ground						
Trt	1538.7	769.3	2.0	15.8	17.5	<0.001
Soil	221.0	110.5	2.0	17.0	4.0	0.037
Fire	7.7	7.7	1.0	71.1	0.2	0.679
Year	42.5	42.5	1.0	71.1	1.0	0.331
Trt:Soil	518.5	129.6	4.0	17.3	2.9	0.052
Trt:Fire	424.1	212.0	2.0	71.1	4.8	0.011
Soil:Fire	210.9	105.5	2.0	71.1	2.4	0.100
Percent Herbaceous Cover						
Trt	11424.2	5712.1	2.0	15.0	19.8	<0.001
Soil	733.3	366.6	2.0	17.3	1.5	0.258
Fire	206.3	206.3	1.0	44.1	0.7	0.403
Year	440.9	440.9	1.0	22.0	1.5	0.230
Trt:Soil	2299.2	574.8	4.0	17.2	2.0	0.142
Trt:Fire	25.6	12.8	2.0	44.1	0.0	0.957
Soil:Fire	1260.1	630.1	2.0	44.1	2.2	0.126

in the study region (Lyons et al. 1998) for all treatments on all soil types (Table 5). Therefore, the fire temperatures were low on average for all treatments and the resulting low intensity burns did not carry across the patchily distributed fuels. As a result, percent of the plot scorched was low and extremely variable for all treatments on all soil types (Table 5).

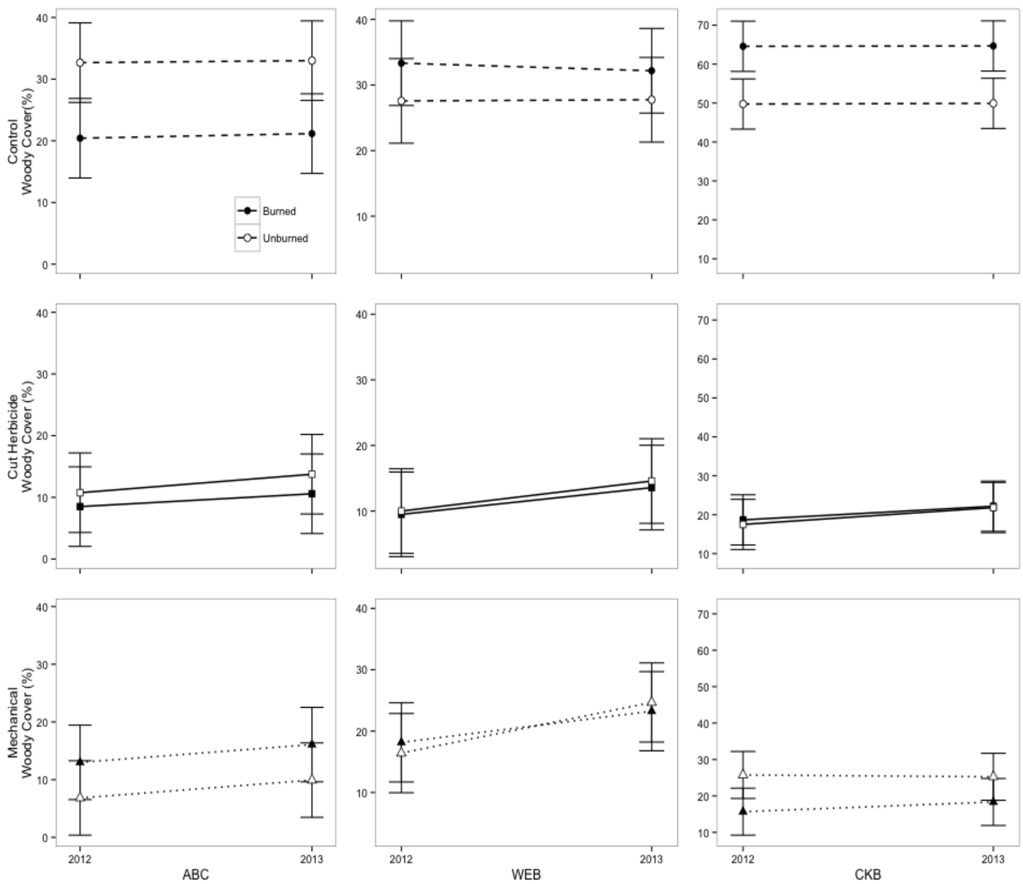


Fig. 5 Changes in total woody brush cover (mean ±SE) in response to prescribed burning in plots previously exposed to different methods of brush removal (untreated control, cut stump, and roller chopping) on three different soil types: sandy(ABC), sandy loam (WEB), and clay loam (CKB). Year 2012 represents pre-burn conditions.

Table 5: Mean and standard error for fuel load, scorch, and temperature among brush removal treatments (untreated control, hand-cutting followed by herbicide, and roller-chopping) on different soil types (ABC is sandy, WEB is sandy loam, CKB is clay). Fuel was measured immediately prior to conducting prescribed burns, scorch immediately following burns, and temperatures were recorded during the fire.

	Fuel Load (kg/ha)		**Scorch (%)**		**Temp (ºC)**	
	Mean	**SE**	**Mean**	**SE**	**Mean**	**SE**
ABC						
Control	1356.57	278.57	<1	0.33	13	7.09
Cut Herbicide	1189.52	197.96	15	12.58	114	64.66
Mechanical	785.51	75.62	9	4.70	69	34.96
WEB						
Control	1688.23	402.96	5	2.33	89	41.52
Cut Herbicide	1438.28	267.14	47	12.03	210	90.13
Mechanical	1154.65	176.72	15	7.67	93	22.86
CKB						
Control	878.44	148.34	1	1.33	65	7.47
Cut Herbicide	1940.30	719.05	23	20.55	80	55.31
Mechanical	676.50	58.99	7	2.03	60	9.99

Compositional Contributions to Resilience

NMDS shows composition shifts do not necessarily correspond with the ability to overcome resilience of the shrubland state and move into a new, grassland-dominated basin of attraction. Composition did not differ among soils prior to treatment (standard errors on two NMDS axes overlap for all soil types in 2010). The CKB soils, which experienced large changes in woody and herbaceous cover in response to both brush removal treatments maintained similar composition on control plots and treated plots for the duration of the study (Fig. 6). Standard errors on two NMDS axes did not overlap in

2010 and 2013 for cut herbicide plots on ABC soils and for cut herbicide and mechanically treated plots on WEB soils, suggesting a trend toward compositional shifts following brush removal treatment on these two soils (Fig. 6).

Change in relative abundance of species varied among treatments and soils as well (Table 7). The most dominant species, honey mesquite (*Prosopis glandulosa*) decreased significantly more in cut herbicide plots than in mechanically treated or control plots overall ($F=5.46$, $p=0.008$). However, change in relative abundance of mesquite was not different among treatments on WEB soils ($F=0.6$, $p=0.567$). The only other species exhibiting significant overall treatment differences was prickly pear (*Opuntia englemannii*), which increased significantly more in relative abundance in cut herbicide plots over control and mechanically treated plots ($F=8.527$, $p=0.0008$). Whitebrush (*Aloysia gratissima*) relative abundance increased more in cut herbicide plots on ABC soils than control or mechanically treated plots ($F=5.39$, $p=0.012$). Similarly, spiny hackberry (*Celtis ehrenbergiana*) increased in relative abundance more on mechanical plots in WEB soils than on control or cut herbicide plots ($F=5.47$, $p=0.008$).

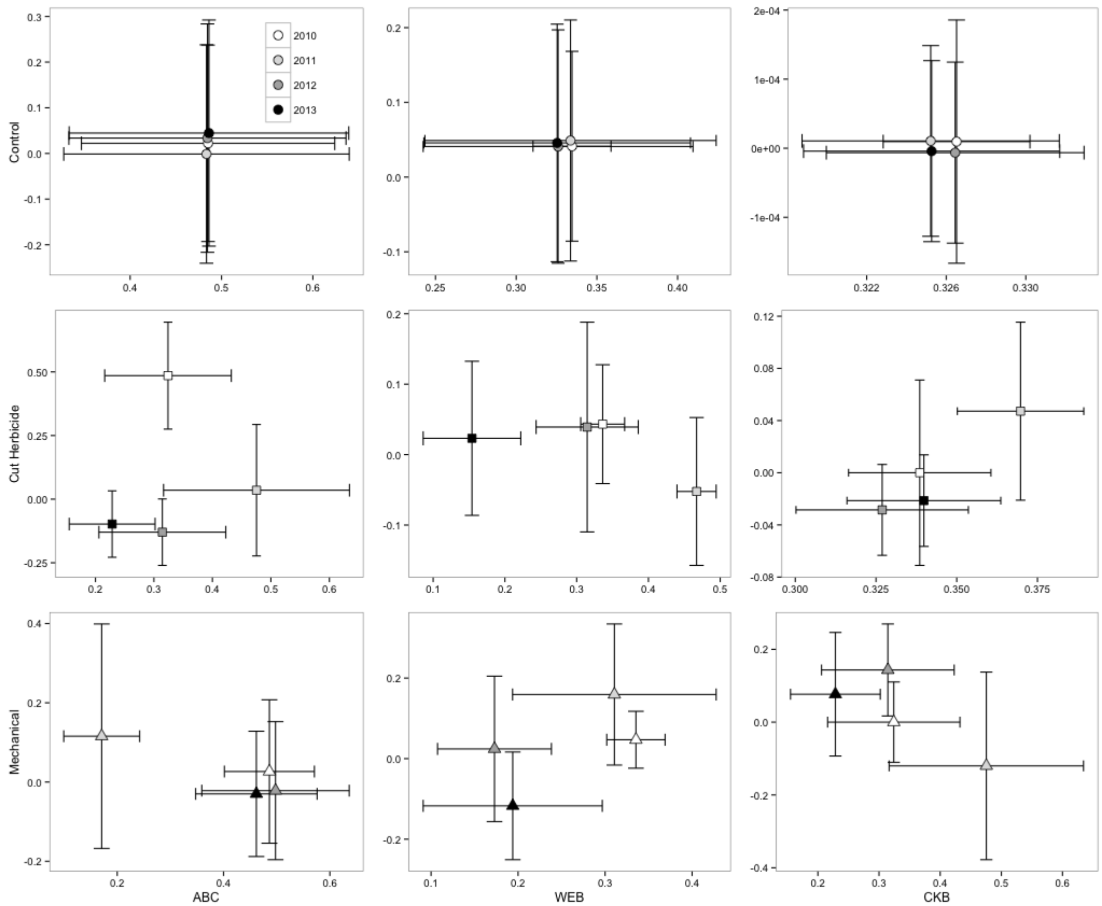


Fig. 6 Nonmetric multidimensional scaling of woody brush communities for years 2010 (pretreatement) through 2013 for each soil*brush removal treatment combination. The mean and standard error on NMDS axis 1 and NMDS axis 2 for each year of the study are displayed in order to assess divergence in community compositional trajectories over time following treatment. Treatments and years are coded as follows: Shapes represent brush removal treatments of control (circles), cut stump (squares), and roller chop (triangles); Colors represent years: 2010(white), 2011 (light gray), 2012 (dark gray), and 2013(black).

Table 6: Mean change and standard error in relative abundance between 2010 and 2013 of the 8 most dominant species for each treatment (untreated control, hand-cutting followed by herbicide, and roller-chopping) on different soil types (ABC is sandy, WEB is sandy loam, CKB is clay). Bold species differed significantly in relative abundance among treatments.

Species	**Scientific name**	**Control**		**Cut herbicide**		**Mechanical**	
				ABC			
Mesquite	***Prosopis glandulosa***	**-0.014**	**0.014**	**-0.095**	**0.054**	**-0.031**	**0.024**
Blackbrush	*Acacia rigidula*	-0.005	0.010	0.010	0.014	0.004	0.018
Whitebrush	***Aloysia gratissima***	**0.022**	**0.011**	**0.067**	**0.037**	**0.003**	**0.024**
Twisted Acacia	*Acacia schaffneri*	0.002	0.009	0.015	0.008	-0.002	0.023
Hackberry	*Celtis ehrenbergiana*	0.017	0.023	0.028	0.027	0.044	0.019
Guyacon	*Guaiacum angustifolium*	0.010	0.011	-0.013	0.023	0.017	0.009
Persimmon	*Diospyros texana*	-0.002	0.018	-0.067	0.036	-0.031	0.022
Brasil	*Condalia hookeri*	-0.062	0.022	-0.073	0.020	-0.034	0.020
				WEB			
Mesquite	*Prosopis glandulosa*	-0.015	0.013	-0.022	0.016	-0.019	0.038
Blackbrush	*Acacia rigidula*	0.001	0.008	0.035	0.038	-0.014	0.014
Whitebrush	*Aloysia gratissima*	0.018	0.004	0.041	0.012	0.024	0.019
Twisted Acacia	*Acacia schaffneri*	0.004	0.008	-0.016	0.030	0.012	0.019
Hackberry	***Celtis ehrenbergiana***	**0.006**	**0.008**	**0.004**	**0.014**	**0.047**	**0.031**
Guyacon	*Guaiacum angustifolium*	0.018	0.012	-0.027	0.012	-0.017	0.006
Persimmon	*Diospyros texana*	0.008	0.005	-0.023	0.012	-0.015	0.009
Brasil	*Condalia hookeri*	-0.016	0.009	-0.015	0.018	-0.021	0.003
				CKB			
Mesquite	***Prosopis glandulosa***	**-0.028**	**0.021**	**-0.162**	**0.055**	**0.009**	**0.021**
Blackbrush	*Acacia rigidula*	0.013	0.006	-0.032	0.019	-0.033	0.031
Whitebrush	*Aloysia gratissima*	0.032	0.013	-0.014	0.060	0.030	0.039
Twisted Acacia	*Acacia schaffneri*	0.021	0.017	0.042	0.026	0.000	0.013
Hackberry	*Celtis ehrenbergiana*	-0.003	0.004	0.005	0.013	0.015	0.010
Guyacon	*Guaiacum angustifolium*	0.000	0.020	-0.016	0.023	-0.064	0.019
Persimmon	*Diospyros texana*	-0.008	0.020	-0.005	0.005	-0.003	0.009
Brasil	*Condalia hookeri*	-0.008	0.002	-0.004	0.012	-0.021	0.009

Discussion

The results of this study demonstrate that the resilience of *Prosopis-Acacia* dominated rangelands to brush removal varies across a range of soil types. Shrubland community composition did not differ among soil types prior to mechanical or chemical treatment, yet showed contrasting abilities to recover, suggesting that soil texture mediates shrubland resilience to perturbation. Shrublands on fine-textured clay soils were the least resilient of the shrubland-soil type associations studied here. On both coarser sandy and sandy-loam soils, woody plants quickly regained pretreatment levels of dominance and the reestablishment of grassland dominance was short-lived. This rapid return to a pre-disturbance configuration without an intervening perturbation is evidence that the resilience of the pre-disturbance state has not been overcome and the self-perpetuating processes and structures of the desired grass-dominated state have not been reinstated on these soils (Allen et al. 2005). However, grass-dominance persisted and woody plant abundance remained low on clay soils for the duration of the study. Given the lower resilience of shrublands on clay soils, restoration practitioners are more likely to achieve long-term success in these areas and should strategically implement interventions accordingly.

Interactions between herbaceous and woody plants has been found to vary as a function of soil texture in studies undertaken in other rangeland systems, suggesting that our resilience-based approach is likely to be transferrable to a broad range of terrestrial systems and can help prioritize restoration actions with alternative shrubland and grassland states given a sufficiently broad range of soil conditions. In South Africa, acacia growth increased where grass was removed on fine-textured soils, but similar grass removal had no effect on acacia growth on coarser textured soils, suggesting that grasses limited the recharge of deeper soils on fine-textured soils, but not on coarse-textured soils (Knoop and Walker 1985). Tree height and canopy cover were found to decrease with increasing clay content in tropical savannas in Australia (Fensham and Kirkpatrick 1992, Williams et al. 1996). In California oak savanna, grasses were found more often on soils with higher clay content, while trees occurred in areas with lower clay content, suggesting that resource heterogeneity related to soil texture drives the distribution of grasses and woody plants in these systems (Robinson et al. 2010). Similarly, a study conducted across the central grassland region of the United States found that soil texture affected the relative abundance of plant functional groups, with higher woody plant abundance on coarse-textured soils and a greater proportion of grasses on fine-textured soils in semi-arid areas (Lane et al. 1998).

While nuanced theories of grass-tree co-dominance in arid and semi-arid ecosystems stress the importance of complex interactions among temporal and spatial dynamics in resource availability and variability and the importance of disturbance regimes, especially fire and grazing for determining the relative abundance of woody and herbaceous vegetation (Scholes and Archer 1997, Rebertus and Burns 1997, Jeltsch et al. 2000), spatial partitioning of water and soil nutrients often plays an important role semi-arid savanna dynamics (Sankaran et al. 2004, Ward et al. 2013). According to Walter's two layer hypothesis, which has been shown to hold for systems similar to the focal area of this study (Ansley et al. 2007), spatial niche separation often drives grass-tree co-dominance in savannas (Walter and Mueller-DomBois 1971, Scholes and Archer 1997). Grasses can outcompete trees for surface soil water, but trees have access to deeper soil water unavailable to grasses because it is below their rooting zone. Thus, as long as there is adequate partitioning of rooting depth relative to soil water distribution, grasses and trees will coexists. Given these dynamics, soil water-holding capacity and infiltration patterns which are largely related to soil texture (Larcher 2003) will be important determinants of post-treatment grass-tree dynamics. The rapid return of woody plant dominance on sandy and loam soils in this study highlights the relationship of soil water dynamics and woody plant resilience to brush thinning treatments. While brush removal overcame the resilience of the woody-dominated state on finer textured clay soils with greater water-holding capacity, they were ineffective at overcoming the resilience of the woody-dominated state on coarser texture soils with lower water storage capacity and higher infiltration rates which tip the competitive balance towards woody plants, increasing their resilience on these soils. In addition to large water-holding capacity that may favor grass dominance, clay soils in the study area had higher pre-treatment densities of mature woody plants. Studies have found that savanna trees enhance physiochemical soil properties (Tiedemann and Klemmedson 1973). Several studies found higher pools of nitrogen available for mineralization, increased soil organic carbon, and increased abundance of microbes under trees than in open areas in savanna systems (Belsky et al. 1989, Weltzin and Coughenour 1990, Scholes and Archer 1997). This enhancement can lead to greater grass response to brush removal in areas with previous high densities of woody plants relative to areas with lower initial densities (Scholes and Archer 1997).

Grass-tree co-dominance, however, is not constant within savanna ecosystems (Sankaran et al. 2004, Wiegand et al. 2006). Rather, the area occupied by grassland states is decreasing and shrubland states increasing in many semi-arid rangelands worldwide (Wiegand et al. 2005, VanAuken 2009). This is often attributed to a combination of altered disturbance regimes and grazing management practices, and global climate change including altered patterns of precipitation that provide a competitive advantage to tress over grasses (Scholes and Archer 1997). Therefore maintenance of grass dominance in rangeland systems with alternate stable configurations is dependent on external drivers, especially fire and variation in rainfall (Higgins et al. 2000). Fire has been shown to be a crucial feedback for maintaining a grass-dominated state in rangeland systems (Skarpe 1992, Higgins et al. 2000, Van Langevelde et al. 2003, Moreno et al. 2014). Fluctuations in precipitation are not sufficient for maintaining grass-tree co-dominance in the absence of disturbances such as fire and grazing in savanna systems (Jeltsch et al. 1996, Anderies et al. 2002). Especially in systems like the focal system of this study in which the shrub component is dominated by resprouting species (Wright and Clarke 2007). Fire can reduce recruitment and development of seedlings by killing the immature woody plants while leaving the root crown of grasses intact (Higgins et al. 2000). Frequently, large recruitment events for woody plants correspond to periods of high moisture availability and high grass-fuel accumulation as well. This creates a self-reinforcing feedback through which the otherwise inferior competitor (grass) indirectly affects the dominant competitor (woody plants) in grass-dominated rangelands. This feedback loop is moderated by variability in rainfall in semi-arid systems which limits recruitment events and creates great potential for dry periods with higher likelihood of fires to immediately follow wet periods of woody seedling recruitment and herbaceous fuel accumulation (Bond 2008, Prior et al. 2010). Additionally, fire intensity increases with accumulation of fine fuels, increasing the probability that a new woody recruit will not escape the "fire trap" (Hoffmann et al. 2009, Grady and Hoffmann 2012). However, reestablishment of the fire feedback in grass dominated systems is sensitive to climatic conditions following woody plant reduction. In our study, years of below average rainfall following brush removal led to low fuel accumulations across treatments (Long et al. 2013, Grigg 2014). Low fuel loads resulted in burns with insufficient intensity to reduce woody cover, which on many plots already had high recovery of the medium woody plant height class. Many studies show that mechanical restoration of grass-dominated states is insufficient to prevent regrowth of woody species, but periodic burns following restoration maintain the system in the grass dominated configuration (Brockway et al. 2002, Smit 2004, Briggs et al. 2005, Watts et al. 2006, Ansley et al. 2006, Archer 2010). Therefore, the additional increase in herbaceous cover in cut herbicide plots relative to mechanically treated plots on CKB soils in 2013, which experienced far greater growing-season precipitation suggests higher likelihood of maintaining grass dominance in cut-herbicide plots, as the fire feedback will be more easily restored there because of higher fuel accumulation.

Resilience of a state is often tied to the specific traits of the species comprising it. Thus, shifts to alternate basins of attraction can lead to divergent species assemblages (Peterson et al. 1998, Folke et al. 2004). Trajectories of plant community composition through time can be indicated by looking at the movement of the community through ordination space over time (Hobbs and Gimingham 1984, Malanson and Trabaud 1987). Despite similar pre-treatment compositions, our ordinations of woody plant communities over time revealed the trajectories to be divergent on sandy and loam soils for mechanical treatments, but to remain within the range of variability of pretreatment compositions for both brush removal treatments on clay soils. This is counterintuitive given that clay soils showed the greatest magnitude of response to brush clearing and were the only soils where the grass dominated state was restored. However, it is possible to overcome the resilience of a particular configuration of the system without causing compositional shifts (Lavorel and Garnier 2002). The divergence in the relative abundance of grasses to woody plants results in a shift from the self-perpetuating dynamics maintaining the shrubland system, including inhibition of fire (Anderies et al. 2002, Hirota et al. 2011) and redistribution of water and nutrients to deeper soils (Vetaas 1992, Scholz et al. 2002), to processes perpetuating grass dominances, such as the feedback between increased grass cover and increased infiltration of precipitation into shallow soil layers (Walker et al. 1981). This shift occurs despite the presence of all the species, albeit in lower abundance, that comprise the woody-dominated configuration (Walker et al. 1997).

Similarly, because of functional redundancy among species in a community, it is possible to have species community changes with no attendant shift in system functioning or processes, (Walker et al. 1999, Lavorel and Garnier 2002). For instance, the woody plant composition can shift, but the positive feedback by which shading from trees suppresses grass growth, lowering the likelihood that fire will be able to reduce woody plant cover, is maintained regardless of the specific woody plants present. Therefore, the divergence of composition in sand and loam soils on the cut herbicide plots is not necessarily indicative of a state shift, but rather could be the result of differences among species with regards to vegetative regeneration. Mechanical brush removal that destroyed root crowns increased relative abundance of mesquite and acacia in one study in the region of our study site by 70% (Fulbright 1996). These species were the only ones in this system with the capacity to regenerate from lateral roots, and thus they experienced a competitive advantage following the destruction of the meristematic tissues relative to species that could not resprout from roots. However, surface removal of woody brush which did not injure root crowns resulted in similar community composition before and following treatment because interspecific patterns of growth were not altered through the disturbance (Fulbright and Beasom 1987). In our study, spiny hackberry and whitebrush increased in relative abundance in mechanically treated plots. These species both have large root crowns that increase the amount of meristematic tissue and belowground carbon storage available for resprouting (Flinn et al. 1992). Other studies have observed long term increases in relative cover for these species following mechanical brush removal (Scifres et al. 1977, Koerth et al. 1989).

Regardless of mechanism for divergence, the shift in composition is not sufficient to push the system out of the woody-dominated basin of attraction, as hackberry and whitebrush are functionally similar to the previously dominant honey mesquite and acacia species especially regarding system processes such as maintaining low levels of herbaceous biomass and reducing the potential for fire-feedbacks to be reinstated in the system.

Conclusion

Operationalizing resilience in restoration interventions has proven extraordinarily difficult since the concept was introduced four decades ago (see Holling 1973). Difficulties in applying the concept are often a function of the complexities of interactions in ecological systems and their influence on underlying mechanisms driving transitions among alternative states. Complex systems often exhibit non-linear dynamics (Peters et al. 2004) and the resilience of alternate configurations depends on the interactions among the species present in the system and their abiotic environment, increasing the complexity of system responses to disturbance (Carpenter et al. 2001, Anderies et al. 2002, Allen et al. 2005). In this study, I show a simple approach that can provide a basis for prioritizing restoration actions by identifying differences in the relative resilience of a community across an environmental gradient. In our study region, our results show mechanical and chemical brush controls are most likely to meet long-term restoration goals on fine-textured clay soils, where shrubland resilience is lowest. Similar experimental approaches can provide a foundation for operationalizing resilience in restoration and prioritizing management actions, which is critical given the limitations associated with broad-scale application of mechanical and chemical brush control (Twidwell et al. 2013).

CHAPTER III

GRASS MORTALITY AND TURNOVER FOLLOWING CORE RANGELAND RESTORATION PRACTICES

Introduction

Survival, life expectancy, and life span are key demographic parameters that determine individual plant responses to disturbance and management in rangelands (Lauenroth and Adler 2008). Species that are dominant in many ecosystems are more likely to have longer life spans and higher life expectancies (Lorimer et al. 2001, Grime 2007b, Lauenroth and Adler 2008). This relationship between longevity and dominance is hypothesized to occur because species with longer life spans exhibit fewer fluctuations in population growth rates, which results in those populations maintaining persistence in systems while other species fluctuate in relative abundance and experience local extinctions (Schoener 1983, Ehrlén and Lehtilä 2002). Persistence allows for exploitation of optimal conditions for population growth by maintaining a constant presence in the system and it provides a mechanism for continued growth under conditions of environmental stress and disturbance (Ozinga et al. 2007, Lauenroth and Adler 2008). Therefore, differential longevity among species has a potentially long-term influence on plant community composition and structure (Harcombe 1987, Pacala et al. 1996). However, the relationship between longevity and dominance is mediated by disturbance. There is often a trade-off among life history strategies with long-lived plant species lacking traits necessary for rapid recolonization following disturbance (Crawley and Ross 1990, Grime 2007a). Therefore, low survival of long-lived species in response to intense disturbance can lead to community compositional shifts as long-lived, slow-growing species are replaced by rapidly colonizing, fast-growing species (Tilman 1990, Louault et al. 2005). Interactions between species are mediated by species-specific interactions with the abiotic environment because of differential resource availability associated with differing abiotic conditions (Grime 1977, Fynn et al. 2005). This leads to additional variability in species compositional response to disturbance beyond that resulting from the interactions among the biotic components of the ecosystem.

In rangelands, management interventions have often centered on a solitary utilitarian objective and, as a result, have sought to minimize disturbances that decrease survival of long-lived perennial grasses and provide opportunities for shorter-lived species that respond to increases in bare ground (Herrick et al. 2006, Fuhlendorf et al. 2012, Twidwell et al. 2013a). Such turnover is presumed to set the stage for undesirable regime shifts in grassland community composition, increasing the dominance of less desirable annuals and weedy perennial species. The potential for rangeland management actions to cause grass mortality has therefore been at the forefront of rangeland decision-making historically (Wright and Klemmedson 1965, Wright 1970, 1982, Johnson and Strang 1983) and is one of the leading reasons why some managers avoid using fire and instead prefer mechanical or chemical options for controlling woody invaders (Taylor et al. 2011). However, the relative contributions of different rangeland management techniques to grass survival and turnover are not known for individual species because the discipline has largely focused on structural metrics of cover or biomass instead of tracking the mortality of individual plant tufts (Lauenroth and Adler 2008, Zimmermann et al. 2010). This important knowledge gap needs to be addressed to inform rangeland managers of the relative effect of various interventions on grass mortality and turnover and ensure that decisions are not based on personal perception or observer bias.

In this study, I quantified survival and recruitment of perennial grass species in response to core rangeland restoration practices replicated across multiple soil types in order to determine the potential for different interventions to cause shifts to undesirable grass community assemblages. To do this, I mapped individual perennial grass tufts and recorded basal area in permanently established plots on an annual basis (Weaver and Clements 1938, Lauenroth and Adler 2008, Silvertown and Charlesworth 2009). I used these maps to track survival, recruitment, and relative turnover of grass species in response to mechanical brush removal, chemical control of woody species, low intensity prescribed burning, and untreated controls. Additionally I performed ordinations of the entire grass community, both annuals and perennials, to explore species compositional shifts resulting from management interventions across a soil gradient ranging from fine clays to coarse sands. The results of this study provide information necessary for managers to choose interventions based on the ecological outcomes of land management actions and move away from current decision-making based on social conventions.

Methods

Study Site

This research was conducted at the Chaparossa Ranch in Zavala county in southwest Texas (29 lat, -100 long). The Chaparossa Ranch is a privately owned hunting and cattle enterprise spanning 36,360 ha. Individual pastoral units vary in size, ranging from 500-5,000 ha. Soils vary within the larger pastures, ranging from fine clays to sandy loams. Vegetation community composition is consistent across the site. Many of the sites are best characterized as a heavily encroached *Prosopis-Acacia* shrubland dominated by *Acacia rigidula* Benth, *Prosopis glandulosa* Torr, and *Acacia schaffneri* (S. Watson) F.J. Herm. The dominant grasses at the study site include *Hilaria belangeri* (Steud.) Nash, *Bouteloua hirsuta* Lag., *Aristida purpurea* Nutt., and *Heteropogon contortus* (L.) P. Beauv. ex Roem. & Schult. The site is subtropical with average annual rainfall of 560mm, bimodally distributed with a majority of precipitation events occurring in spring and fall (Jacoby and Meadors 1983).

Experimental Design

Three pastures were selected to implement shrubland restoration treatments within a randomized complete block design (Fig. 7). The three pastures (VAT, NBW, and MAT) were treated as blocks because they differed in land-use history. The VAT pasture was grazed with high intensity, low duration grazing during the recent past. The NBW pasture underwent extensive herbicide testing in the 1960s across much of the pasture and was periodically grazed since. No shrub management was reported for the MAT pasture and it also was periodically grazed. All pastures are currently moderately grazed with rotation into pastures depending on site conditions. Within each pastoral block, three soil types were selected to represent the range of textures observed on the study site. Soil types were Antosa-Bobillo sand association (ABC), Webb fine sandy loam soils (WEB), and Chacon clay loam soils (CKB) (Soil Survey Staff 2013). Three core rangeland restoration treatments, chemical (cut-herbicide), mechanical (roller-chop), and fire, were compared to an untreated control. At the beginning of the study a total 54 plots, each approximately 40m x 25m, were established across the study area to implement mechanical and chemical treatments in year 1 with two repetitions in each soil-site block for a total of 18 plots treated with roller-chopping, 18 plots treated with cut-herbicide, and 18 untreated controls. Roller-chopping uses a cylindrical drum (Pasture Aerator, Lawson Mfg. Inc., now RanchWorx, Palm Harbor, FL) equipped with blades towed behind a tractor to cut and crush woody vegetation at the soil surface (Fulbright et al. 1991, Blanco et al. 2005). I used roller-chopping as a mechanical method for this study because it provided a gradient of soil disturbance with high disturbance from roller-chopping, low soil disturbance from cut- herbicide treatment, and no soil disturbance in control plots. In cut herbicide treatments, I cut all shrubs and trees in the plot at the base and sprayed a 15% Remedy herbicide (Dow AgroSciences LLC, Indianapolis, IN) and diesel mixture on the cut stumps. Within each soil-site block two years following mechanical and chemical treatments, a prescribed fire treatment was randomly assigned to one of each replicates for a total of 27 burned plots (9 roller-chopped, 9 cut-herbicide, and 9 untreated controls were burned). Burned plots were separated by disked fire-breaks and ignited separately with ring fires. Mechanical and chemical treatments had no effect on grass mortality in burned plots ($F=1.15$, $p=0.32$), so I were able to directly compare burn treatments to chemical, mechanical, and untreated control plots.

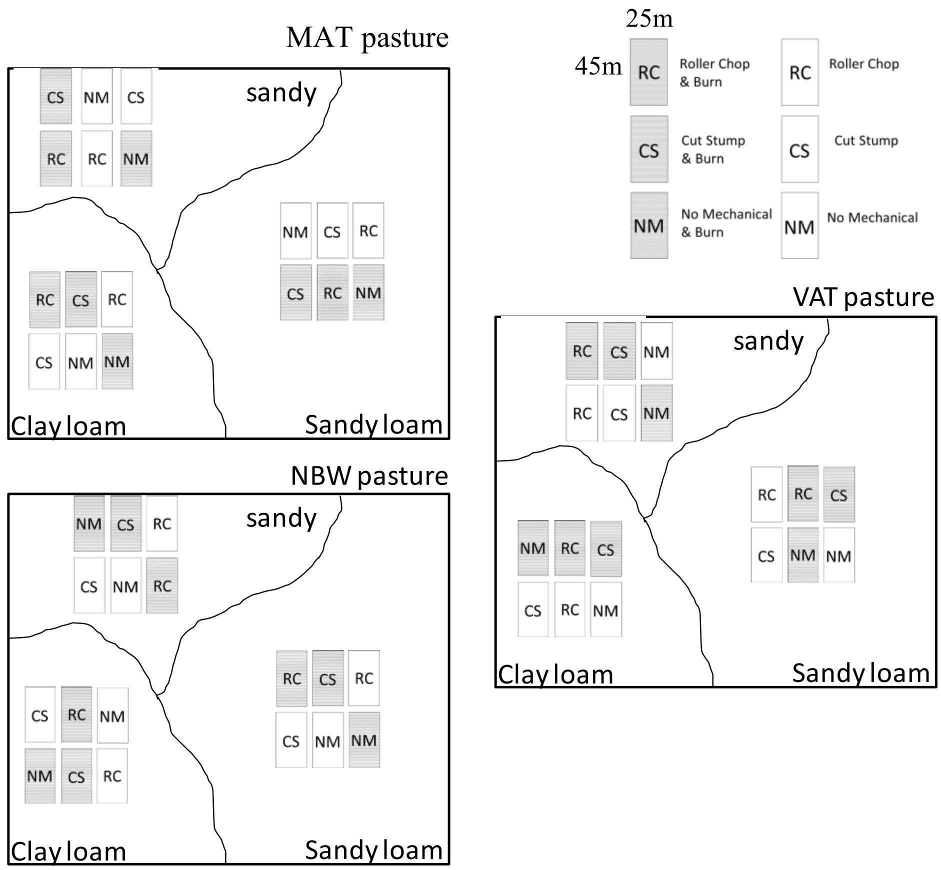


Fig. 7 Conceptual diagram of experimental design.

I established 5 (1m²) subplots for sampling in random locations within each plot, marking the corners with rebar for relocation. Each year during peak perennial grass production for the duration of the study, starting prior to the implementation of brush removal treatments in late summer/early fall 2010 and ending in 2013, I mapped the basal area of each perennial grass tuft in the subplot by drawing it to the nearest cm² on gridded paper with one square representing 1cm². I measured the distance from the subplot corner posts to the center of the grass tuft to map its location within the subplot and identified each tuft to species. Fire temperatures at each subplot were recorded during the prescribed burn using ceramic tile pyrometers painted with 10 temperature-indicating lacquers (OMEGALAQ® Liquid Temperature Lacquers; Omega, Inc.) that melted from 79˚C to 640˚C. Percent scorch was visually estimated immediately following the burns for each subplot.

Data Analysis

Using the mapped basal areas, I were able to determine survival, recruitment, and relative species turnover of grasses within subplots (Fig. 8). I defined individuals as individual perennial grass tufts for purposes of quantifying survival and recruitment. This assumption is reasonable given that the majority of the grass species at the study site are perennial warm season tufted bunch grasses (Appendix A). I did not include annual species in the analysis because by definition they undergo 100% mortality at the end of each growing season. However, they accounted for less than 10% of the annual average herbaceous cover on all soil types. Additionally, survival for burned plots was calculated based on a subset of the subplots that actually burned during the prescribed fire because the plot level prescribed burns were patchy given below optimal plot level fuel loads. Mortality, the inverse of survival, was considered to occur when an individual was not present in the location it was mapped in the previous year, because it was now bare ground or occupied by an individual of a different species. Recruitment was considered to occur when an individual was present in a location it was not previously mapped in because that spot was bare ground or previously occupied by another species. Relative species turnover quantifies the proportion of the total species

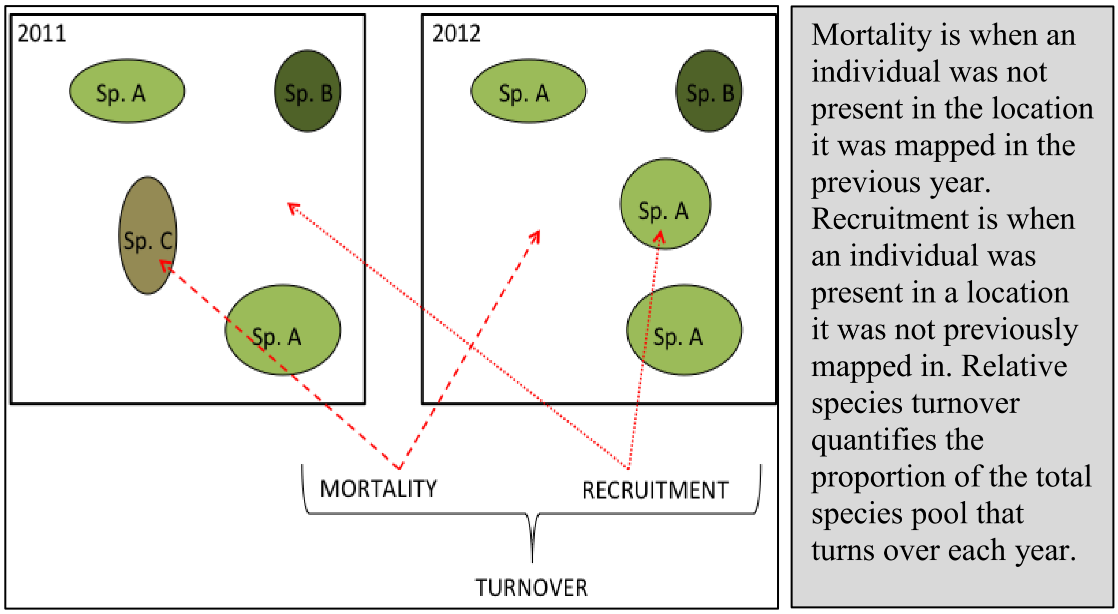


Fig. 8 Conceptual diagram of mortality, recruitment, and species turnover.

pool that turns over each year. I calculated relative species turnover for each subplot for each year using the following equation (Diamond 1969).

$$Turnover@\ time\ t + 1 = (U_t + U_{t+1})/(S_t + S_{t+1})$$

Where U_t is the number of species unique to the subplot during a given year

U_{t+1} is the number of species unique to the subplot in the subsequent year

S_t is the total number of species in the subplot during a given year

S_{t+1} is the total number of species in the subplot in the subsequent year

I tested for differences in mortality, recruitment, relative species turnover, and percent cover of bare-ground among brush removal treatments and soil types using

ANOVA for the randomized complete block design or repeated measures ANOVA for the randomized complete block design where appropriate, with pasture modeled as a random effect because the pastures represent a random sample of all potential land-use histories. I tested for differences in mortality among treatments in the first year following the treatment. I only explored the first year following treatment for mortality because fires were set one year prior to the end of the study due to inadequate fuel loading prior to that. Mortality resulting from fire was determined based on mortality in a subset of subplots that had actually burned averaged over the entire plot. I included burned plots that had previously been mechanically treated, chemically treated, and those that had no prior treatment. Additionally, I compared mortality in control plots between 2011 and 2013 to test that it would be appropriate to compare mortality in the sampling period following cut-herbicide and roller-chopping (2011) to mortality in the sampling period following prescribed burning (2013). I also tested for differences in recruitment and relative species turnover among control, chemical, and mechanical treatments during all years of the study. Additionally, I tested for differences in bare-ground among treatments on different soils in the sampling period following treatment to determine relative levels of soil disturbance caused by the treatments. I performed multiple comparisons to compare the mean response of each soil type and treatment at each year using Tukey's HSD. I set $\alpha=0.05$ to determine significance in all analyses.

I used multivariate analyses to test for differences in the trajectory of grass community composition following treatments. Permutational multivariate analysis of variance (PERMANOVA) was used to test for differences in community composition resulting from different brush removal treatments on different soil types. PERMANOVA is commonly used in ecological community analyses where data often do not conform to the assumptions of MANOVA (Anderson et al. 2005). Nonmetric multidimensional scaling (NMDS) with Bray-Curtis distances (Beals 1984) was used to visualize differences in the grass community composition among soils and brush removal treatments. All analyses were performed using the R statistical computing package (R Development Core Team 2010).

Results

Mortality and Recruitment

Perennial grass mortality differed among core rangeland restoration treatments in the year following treatment initiation (Table 8). Since neither brush removal treatment had an effect on mortality of burned versus unburned plots on any soils (F=1.15, p=0.32) and mortality in control plots did not differ in the sampling period following cut-herbicide and roller-chopping treatments and the sampling period following prescribed burns (F=1.72, p=0.19), it was possible to directly compare the amount of

Table 7. Differences in mortality among treatments one year following treatment on different soil types.

	Sum sq	**Mean sq**	**NumDF**	**DenDF**	**F-value**	**p**
Treatment	232.86	77.62	3.00	66.00	33.74	<0.001
Soil	7.15	3.57	2.00	66.00	1.72	0.187
Pasture	1.23	0.62	2.00	66.00	0.26	0.770
Treatment:Soil	18.93	3.16	6.00	66.00	1.38	0.237

perennial grass mortality resulting from prescribed fire with the amount of perennial grass mortality resulting from the brush removal treatments and the untreated controls. Mean mortality of perennial grasses was higher in mechanically treated plots than burned plots, cut herbicide treated plots and controls on all soils ($F=6.58$; $p=0.01$). There were similar amounts of mortality in burned plots, cut herbicide plots, and untreated controls on all soil types (Table 8, Fig. 9). Mortality of perennial grasses as a result of mechanical treatment was higher on sandy loam WEB soils than on sandy ABC or clay CKB soils. Recruitment did not differ significantly among brush removal treatments overall ($F=1.80$; $p=0.20$).

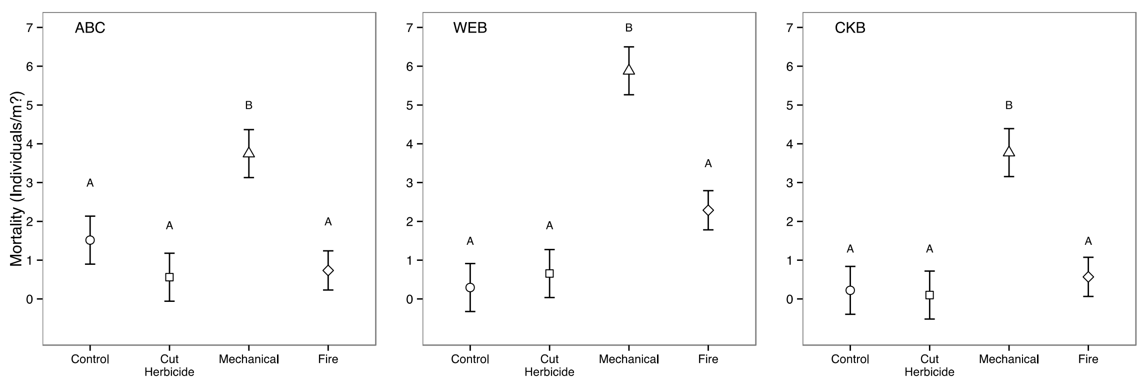


Fig. 9 Perennial grass mortality (mean ±SE) measured one year following treatment for brush removal treatments (control, cut herbicide, and mechanical) and fire on three different soil types: sandy(ABC), sandy loam (WEB), and clay loam (CKB).

Relative Species Turnover

I found a significant treatment-by-year interaction in relative species turnover (Table 9, fig. 10). On sandy ABC soils and clay CKB soils, turnover was higher for mechanically treated plots in the first and second year following treatment (Tukey's HSD: ABC, control v. mechanical, p=<0.001; mechanical v. cut herbicide, p=0.002; CKB, control v. mechanical, p=<0.001; mechanical v. cut herbicide, p=<0.001), but was not different from control or cut herbicide plots three years following treatment (Tukey's HSD: ABC, control v. mechanical, p=0.65; mechanical v. cut herbicide, p=0.26; CKB, control v. mechanical, p=0.26; mechanical v. cut herbicide, p=0.12). On sandy loam soils (WEB), cut herbicide plots had higher relative species turnover than control plots the first year following treatment (Tukey's HSD p=0.004). Mechanically treated plots on WEB soils had higher turnover in the first and second year following treatment but did not differ from control plots in the third year following brush removal (Tukey's HSD: 1st year, p=0.012; 2nd year, p=0.025; 3rd year, p=0.240).

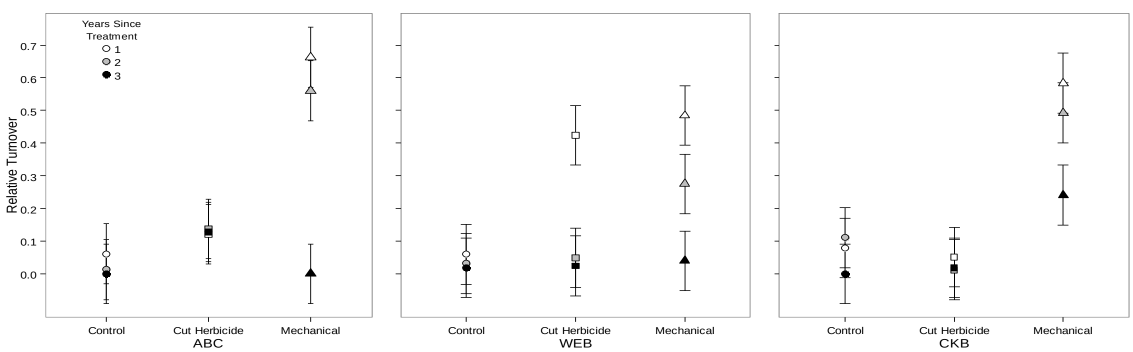


Fig. 10 Species turnover (measured as the sum of the number of species unique to time t plus the number of species unique to time t+1divided by total number of species in time t+ total number of species in time t+1)(mean ±SE) in response to different methods of brush removal (control, cut herbicide, and mechanical) on three different soil types: sandy(ABC), sandy loam (WEB), and clay loam (CKB) one (white), two (gray), and three (black)years following treatment.

Table 8. Differences in turnover in response to different brush removal treatments on different soil types.

	Sum Sq	Mean Sq	NumDF	DenDF	F-value	p
Turnover						
Treatment	1.94	0.97	2	15.52	30.09	<0.001
Soil	0.02	0.01	2	14.93	0.29	0.752
Year	0.97	0.49	2	35.58	15.61	0.000
Pasture	0.16	0.08	2	14.96	2.41	0.124
Treatment:Soil	0.29	0.07	4	14.85	2.21	0.118
Treatment:Year	0.74	0.19	4	35.54	5.77	0.001
Soil:Year	0.16	0.04	4	42.28	1.14	0.350
Treatment:Soil:Year	0.33	0.04	8	41.84	1.27	0.285

Relative Disturbance Resulting from Treatments

Change in percent bare-ground from pretreatment sampling to the sampling period following initiation of treatment differed among treatments and across soils. Differences were similar among treatments on sandy loam WEB and clay CKB soils (Tukey HSD: WEB v. CKB, p=0.97). However, roller-chopping increased bare-ground more on sandy ABC plots than on clay plots and sandy loam plots (Tukey HSD: ABC v. WEB, p=0.03; ABC v. CKB, p=0.001). In fact mean percent bare-ground was almost 60% greater on sandy soils in the sampling period following treatment than during pretreatment sampling, while it was only 6 and 19% greater after treatment on clay and sandy loam soils respectively.

Community Composition

Permutational multivariate analysis of variance revealed a significant treatment by soil interaction (table 10). In sandy ABC soils (shown in black in Fig. 11), mechanically treated plots separated from control and cut herbicide plots in ordination space. This shift was driven by an increased abundance of *Heteropogon contortus*, *Eragrostis barrelieri*, and *Cenchrus spinifex*, as evidenced by these species' occupation of the same ordination space as mechanically treated plots on ABC soils (Fig. 11). On sandy loam soils, there was no directional shift in community composition as a result of brush control treatments. All three treatments occupied the same ordination space on these soils (Fig. 11). Similarly, there was no compositional shift relative to treatments on clay soils. There was also much less variation in community composition on clay soils. All three treatments were tightly clustered in ordination space while the mechanical treatments showed greater variation in community composition on sandy and sandy loam soils (Fig. 11).

Table 9: Permutational multivariate analysis of variance results of tests for differences in grass communities resulting from different brush removal treatments on different soil types in 2010 (representing pretreatment conditions) through 2013.

	Df	Sums Sq	Mean Sq	F-value	R^2	p
Treatment	2	1.68	0.84	3.63	0.02	<0.001
Soil	2	16.30	8.15	35.21	0.24	<0.001
Year	1	0.97	0.97	4.17	0.01	0.002
Treatment:Soil	4	2.96	0.74	3.20	0.04	<0.001
Treatment:Year	2	0.47	0.24	1.02	0.01	0.410
Soil:Year	2	0.51	0.25	1.10	0.01	0.316
Treatment:Soil:Year	4	0.57	0.14	0.62	0.01	0.963
Residuals	198	45.83	0.23		0.66	
Total	215	69.28			1.00	

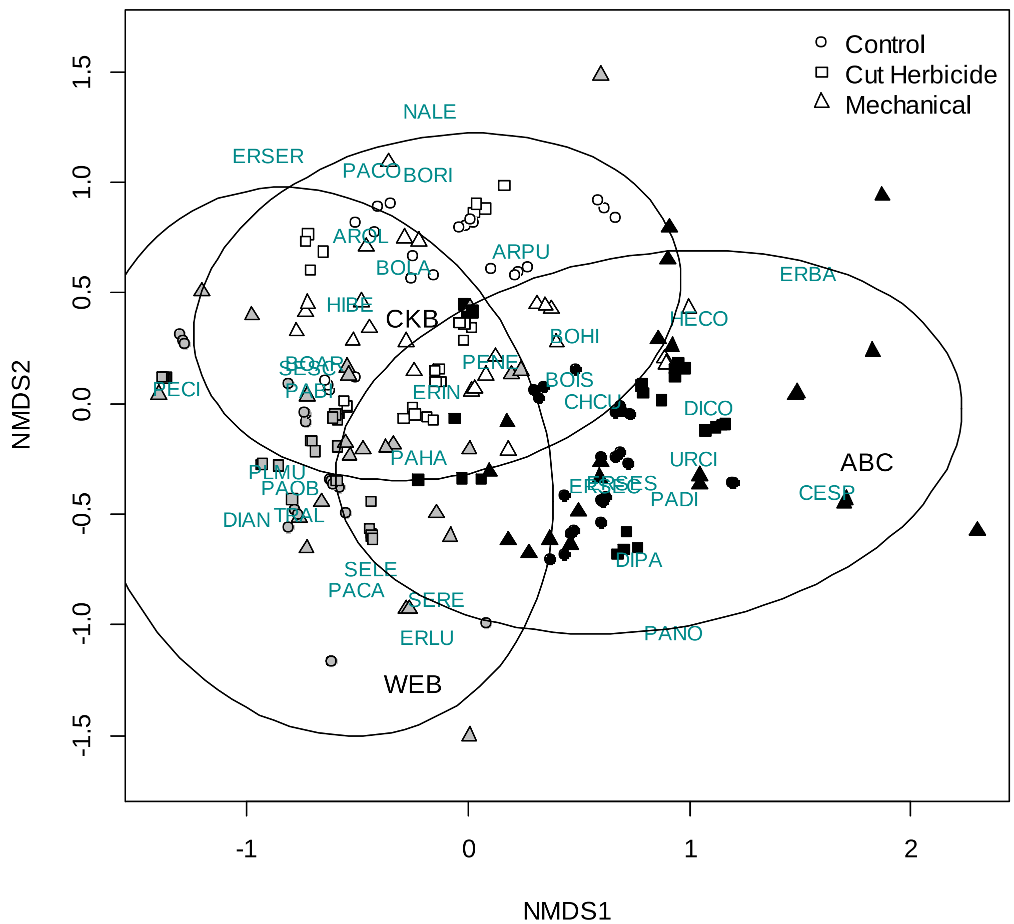


Fig. 11 Nonmetric multidimensional scaling (NMDS) of the grass community for years 2010 (pretreatment) through 2013. Treatments and years are coded as follows: Shapes represent brush removal treatments of control (circles), cut herbicide (squares), and mechanical (triangles); Colors represent soils: ABC (black), WEB (gray), and CKB(white). Ellipses delineate multidimensional standard errors for the different soils.

Discussion

The results of this study showed differential perennial grass mortality, species turnover, and grass community composition shifts among core rangeland restoration treatments and across soils. Woody plants compete with the herbaceous understory in mixed woody-herbaceous systems for essential resources such as light, nutrients, and water (Smit and Rethman 2000). As a result, brush removal often results in higher grass abundance in previously encroached rangelands (Ansley et al. 2004, Throop and Archer 2007). However, as this study shows, grass community response to release from competition with shrubs is not static, but rather varies depending upon brush removal method employed, post removal conditions, and underlying biotic and abiotic characteristics of the system (Morton and Melgoza 1991, Scholes and Archer 1997, McClaran and Angell 2006). This study shows high grass mortality in mechanically treated plots where intense soil disturbance caused direct damage to perennial grasses. Despite high levels of mortality, on finer soils, the herbaceous community did not change following mechanical treatment, but maintained its pre-treatment composition. However, on coarse sandy soils, high levels of mortality resulted in a compositional shift toward annuals and "weedy" perennials capable of rapid colonization following disturbance.

The effects of increased soil resource availability on the herbaceous community following brush removal are mediated by the physiology of the individual species present as they interact with one another and with the abiotic environment (Scholes and Archer 1997, Bestelmeyer et al. 2006, McClaran and Angell 2006) In this study, I found highest levels of mortality from mechanical treatments in intermediate textured WEB soils. This is likely the result of an interaction between the extent of soil disturbance which often varies with soil texture (Pinard et al. 2000) and the individual tolerances of the species present on the different soils to disturbance (Scholes 1990, Buonopane et al. 2005). These intermediate textured soils also had high levels of recruitment, but lower relative species turnover, suggesting that despite high mortality following mechanical brush removal, the grass community on this soil type rebounded quickly following the release from competition with woody brush and was not replaced with new species following mechanical disturbance. In fact, intermediate WEB soils also had the highest levels of recruitment on chemically treated plots as well, suggesting the species present on this soil type were able to exploit increased resource availability following brush removal. *Hilaria belangeri* and *Urochloa ciliatissima* were both abundant on sandy loam WEB soils before and after brush removal. These grasses exhibit a rhizomatous growth form which can more quickly colonize disturbed areas through vegetative spread than cespitose species that rely on sexual reproduction (Mack 1989, Skarpe 2001).

Other studies have shown that species turnover differs with scale of inquiry. Turnover is found to be higher at smaller scales, with response to disturbance more stable at broader scales (van der Maarel and Sykes 1993, Plotkin et al. 2000, Ulrich and Buszko 2003, Bossuyt and Hermy 2004). The results of this study suggest that variation in species turnover on soils of differing textures likely contributes to this difference in turnover with respect to scale. This is consistent with findings that plant species turnover is highly dependent on local biotic and abiotic conditions (van der Maarel and

Sykes 1993, Holt et al. 1995). Thus as heterogeneity of conditions increases with scale, localized high rates of turnover are offset by lower rates elsewhere in the landscape.

Despite higher levels of mortality on all soils following mechanical brush removal, compositional shifts only occurred on sandy soils. Sandy soils had lower levels of mortality and recruitment following mechanical brush removal, but higher relative species turnover, suggesting that community trajectories following disturbance on sandy soils diverged, with new species replacing the individuals killed by the mechanical treatment more often than on the other soils. The shift in species composition on mechanically treated sandy soils was driven by increases in *Heteropogon Contortus, Cenchrus spinifex, and Eragrostis barrelieri.* These species were able to become dominant because soil disturbance from mechanical treatments was more intense on sandy soils. Bare-ground on sandy soils increased by almost 60% on average following mechanical treatment, while bare-ground on sandy loam and clay soils increased by only 19 and 6% respectively. This is consistent with other studies that showed the extent of soil disturbance from mechanical equipment to be greater on coarser soils (Jusoff and Majid 1992, Pinard et al. 2000). While competitive environments are altered by all levels of disturbance (Grime 1977, Huston 1979), patches of disturbed soil interspersed among patches of established individuals create a more favorable environment for perennial species, as they can spread vegetatively or through seed from the established patches into adjacent disturbed sites where there is little competition for the new individual (Paine and Levin 1981). In the mechanically treated sandy soils of this study, however, high levels of bare-ground with few established individuals remaining to spread into the newly available habitat favored “weedy” species such as *Heteropogon Contortus, Cenchrus spinifex, and Eragrostis barrelieri* because they readily establish in disturbed areas where there is little competition from established individuals.

The competitive advantage of *Heteropogon Contortus, Cenchrus spinifex, and Eragrostis barrelieri* over longer-lived perennial species is also likely related to the lower water-holding capacity of course sandy soils which favors plants that are able to react quickly to biologically available water (Hamerlynck et al. 2002, Huxman et al. 2005). Annual species, such as *Eragrostis barrelieri* have high reproductive output and high allocation of resources to rapid growth, allowing them to respond quickly and efficiently to resource availability and reduced competition following high levels of disturbance (Grime 1977). Similarly, while *Heteropogon contortus* and *Cenchrus spinifex* are both perennial species, *Heteropogon* has been found to readily establish in disturbed areas by taking advantage of episodically favorable precipitation conditions and diminished competition (Dye and Walker 1987). *Cenchrus* is a short-lived perennial which is self-compatible and therefore capable of high reproductive output (Liebman et al. 2001). It is considered a “weedy” species that colonizes quickly in disturbed areas, especially on sandy soils (Matocha et al. 2010). Increases in these species comport with findings in other brush removal studies of shifts to dominance of “increaser” species, or species adapted to exploit disturbance (Bedunah and Sosebee 1984, Angassa 2002, Angassa and Oba 2009, DeMaso et al. 2013). Community shift can be transient, however. It is difficult to know whether the shift to annual and “weedy” perennial species will persist, and it can be highly dependent on soils and post-disturbance conditions. In one study, a shift in dominance to *Aristida spp.* following mesquite removal persisted long after mesquite regained pre-removal cover, but only in areas with sufficient precipitation (McClaran and Angell 2006), while in others, compositional shifts were short-lived (Rogers et al. 2004, Browning et al. 2008).

Management Implications

Understanding of the potential effects of management actions on mortality and turnover of perennial grass species and the potential for those effects to translate into a community shift toward less desirable species is necessary for managers to achieve restoration goals in woody brush encroached semiarid rangelands. Interventions that lead to high levels of soil disturbance can cause high levels of perennial grass mortality and lead to higher likelihood of community compositional shifts depending upon the specific propagules available for colonization, and post-treatment climatic conditions (Pacala et al. 1996, Laurance et al. 2006). Managers should therefore be cautious in applying mechanical treatments, especially on sandy soils that undergo more intense disturbance from heavy machinery than finer soils. Historically, managers debated the use of fire as a rangeland management tool due to concerns regarding excessive perennial grass mortality following prescribed fire (Wright and Klemmedson 1965, Wright 1982, Johnson and Strang 1983). The same concerns have not been expressed regarding the use of chemical and mechanical brush control methods in encroached rangelands (Kreuter et al. 2001, McGinty and Ueckert 2001, Hamilton 2004). The findings of this study show that perennial grass mortality is higher following mechanical brush control than from fire. In fact, levels of mortality from fire were similar to baseline mortality in control plots. This finding is supported by a long-term study in a similar system which found that even high intensity fire conducted during drought when perennial grasses were already experiencing stress did not result in the loss of desirable forage species from the system (Taylor et al 2011), showing that hesitation to use fire for fear of grass mortality is not empirically grounded. A more complete understanding of the effects of core rangeland restoration techniques might provide incentive for managers to use the most effective and efficient methods for brush management and removal rather than basing management decisions on social conventions. In addition, studies such as this one that elucidate the potential for different trajectories of change following brush removal in communities with different baseline biotic and abiotic conditions provide valuable information for the development of frameworks which can be applied on the landscape to effectively manage brush encroached rangelands.

CHAPTER IV

THRESHOLDS OF DROUGHT-INDUCED WOODY PLANT MORTALITY IN AN ENCROACHED SEMIARID SAVANNA

Introduction

While great uncertainty surrounds predictions regarding future precipitation patterns resulting from climate change in the next century, predictions all suggest that regardless of shifts in mean precipitation, variability will increase (Breshears et al. 2008, Adams et al. 2009). Extreme variability in precipitation patterns increases the likelihood of drought events (Anderegg et al. 2013a). As a result, the frequency, extent (both temporal and spatial), and severity of drought is likely to increase even in areas with little change in mean annual precipitation (Dai 2011, 2013). There is much concern that large-scale forest die-off could increase in coming decades with large predicted temporal and spatial shifts in soil water availability resulting from increases in precipitation variability and drought events (McDowell et al. 2008). Indeed there is already evidence of increases in forest die-off in most forested biomes worldwide in the past several decades (Allen et al. 2010, Anderegg et al. 2013b). Drought-induced forest die-offs could have strong impacts on ecosystem structure and functioning. While tree mortality occurs naturally in any system, massive die-offs that are species and site specific could have lasting effects on ecosystem heterogeneity (Floyd et al. 2009, Anderegg et al. 2013b), understory composition (Kane et al. 2011, Anderegg et al. 2012), ecohydrological processes (Adams et al. 2012), biogeochemical cycling (Edburg et al.

2012), disturbance dynamics (Bigler et al. 2005), and provision of ecosystem services to human populations (Anderegg et al. 2013b), including carbon sequestration (Allen et al. 2010, Pan et al. 2011).

Drought-induced mortality events often emerge abruptly during prolonged drought events rather than exhibiting a gradual increase in mortality across the duration of the drought (Allen et al. 2010, Carnicer et al. 2011, Anderegg et al. 2013a). This sudden increase in mortality in response to drought events is likely related to physiological tipping-points, or thresholds related to water-stress (Lenton et al. 2008). Plants are adapted to deal with water-stress, but prolonged or severe drought stress can lead to xylem cavitation, diminishing water transport to leaves (Pockman and Sperry 2000, Carnicer et al. 2011). Woody plants experience mortality after crossing a certain species-specific threshold of conductivity loss, resulting in apparently abrupt mortality at some point during the course of an extended drought (Urli et al. 2013). Given that physiological thresholds are species-specific, die-off can be large-scale in forests dominated by a single species. Whereas die-off in mixed species stands are more likely to occur gradually as species with lower water stress succumb to drought induced hydraulic failure first followed by those with higher water-stress thresholds as drought intensity or duration increases (Bond and Kavanagh 1999, Zweifel et al. 2009). Additionally, drought mortality in trees is not necessarily always the result of hydraulic failure. Other mechanisms, such as carbon starvation resulting from stomatal closure in response to limited water availability have been posited to play a role in forest die-offs (McDowell and Sevanto 2010, Sevanto et al. 2014). Carbon starvation is more likely during prolonged drought events as plants can regulate carbon allocation, but ultimately metabolic needs will exceed input if stomatal closure is long-term (McDowell 2011). This could result in large increases in mortality beyond the initial die-off from hydraulic failure.

I initially established this study to assess differences in mortality extent and pattern resulting from the droughts of the 1950s and the 2000s after observing an abrupt drought-induced die-off of ashe juniper in an area where drought mortality had been assessed following the 1950s drought (Merrill and Young 1959) and land-use had remained consistent since that 1959 study. The results of this analysis are detailed in Twidwell et al. (2014) and showed that the extended severe drought of the 1950s resulted in greater levels of mortality than that observed in 2011, and that woody plant mortality resulting from the 2000s drought was highly species-specific and dependent on topoedaphic characteristics and land management. However, the drought persisted through 2013, and while it never attained the severity (Palmer Drought Severity Index values of -4) of the 1950s drought, the duration of the two drought periods was similar. Given the additional 2 years of drought, our objectives in this study are to determine if additional years of water stress would increase rates of mortality over those observed in 2011. I also wanted to determine if there were changes in the patterns of dieback across soils and pastoral management. An additional objective of this study is to explore the effects of the prolonged drought on herbaceous cover across different soils and pastoral treatments and determine the extent to which temporal and spatial patterns of grass die-back differ from woody die-back.

Methods

Study Site

This study was conducted at the Sonora, Texas A&M AgriLife Research Station (31°N; 100°W) on the Edwards Plateau, where long-term research on vegetation dynamics has been occurring for over 90 years and experimental treatments featuring different browsing manipulations have been consistently applied in some pastoral units since 1948 (Fuhlendorf & Smeins 1997; Taylor et al. 2012). The research station is positioned at an elevation of 730 m. The mean frost free period is 240 days (station records, 1919-2011). Mean annual precipitation is 570 mm (station records, 1919-2011) but is highly variable within and among years (range = 156-1054 mm). Rainfall has a pronounced bimodal regime with peak levels occurring during May-June and September-October with frequently occurring prolonged droughts during the summer months. The dominant ecosystem on the research station historically was live oak savanna, but many areas, including the location of the study have transitioned into patches of closed canopy ashe juniper forest interspersed among more open oak savanna. The dominant woody plant species are live oak (*Quercus virginiana* Mill.), pungent oak (*Quercus pungens* Liebm.), Texas persimmon (*Diospyros texana* Scheele), Ashe juniper (*Juniperus ashei* J. Buchholz), *Celtis* spp., catclaw (*Acacia greggii* A. Gray), algerita [*Mahonia trifoliolata* (Moric.) Fedde], prickly ash [*Zanthoxylum fagara* (L.) Sarg.], netleaf forestiera (*Forestiera reticulata* Torr.), and downy forestiera (*Forestiera pubescens* Nutt.).

Experimental Design and Data Collection

In 1949, 10 belt transects, 30.48 m long and 0.3048 m wide, were established to estimate the cover of woody plant species in six pastoral units at the Texas A&M Agrilife Research Station located between Sonora and Rocksprings (Merrill and Young 1959). In 1958, transects were resampled to determine the effect of the drought that occurred in 1951-1957 on woody plant mortality and cover.

I followed the design and sampling of Merrill and Young (1959) to compare the effects of the 1950s drought to woody plant mortality levels and dieback that have occurred as a result of the drought of the 2000s, the second worst drought period at this site in the last 90 years (Figure 11). In June of 2011 ten belt transects, each 30.48 m long and 0.3048 m wide, were randomly established in pastoral units that have been managed consistently since Merrill and Young (1959) established their study in 1949. Pastoral units were 40 ha and included a livestock exclosure unit, a high-fenced deer and livestock exclosure unit, and two units that have been annually stocked with livestock under Merrill's four-pasture deferred rotation system since the 1949 (Merrill 1954). Since drought conditions continued following the 2011 sampling period, I resampled the transects in May 2013 in order to determine additional mortality occurring since the 2011.

Following the sampling protocol of Merrill and Young (1959), woody plants located along each transect were classified in three categories: plants alive, plants with trunks or stems dead but with resprouting stems from the base, and plants dead. Plants were assumed dead if they had no live foliage at the time of the sampling. Canopy cover of each species in the understory (< 2.1 m) and overstory (≥ 2.1 m) and total woody canopy cover were measured along each transect using the line-intercept method (Floyd and Anderson 1987). Additionally, I measured dieback by determining the portion of tree crown intercepting a transect which had no live foliage. A height of 2.1 m was chosen to separate understory and overstory layers to remain consistent with Merrill and Young (1959). I also measured cover of grass and bare-ground intersecting each transect for more than 0.25m. Each transect was characterized as being located on one of three soil depths: deep soils, which are typically found in the lowland areas, shallow soils, which are associated with upper divides, and rocky draws, which are large areas of exposed bedrock.

Data Analysis

Repeated measures analysis of variance (ANOVA) was performed in order to determine if significant increases in mortality had occurred during the two years of drought subsequent to the 2011 sampling period. I looked at differences in total mortality and each dominant species individually. I included pastoral treatment, soils, and their interactions with year of sampling to determine if significant increases in mortality in 2013 differed among pastoral treatments or soils. Context dependent mortality resulting from the drought was tested for each species. The proportion of variation (R^2) in mortality of all individuals, understory individuals, and overstory individuals explained by the total woody plant cover, total grass cover, and density of patches of cover type along a given transect was determined for all woody plant species and for each of the dominant woody plant species individually. Density of patches of cover type is a measure of the configuration of woody brush and grass along a transect. I counted the number of distinct patches of woody plant cover and grass cover which intersected the transect for more than 0.25m and divided the number by the total cover of woody plants along the transect. Using this index, I were able to characterize the extent of clustering of woody plants along the transect. Transects with higher patch density had less clustering of woody plants and instead were comprised of more evenly-spaced woody plants separated by patches of open grassland).

I calculated Palmer Drought Severity Index (PDSI) values for the history of the Sonora, Texas Agrilife Research Station using the Self-Calibrating Palmer Drought Severity Index (SC-PDSI; available at http://greenleaf.unl.edu/) and station weather records dating back to 1919. The PDSI method I used can overestimate changes in drought occurring over time because it uses a simplified model of evapotranspiration (Sheffield et al. 2012). However, historical on-site weather records were not available to use more physically complex PDSI methods. I used the available water capacity of the dominant soil type in this region, Tarrant soils, in the PDSI calculation (Soil Survey 2012). Precipitation anomaly was calculated as the difference between monthly on-site precipitation according to station records and the National Oceanic and Atmospheric Administration's (NOAA) 1981–2010 U.S. Climate Normals average for Sonora,Texas (Arguez et al. 2012).

Results

Palmer Drought Severity

The drought of the 1950s was more prolonged and severe than any period since the station was established in 1919 (Fig. 12). For 132 consecutive weeks (1951-1954), PDSI values were ≤ -3 (the PDSI value associated with "severe drought"; Heim 2002). In fact, ninety-nine of the top 100 drought weeks, according to PDSI calculations from 1919-2011, occurred between 1951 and 1956. Ninety-seven of those weeks had PDSI values ≤ -4. This extended drought period was followed by a second severe drought (PDSI ≤ -3) that occurred for 32 consecutive weeks from 1957-1958. In more recent

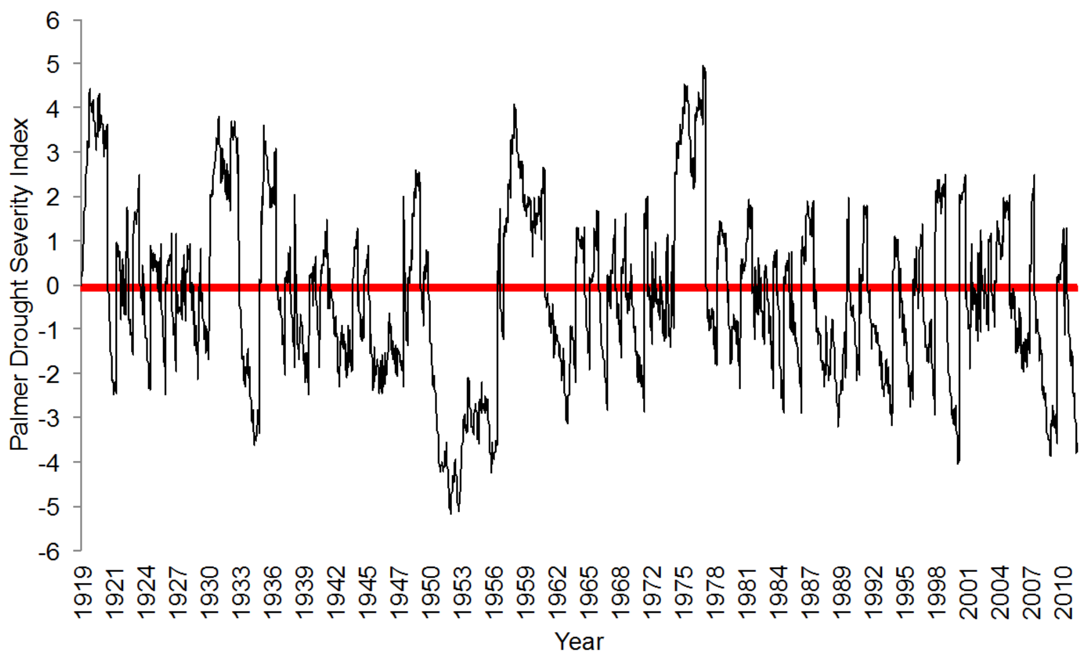


Fig 12. Palmer drought severity index for the century leading up to the study. Negative values indicate drought. Values below -3 are indications of severe drought.

years, PDSI values have exhibited high intra-annual variability (Fig. 12), with severe droughts occurring in 2000, 2008, 2009, and 2011. In 2000, PDSI $\leq$ -3 for 12 consecutive weeks and was the only week outside the 1950s to reach a value below -4 (PDSI = -4.03; ranked as 93rd worst drought week since 1919). The drought of 2008-2009 was more severe for a longer period, with PDSI values $\leq$ -3 for 42 of 46 weeks (low PDSI = -3.87). At the time of conducting this study in June 2011, PDSI was -3.66 and had reached a low of -3.78 four weeks prior. The closest comparison outside the 1950s and 2000s occurred from 1934-1935, when PDSI $\leq$ -3 for 25 consecutive weeks and reached a low of -3.62.

Precipitation Anomaly

Beginning in September 2010, precipitation at the station was below the 1981-2010 average for the area every month (Fig. 13). Then, late in 2011 and early in 2012, there were several months with higher than average monthly precipitation, but they were immediately followed by a period of 13 months during which precipitation in 11 months fell below the monthly average and in 2 months was slightly (<25mm) above (Fig. 13). This 11 month stretch directly preceded the second sampling period in May 2013.

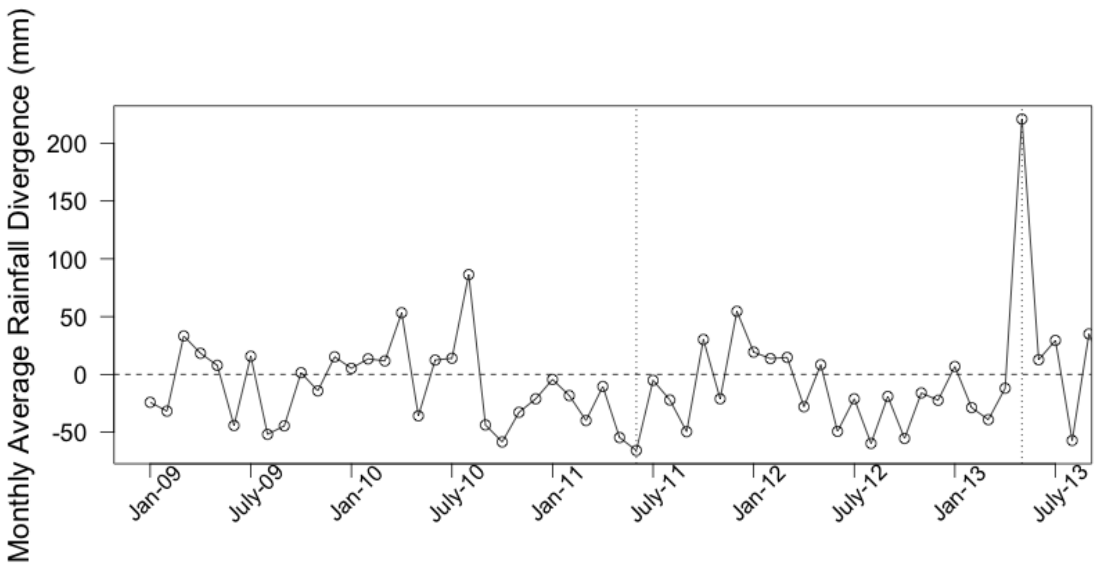


Fig. 13 Precipitation anomaly for the Sonora Agrilife Research Center during January 2009 through September 2013. Points represent monthly divergence from the thirty-year monthly average. Vertical dashed lines indicate sampling periods.

Mortality and Cover

In 2011, the drought of the 2000s had killed 22% of trees (128 of 580 individuals), decreasing woody cover on the transects from 58±4% to 41±% (Twidwell et al. 2014). Mortality of all species present had only increased to 24% (142 out of 583 individuals) by 2013 after 14 additional months of below average precipitation in the study area. Additionally, mortality of the dominant species did not increase significantly following the 2011 sampling period (F=0.071, p=0.791) (Fig. 14). Ashe Juniper mortality increased from 21% to 25% and Shin Oak mortality increased from 26% to 28%, but these increases were not statistically significant (JUAS: F=0.295, p=0.589, QUPI: F=0.382, p=0.542). Live Oak mortality did not increase between 2011 and 2013

(F=0.408, p=0.530). Cover dropped from 58±4% before the drought of the 2000s to 41±4% in 2011 (Twidwell et al. 2014) and had not significantly decreased between 2011 and 2013 when it was 39±4%. Similarly patterns of dieback did not differ from the patterns in 2011 with higher mortality in deep soils and no difference among pastoral treatments or between understory and overstory layers (Twidwell et al. 2014), which was not surprising given that there was not enough additional mortality to alter patters of dieback already established by 2011. Grass cover, however, did show a significant decrease from 12.3±1.2 % in 2011 to 4.8±0.7% in 2013 (Fig. 15).

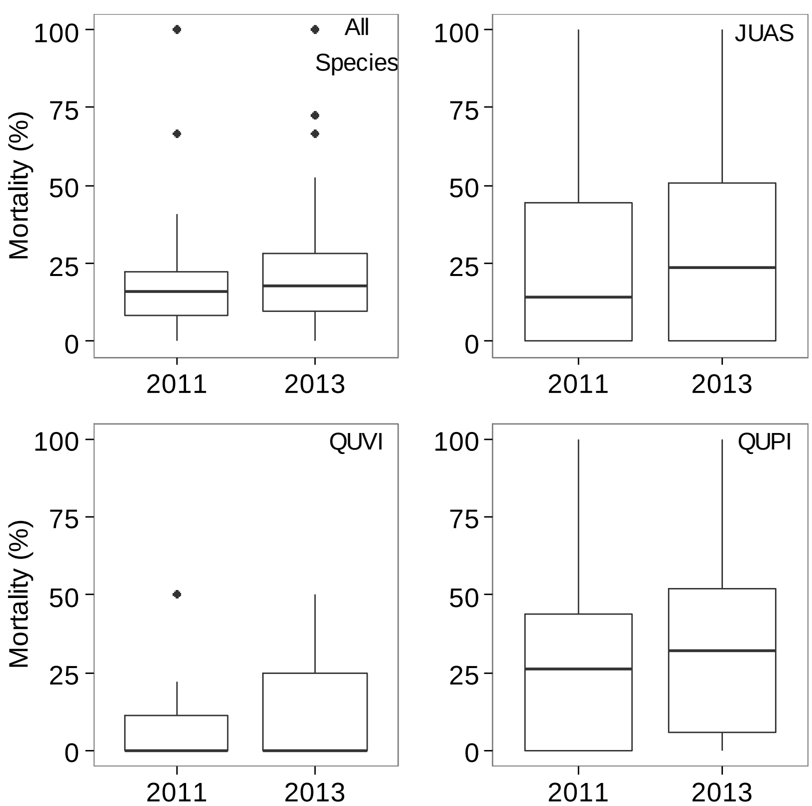


Fig. 14 Differences in percent mortality observed in 2011 and 2013 for all species, Ashe Juniper (JUAS), Live Oak (QUVI), and Shin Oak (QUPI).

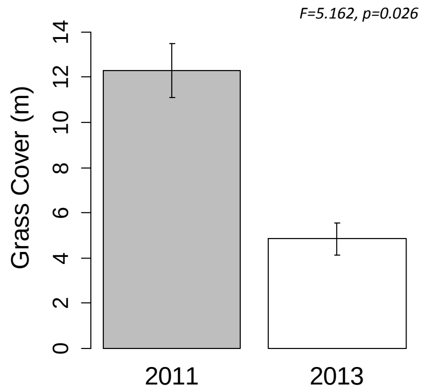


Fig. 15 Differences in grass cover along transects in 2011 and 2013.

Context-Specific Dieback

Neither total, understory, or overstory mortality was correlated with either grass cover or woody cover (Table 11). However, patch density was significantly correlated with both understory ($F=4.227$, $p=0.043$) and overstory ($F=4.422$, $p=0.019$) mortality. Patch density did not explain a large amount of the variability in mortality however (understory $R^2=0.14$, overstory $R^2=0.23$) (Fig. 16).

Table 10: Correlation between percent woody plant mortality on a transect and percent grass cover, percent woody cover, and patch density (a measure of patchiness of woody plant distribution).

	R^2	F	p
Grass Cover			
All Individuals	-0.03	0.06	0.81
Understory	-0.01	0.92	0.34
Overstory	-0.02	0.42	0.52
Woody Cover			
All Individuals	-0.03	0.01	0.94
Understory	0.01	1.04	0.31
Overstory	-0.03	0.04	0.85
Patch Density			
All Individuals	0.07	1.61	0.21
Understory	0.14	4.23	**0.04**
Overstory	0.23	5.72	**0.02**

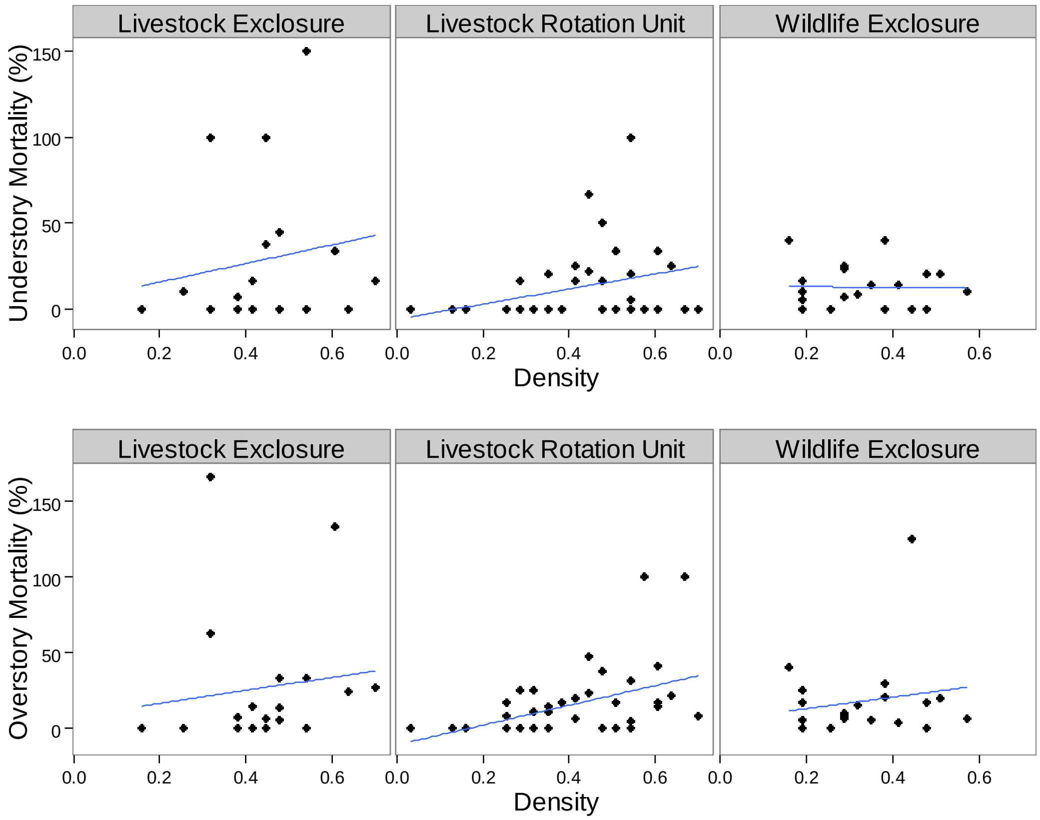


Fig. 16 Regression of understory and overstory mortality of woody plants against patch density (number of patches/area) for each pastoral treatment.

Discussion

Despite continued prolonged drought for almost two years following the first sampling period in 2011, little additional mortality occurred between the 2011 sampling period and the 2013 sampling period. This is consistent with other observations of drought-induced die-back where episodic mortality occurred at some point during prolonged drought and little mortality occurred following the major die-back event (Suarez et al. 2004, Miriti et al. 2007, Koepke et al. 2010). This type of episodic mortality is consistent with theories of drought-induced hydraulic failure where a threshold of conductivity loss leads to mortality (Urli et al. 2013). While these physiological thresholds vary among species, they are fairly consistent within species because they are driven by the structure of the plant vessels and species-specific responses to reduced water availability (Miriti et al. 2007). On average, loss of 50% of stem conductivity has been found to lead to mortality in conifers while approximately 88% leads to death in many angiosperms (Urli et al. 2013). Despite continued drought stress, plants with a higher physiological threshold to water stress that survived the first several months of drought were never pushed over the conductivity loss threshold. This could be the result of greater fluctuation in departure from monthly precipitation normals in the period following the 2011 sampling. Between 2011 and 2013, there was greater variability in precipitation than in the months preceding the 2011 sampling period which were all far below monthly normals. This pattern of mortality with little additional die-off despite continued water stress suggests that carbon starvation is not the dominant mechanism for woody plant mortality at the study site. If carbon starvation were responsible for die-offs here, the lengthened drought period would lead to additional mortality as plants exceeded their reserves of carbon and metabolic demands overtook the ability to survive with limited carbon inputs resulting from prolonged stomatal closure (McDowell and Sevanto 2010, Sevanto et al. 2014). Carbon starvation is less likely in anisohydric species, like our dominant ashe juniper, which do not regulate leaf water potentials through stomatal closure during periods of limited water availability (McDowell et al. 2008). This increases vulnerability to cavitation, but makes carbon starvation less likely as respiration continues throughout the drought (Pockman and Sperry 2000, McDowell 2011).

While there was little additional mortality of woody plants during the two years of drought between sampling periods, grass cover declined precipitously. This decline occurred despite the release from competition with trees for light and soils water attendant to the 19% decrease in woody plant canopy cover from pre-drought levels, suggesting that drought was still severe enough in 2013 and no water was available for grasses to survive let alone expand into gaps created in the ashe juniper canopy as they did in *Juniperus monosperma* woodland after drought dieback in Arizona (Kane et al. 2011). This continued grass mortality for the duration of prolonged drought is consistent with findings of drought-induced mortality of *Bouteloua gracilis* following severe drought in New Mexico (Allen 2007) and large-scale mortality of multiple grass species in Arizona throughout a prolonged drought (McAuliffe et al. 2006).

Differences in response to prolonged drought stress between grasses and woody plants highlight the specificity of physiological stress tolerance thresholds. The physiological characteristics of different plant functional types lead them to interact differentially to pulse stressors of different severity and duration (Schwinning and Sala 2004). As a result, the interaction between functional groups with unique stress-tolerance thresholds and spatiotemporal partitioning of resource availability drives community dynamics and structure in semi-arid and arid systems (Chesson et al. 2004, Schwinning et al. 2004, Knapp et al. 2008). Thus a dramatic shift in spatiotemporal patterns of precipitation in semiarid regions attendant to climate change could lead to community composition or even structural shifts depending on mortality and replacement dynamics of the species present at the start of major drought events (Miriti et al. 2007). The balance between grass and woody plants in savanna systems is thought to be greatly influenced by climatic variables (Walker et al. 1981, Walker and Noy-Meir 1982, Scholes and Archer 1997, Fensham et al. 2005, Bond 2008). Arid environments with unpredictable precipitation patterns favor grasses over trees because grasses can take advantage of pulses of precipitation more readily than woody plants (Scholes and Archer 1997, Chesson et al. 2004). However, as I observed in this study, under severe prolonged drought, grasses experienced continued high levels of mortality while trees died back in large numbers initially, but little additional mortality occurred with prolonged severe drought. This has potential implications for grass-tree dynamics in savannas in the future with global climate change. While grasses are favored by pulsed rainfall events (Dodd and Lauenroth 1997, Schwinning and Sala 2004), trees in semi-arid areas might be more adapted to withstand long-term drought conditions (Schwinning et al. 2004, Bréda et al. 2006). Therefore, potential for shifts in dominance to grasses in woody brush encroached semi-arid savannas under changing climatic conditions will depend on patters of precipitation and drought events. This study highlights the potential for frequent intense flash droughts that cause high levels of tree die back and moderate grass mortality to promote grass dominance, while long duration droughts maintain woody dominance as grasses continue to experience mortality throughout the drought and trees persist after an initial pulse of mortality.

Interestingly, despite a lack of correlation between total woody cover or total grass cover and levels of overstory or understory woody plant mortality, the distribution of woody plant and grass cover along transects (patch density) did have small, but significant effect on woody plant mortality. Two prevailing hypotheses exist currently that have the potential to explain the increase in mortality with increasing patch density. Since higher patch density means lower clustering of trees along a transect, this measurement can be viewed as a proxy for density. Transects with higher patch density have more trees neighboring open grass patches than other trees. Given that patch density means lower local density of trees, density dependent effects are likely driving the correlation. The weakness of the relationship could be due to drought severity masking the density dependent effects (Floyd et al. 2009). One hypothesis for explaining inverse density-dependent mortality is that higher woody plant densities facilitate the growth and survival of small trees during drought when large overstory trees are present. As stress increases, facilitation becomes more important for these individuals (Holzapfel and Mahall 1999). High densities facilitate seedling establishment and survival by providing an environment with increased soil moisture and nutrients and decreased evaporative loss from wind and solar radiation (Zou et al. 2005, Raventós et al. 2010). This has been shown to increase survival of juveniles in pinyon-juniper forests (Floyd et al. 2009), Mediterranean shrublands (Raventos et al. 2010), and temperate deciduous forests (Pages and Michalet 2003). An alternate hypothesis states that the negative relationship between drought-induced mortality and woody plant density occurs because marginal microsite conditions, which are only able to support low densities of woody plants, experience higher mortality rates than more suitable microsites capable of supporting higher densities (Greenwood and Weisberg 2008).

Conclusions

Predictions of greater precipitation variation and attendant increases in likelihood of extreme precipitation events such as drought have fostered grave concerns regarding forest vulnerability to large-scale die-off (Allen et al. 2010). Many areas have already seen large mortality events and in many cases they have led to dramatic shifts in species composition (McAuliffe et al. 2006, Miriti et al. 2007). This is more likely with large-scale episodic die-off than gradual mortality because of the potential for species replacements to occur (Allen 2007). If drought severity or longevity crosses several species physiological thresholds as was observed in the southwestern United States in 2002, those species can be replaced by more drought tolerant species, causing dramatic compositional shifts. In semiarid systems, community structure and function is driven by episodic mortality (Schwinning et al. 2004). Resources in these systems are patchily distributed over space and time, and the species are adapted to the particular distribution. Therefore, distribution of rainfall is likely to be a greater driver of community dynamics in semiarid systems than mean precipitation (Knapp et al. 2008). Given this, episodic mortality like that observed here has the potential to have lasting and cascading effects on plant community structure and functioning in semi-arid systems as droughts increase in intensity and duration in the coming decades.

CHAPTER V

LEGAL BARRIERS TO EFFECTIVE ECOSYSTEM MANAGEMENT: EXPLORING LINKAGES BETWEEN LIABILITY, REGULATIONS, AND PRESCRIBED FIRE

Introduction

An emphasis on fire suppression has altered fire regimes in many ecosystems worldwide (Reinhardt et al. 2008, Moreno et al. 2014). Historical fire regimes played an important role in maintaining many natural systems (Pyne 1982) and regime alterations can have numerous detrimental effects. For instance, a change from frequent low-intensity fires to infrequent high intensity fire in forests of the southeastern United States resulted in the replacement of many loblolly pines by less valuable forest species (Drewa et al. 2002). Suppression of frequent fires in semi-arid rangeland and mesic grasslands leads to shrub encroachment, lowering forage productivity and degrading habitat for grassland birds and mammals, many of which are threatened and endangered worldwide (Knapp et al. 2008, VanAuken 2009, Twidwell et al. 2013c). Often, fire regime alterations also facilitate invasion by non-native species (D'Antonio 2000, Mooney and Hobbs 2000).

In many systems, fire suppression results in vegetation structures that promote more intense fires (Stephens and Ruth 2005, Keane et al. 2008). Severe fires in systems adapted to low-intensity fires can lead to structural and compositional alterations within the plant community, which often reduce ecosystem resilience (Stephens et al. 2014). Often such plant community shifts following severe fire contribute to soil erosion and sedimentation in streams and reservoirs (McNabb and Swanson 1990). Intense fires can also lead to property loss, injury, and loss of life and may require enormous expenditures to bring them under control.

Prescribed fire is a cost-effective tool for range and forest restoration and management (Van Liew et al. 2012). It allows managers to impose a fire regime tailored to their management objectives. However, despite the benefits of prescribed burning for land and fuels management, landowners often choose not to use fire due to fear of liability (Haines et al. 2001, Yoder 2004, 2008, Toledo et al. 2012, Sun and Tolver 2012).

Generally, civil liability standards in the United States for prescribed fire fall into three categories; strict liability, simple negligence, and gross negligence. A rule specifying strict liability holds burners liable for any property damage caused by an escaped prescribed fire, regardless of the action of the burner; it creates the highest level of liability for anyone using prescribed fire. Only five states have standards that suggest the stringency of strict liability, although the statutes do not all explicitly state that strict liability is the standard. Hawaii, for example, makes escape of fire evidence that, if unrebutted, is sufficient to prove willfulness, malice, or negligence (HRS§185-7). Simple negligence standards require the burner to practice reasonable care in applying a prescribed burn; they are the most common rules for prescribed fire and require the plaintiff to show negligence by the defendant in order for the burner to be liable for damage caused by escaped wildfire. They can either be explicitly stated statutorily as in Texas (Tex.Nat.Res.Code§153.081), or established through case law as in New Mexico.

Gross negligence liability standards provide that, if a burner follows a set of codified regulations regarding burning, a plaintiff must show reckless disregard of the duty of care owed others by the burner. Usually, in states with gross negligence rules, simple negligence will apply if the regulatory requirements are not fulfilled (Sun 2006, Yoder 2008, Sun and Tolver 2012). Statutes identifying gross negligence liability standards have recently been enacted in several states [e.g. Florida Prescribed Burning Act (590,125(3))] (Sun 2006, Coalition of Prescribed Fire Councils 2012).

Recognizing the considerable ecosystem changes that resulted from prolonged fire suppression policies and the need to make prescribed burning available as a management option, many states, especially in the southeastern United States, have undergone statutory reform in order to promote the safe use of prescribed fire. The stated purpose of these statutory reforms, often called "Right to burn" or "Prescribed burning" acts, is to encourage prescribed burning for resource protection, public safety, and land management [e.g., Georgia Prescribed Burning Act (O.C.G.A. § 12-6-146), Tennessee Prescribed Burning Act (T.C.A. § 68-102-146)]. These reforms usually include a statutory statement of the liability standard to be applied in case of loss of control over a prescribed fire. In some cases, simple negligence is applied, but in several states, gross negligence standards have been adopted. Most of the statutes also include regulations that ensure that the burn is carried out safely, and limited liability in the form of a gross negligence standard can be used to incentivize prescribed burn practitioners to receive training and undertake various safety precautions prior to burning. For instance, in Florida, burners who have been certified by the state certified prescribed burn program and have written burn plans and adequate personnel and firebreaks will be subject to a gross negligence standard in court, whereas, those not certified to burn or lacking the requisite preventative measures during the burn will face the more stringent simple negligence standard in the event of an escape.

It is uncertain that these reforms are achieving their intended purpose of encouraging greater use of prescribed burning while maintaining safety and limiting escapes. Stringent regulations included in statutory reforms and mandated for protection under the gross negligence standards might serve as a disincentive to burning (McCullers 2013). Additionally, many states have adopted stringent regulatory requirements for protection under prescribed fire acts but have not suitably incentivized burners to receive training and follow regulations by providing limited liability for those appropriately trained and prepared. While it might be easier to prove negligence if a burner has not followed all regulatory precautions outlined in the statute, he would still be subject to the same level of liability as a burner who had undergone training and planned for the burn following statutory mandates in states that have opted for simple negligence standards for all burners. Therefore, there is little incentive for a land owner to undergo time consuming training in states that retain the same liability standard for certified and noncertified prescribed burn practitioners. Furthermore, some regulations might be more restrictive than others. For instance burn ban regulations that allow counties to ban all burning during periods of high fire danger could limit prescribed fire use more than those with exceptions for certified prescribed burn managers (CPBMs) (e.g. V.T.C.A., Natural Resources Code § 153.004).

Resistance to the use of prescribed fire is strong among private land managers despite the advantages it offers. Even managers who are aware of the benefits and desirous of inexpensive means to achieve management objectives avoid using prescribed fire, often citing potential liability as a major reason for hesitation (Brenner and Wade 2000, Yoder 2008). Several recent studies have examined prescribed fire liability (Haines and Cleaves 1999, Haines et al. 2001, Yoder 2004, 2008, Sun 2006), but none have explored the relationships between liability, regulation, and landowner use of prescribed fire. Specifically, none of these studies included the amount of private land treated with prescribed fire as a variable. In addition, these studies include an examination of state burning laws as of 2005 at the latest, but there have been additional state reforms since 2005 [e.g. Tennessee Prescribed Burning Act (T.C.A. § 68-102-146)] that must be included for an up-to-date exploration of legal drivers of private land burning.

Herein, I attempt to assess the impact of statutory reforms that apply to prescribed burning and identify legal incentives and impediments to prescribed fire application for range and forest restoration and management, as well as hazardous fuel reduction. Specifically, I explore the relationship between prescribed burning laws and the decisions land managers make about fire. To achieve this I compare the use of prescribed fire by landowners in contiguous counties of different states in the southeastern United States with different regulations and legal liability standards.

Data & Methods

Legal Variables

I performed a detailed analysis of the legality regarding prescribed fire for the states of the southeastern United States. I focused our analysis in this region because the state forest services and prescribed fire councils of the southeastern United States maintain complete records of prescribed burn permitting on private land that provided highly reliable data. Few other areas keep such complete and reliable records of prescribed burning on private land. Additionally some states, such as Nevada and California, where some data were available had additional layers of regulations for certain counties related to water and air quality control which would complicate an analysis of state level statutory law.

I completed a search of state statutes and state appellate case law in the Westlaw legal database (Thomson Reuters, New York, NY) using the keywords “prescribed burn”, “prescribed burning”, “prescribed fire”, “controlled burn”, “controlled burning”, and “controlled fire” for six southeastern states: Florida, Georgia, Alabama, South Carolina, North Carolina, and Tennessee. I then reviewed each statute and case to determine the applicable legally relevant variables, including civil liability standard as stated in statutes and applied in case law, regulations, and the use of burn bans to limit prescribed burning during potentially dangerous fire weather.

I identified four requirements for prescribed burning from the state statutes: written burn plans, presence of a CPBM, adequate personnel and firebreaks, and burn permits. Written burn plans prescribe the conditions under which the burn will occur. They define the weather conditions under which the burn will take place, the equipment and personnel that will be on hand during the prescribed fire, and illustrate the ignition technique that will be employed. Some states require a CPBM to remain at the site of the burn until the burn is completed. Some also require that adequate personnel and firebreaks be in place at the time of the burn although few statutes define what is meant by “adequate”. Burn permits are required for burning in all states included in the study. They are applied for electronically or via telephone and require the applicant to list the date, type, location, and areal extent of the burn. They are always issued if there is no burn ban in place in the county. I categorized counties into those requiring only a burn permit for a prescribed burn and those requiring a permit plus one or more of the three additional requirements.

Prescribed Fire Data

I collected county-level permit data for prescribed burning on private land from 2008-2013 in the six southeastern states. However, I excluded Tennessee from the analysis because complete data on private land prescribed burning per county were unavailable. In addition, I included only counties that share a state border with a county in a state with a different liability standard (Fig. 17). For instance, Alabama has a simple negligence standard and Georgia has a gross negligence standard, so the counties that form the border between Alabama and Georgia are included in the analysis. Focusing the analysis on contiguous counties separated by a state border provides a control on observable and unobservable factors influencing the use of

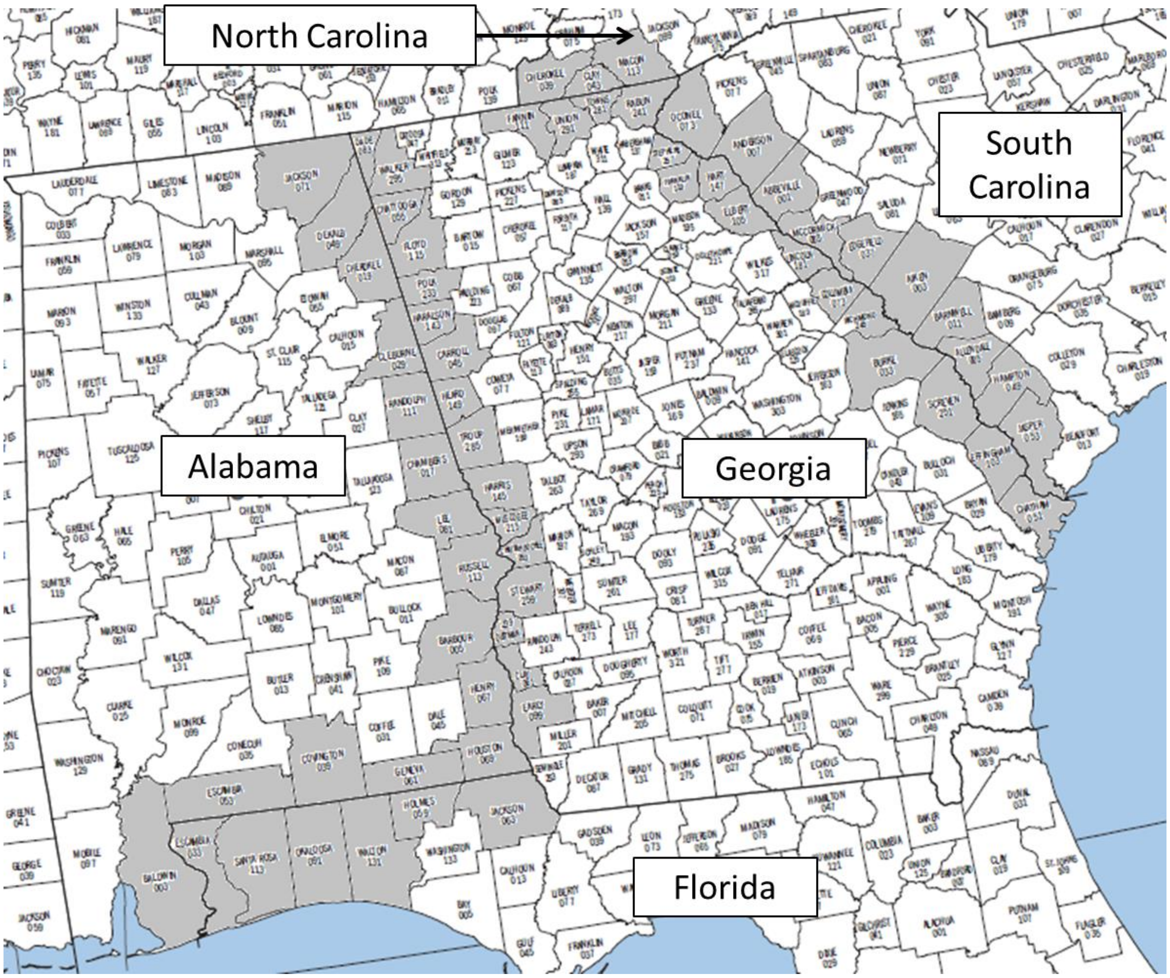


Fig. 17 Study area. Gray counties are included as matched pairs in the analysis

prescribed fire that are unrelated to state regulations and liability standards (Holcombe and Lacombe 2004, Huang 2008). This control is necessary because weather, land cover, vegetation type, topography, and many other variables are likely to play a role in a landowner's decision to conduct a prescribed burn. Matched experimental units that have similar levels of variables such as those listed above limit the potential for omitted variable bias.

Burn permit data provide a proxy for the number of fires and acres burned in a given county for a given year. While not every burn permitted is necessarily carried out, the percentage of burns completed should not differ between matched counties. The permit application process in the states selected are similar with none more onerous than others, so a decision not to follow through with a burn would most likely be related to weather or other factors controlled for through matching counties. Acres permitted per county per year were analyzed as a percentage of total privately owned forest, range, and pasture land for that county and are hereafter referred to as percent land area burned. Number of fires per county per year includes range, forest, and pasture burns permitted and does not include pile burns, agricultural burns, or burning for land clearing associated with construction.

Other Data

I collected data for several control variables in order to avoid incorrect estimates from omitted variable biases: the amount of privately owned forest, pasture, and rangeland in each county, average household income of each county, average county education level, and county population density (from U.S. Census Bureau and the USDA

Economic Research Service). I also explored the potential for additional municipal layers of law governing prescribed fire use, but found no evidence of additional regulatory requirements for landowners within city limits in any of the major municipalities in the study area. Finally, I identified the existence of prescribed burn associations in each study county as such burn associations have been shown to influence the amount of land burned by private landowners (Kreuter et al. 2008, Twidwell et al. 2013c, Toledo et al. 2014).

Statistical Analysis

To examine the effect of legal variables on private landowner use of prescribed fire I used general linear models. The dependent variable in one set of regressions was percent area burned per year per county; this was expressed as area permitted to be burned on private land in a given year for a given county, divided by the total amount of privately owned forest, range, and pastureland in the county. The independent legal variables were included as sets of binary dummy variables (0=simple negligence, 1=gross negligence; 0=permit only required, 1=permit plus additional precautions required; 0=no burn ban exemptions, 1=burn ban exemptions). Strict liability was not included as a variable because none of the states included in the analysis had strict liability rules for prescribed burning. I also included average income, education, and county population density as covariates. I included an identifier for matched counties in the model as a variable with random effects, and used a compound symmetrical covariance matrix for the error term associated with the county identifier in order to account for correlated errors in dependent variables obtained from matched counties.

Another set of regressions examined number of fires per county per year. The same dummy variables, covariates, and a county identifier were used as independent variables in these regression models. In this latter set of regressions, I also included an independent variable for total area of private forest, range, and pasture land in the county to account for land area because the dependent variable was not a percent of the total land area as in the first set of regressions. I determined which legal variables had an influence on the dependent variables with likelihood ratio tests that compared the full model to a model with the dummy variable for the legal parameter of interest excluded.

Because the prescribed burn data were collected along state borders and thus each observation represents a contiguous county, I also tested for possible autocorrelation among observations with a Durban Watson analysis for each study year and for the data averaged over all study years. All data were analyzed using R version 3.0.2 (R Core Team 2013).

Results

Statutory review

Florida was the first state to undergo statutory reform in 1990 with the other southeastern states following suit between 2000 and 2012. The Florida Prescribed Burning Act of 1990 (Fla. Stat. Ann. § 590.125) required a written prescription for a burn and a CPBM to be on site during burning to obtain a permit. In 1999 the act was amended to include requirements for adequate personnel, equipment, and firebreaks and also to change the liability standard from simple to gross negligence, if all regulatory requirements are met; if regulatory requirements are not met, simple negligence applies.

Georgia followed in 2000 with an amendment to its prescribed fire statute that included a gross negligence standard (Ga. Code Ann., § 12-6-148). However, unlike Florida, the Georgia statute does not include a list of regulatory hurdles for protection under the statute. It requires the burner only to obtain a permit from the division of forestry before burning. A Georgia appellate court upheld the standard suggesting that slight diligence was all that a landowner was required to exercise in carrying out a burn given the gross negligence liability standard stated in the statue (*Morgan v. Horton* 2011). Alabama and North Carolina passed right to burn laws in 2011 (Ala. Code 1975 § 9-13-271; N.C.G.S.A. § 106-968) with requirements of a written prescription and the presence of a CPBM, but they maintained a simple negligence standard rather than adopting gross negligence. South Carolina's statute (Code 1976 § 48-34-10) passed in 2012 has the same requirements as Alabama and North Carolina and also has a simple negligence standard. All states allow county commissioners, governors, and forestry division leaders to establish open burning bans during times of dangerous fire weather, but Alabama, Georgia, and North Carolina provide exceptions for CPBMs during burn bans. Georgia provides an exception for any landowner burning for pasture and field management, silvicultural, and ecological purposes. An overview of legal variables is presented in Table 12 and the liability standard for each state is shown in Figure 18.

Table 11: Regulatory requirements and liability standards for each state

	Alabama	North Carolina	South Carolina	Florida	Georgia
Liability Standard	Simple Negligence	Simple Negligence	Simple Negligence	Gross Negligence	Gross Negligence
Burn Permit	Yes	Yes	Yes	Yes	Yes
Certified Prescribed Burn Manager	Yes	Yes	Yes	Yes	No
Written Prescription	Yes	Yes	Yes	Yes	No
Adequate Personnel and Firebreaks	No	No	Yes	Yes	No
Burn Ban Exemptions	Yes	Yes	No	No	Yes

Fig. 18 Liability standard in each state

Effects of liability and regulations

Mean percent land area burned was lower for simple negligence counties than their matched gross negligence counterparts for each year explored (2008-2013) and when averaging over the six-year study period (f=7.2, p=0.009) (Table 13, Figure 19a). However, there was no difference in land area burned between counties that require only permits and counties with additional regulations (f=2.38, p=0.13) and there was no difference between counties with burn ban exceptions for ecological burning and those without (f=0.08, p=0.78). The average annual number of fires was also lower for simple negligence counties than their matched gross negligence counterparts (f=18.74, p=0.0001) (Table 13, Figure 19b). As with acres burned, there were no differences in the number of fires between counties requiring permits and those with regulatory requirements additional to permits (f=0.82, p=0.36), and between counties with burn ban exemptions and those without (f=0.58, p=0.45). I examined the correlation coefficients associated with gross and simple negligence to determine differences in land area burned between matching counties. The statistically significant difference in land area burned between matched counties averaged over the six study years was 9.72 percent greater for gross negligence counties than simple negligence counties (f=7.2, p=0.009). This represents an additional 7919 ha of private land burned, on average, in counties with a gross negligence liability standard.

Figure 20 shows the trend in percent land area burned yearly, separated by liability standard. Gross negligence counties experienced greater percent land area burned than simple negligence counties did for the entire study period, but the difference is reduced during the last several years of the study. This is likely due to increasing drought conditions from 2009-2012 that could have constrained burning enough to diminish the effects of liability. Regardless of legal framework, prescribed burning was likely limited by low fuel accumulation in 2011, which is considered to be the peak of the 2000s drought (Arguez et al. 2012).

Table 12: Number of fires conducted model parameters and estimates.

Year	**Parameter Estimate**			**Likelihood Ratio**	
	Term	**Estimate**	**Std Error**	**f value**	**p-value**
2008	**Liability**	932.2	151.4	37.907	0.0001**
	Permit+	358.7	206.4	3.0197	0.0866
	Burn Ban exception	-268.5	151.6	3.1379	0.0808
	Education	9.895	8.724	-	
	Income	-0.0098	0.0066	-	
	Density	-0.4355	0.3422	-	
	Land area	0.0011	0.0004	-	
2009	**Liability**	764.5	151.6	25.422	0.0001**
	Permit+	367.8	206.7	3.1652	0.0795
	Burn Ban exception	-82.56	151.8	0.2959	0.5882
	Education	14.08	8.737	-	
	Income	-0.0135	0.0066	-	
	Density	-0.5437	0.3427	-	
	Land area	0.0012	0.0004	-	
2010	**Liability**	704.2	151.3	21.65	0.0001**
	Permit+	254.5	206.3	1.5218	0.2214
	Burn ban exception	-92.07	151.5	0.3693	0.5453
	Education	15.81	8.720	-	
	Income	-0.0141	0.0007	-	
	Density	-0.6043	0.3421	-	
	Land area	0.0012	0.0004	-	
2011	**Liability**	717.0	212.7	11.359	0.0012**
	Permit+	84.83	29.00	0.0856	0.7708
	Burn ban exception	-97.91	213.0	0.2114	0.6471
	Education	17.91	12.26	-	
	Income	-0.0185	0.0092	-	
	Density	-0.7869	0.4808	-	
	Land area	0.0012	0.0006	-	

Table 12 continued

Year	Parameter Estimate Term	Estimate	Std Error	Likelihood Ratio f value	p-value
2012	**Liability**	587.8	172.8	11.567	0.0011**
	Permit+	42.78	23.56	0.033	0.8564
	Burn ban exception	-107.9	173.0	0.3892	0.5347
	Education	11.06	9.957	-	
	Income	-0.0111	0.0075	-	
	Density	-0.7091	0.3906	-	
	Land area	0.0011	0.0005	-	
2013	**Liability**	488.8	161.9	9.1139	0.0035**
	Permit+	88.73	22.07	0.1616	0.6889
	Burn ban exception	-91.32	162.1	0.3175	0.5749
	Education	13.72	9.328	-	
	Income	-0.0148	0.0070	-	
	Density	-0.6419	0.3659	-	
	Land area	0.0009	0.0005	-	
All Years	**Liability**	699.1	161.5	18.738	0.0001**
	Permit+	199.6	220.2	0.8215	0.3648
	BB exception	-123.4	161.7	0.5824	0.4479
	Education	13.74	9.305	-	
	Income	-0.0136	0.0070	-	
	Density	-0.6202	0.3650	-	
	Land area	0.0011	0.0005	-	

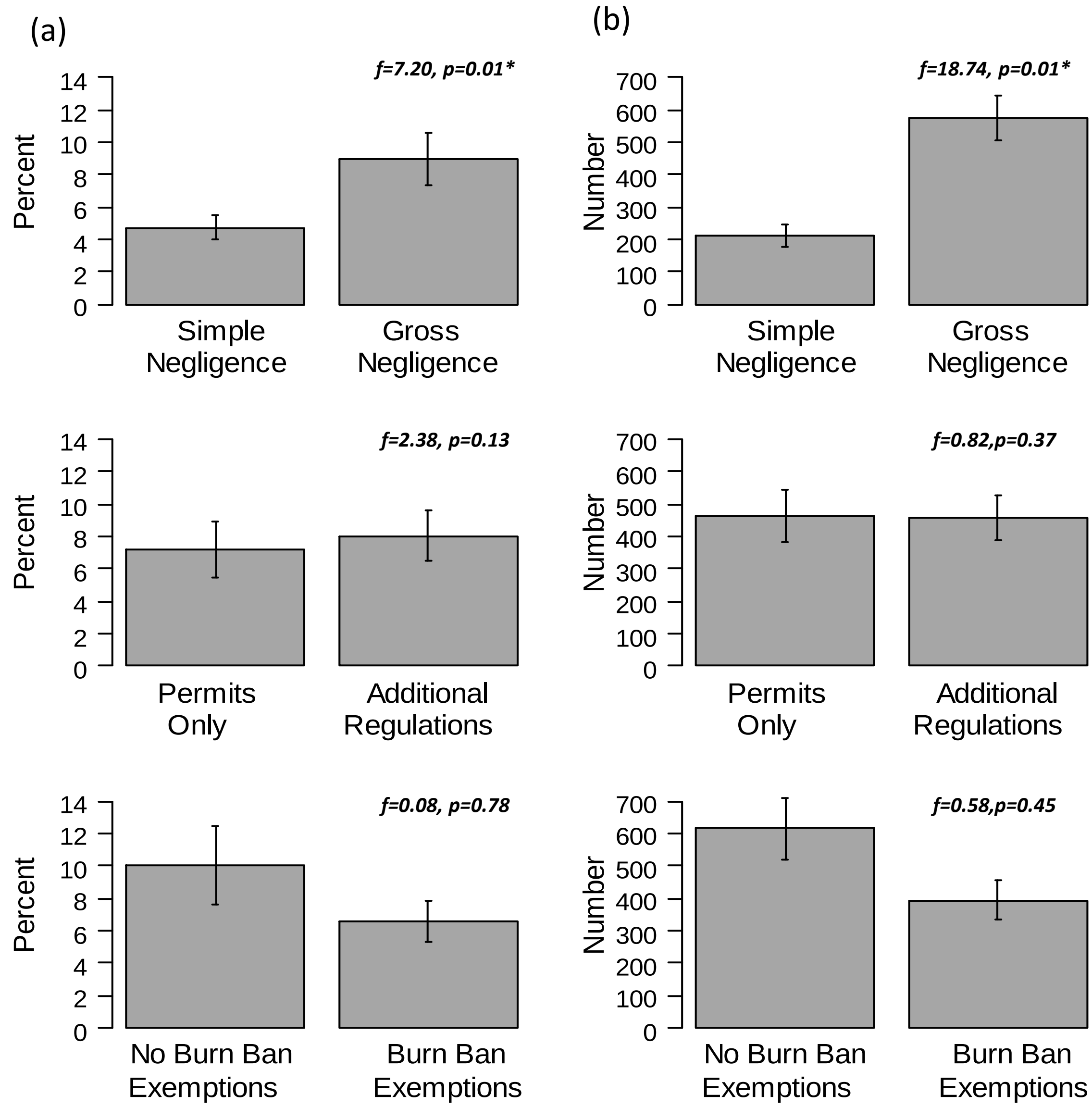


Fig. 19 Differences in (a) average annual percent land area burned and (b) average annual number of burns between counties with simple negligence and gross negligence, permit requirements only and additional requirements, and burn ban exemptions for CPBMs or land management.

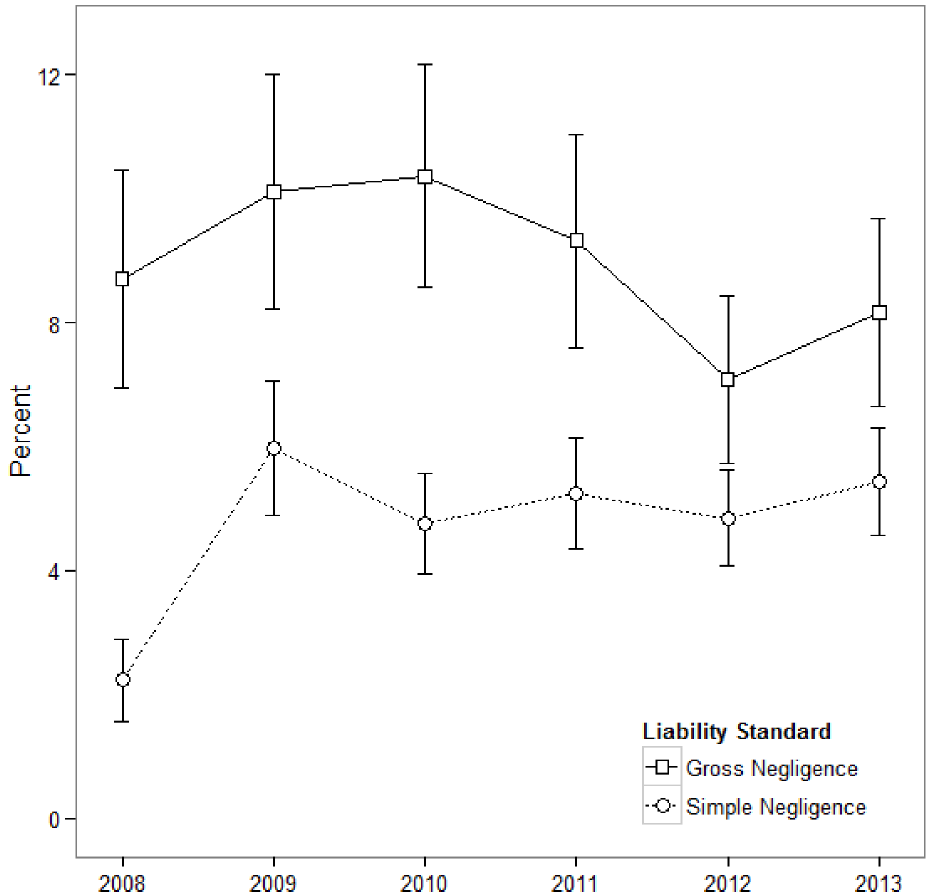


Fig. 20 Percent land area burned each year for counties with gross negligence and simple negligence status for prescribed fire.

Autocorrelation

There was no autocorrelation found among the 79 observations included in this study. This is a concern where data are collected from spatially contiguous units of observation, but a Durbin-Watson test for autocorrelation showed no correlation among percent land area burned or number of fires per county for any of the study years (Table 14).

Table 13: Durbin-Watson test for autocorrelation of observations

	Durbin-Watson	**Autocorrelation**	**p-value**
2008	1.93	0.04	0.25
2009	2.00	-0.01	0.37
2010	1.87	0.06	0.18
2011	1.99	0.01	0.36
2012	1.96	0.02	0.30
2013	1.93	0.03	0.26
Average all years	1.94	0.03	0.27

Discussion

The Prescribed Fire Acts passed in southeastern United States have focused on the importance of fire as an historical part of southern forests and grasslands. The Acts seek to promote the use of fire because of the broader benefits it provides to the general public, such as reducing wildfire risk and maintaining ecosystem health, as well as ecological and economic benefits, such as inexpensive brush control and grassland revitalization, which accrue directly to the burner. Prescribed fire stimulates essential ecosystem services, such as nutrient cycling (Noss et al. 2006), improved forage quality (Collins and Wallace 1990, Knapp et al. 2008) and disease and pest control (DiTomaso et al. 2006). Through the selective application of fire, land managers can increase spatial and temporal heterogeneity of plant and soil microbial communities (Turner et al. 1994, Chang 1996, Fuhlendorf et al. 2012, McGranahan et al. 2012), diversifying natural areas and improving wildlife habitat. In addition, providing a low cost land management option allows many land owners and managers to maintain forests and rangelands in ecologically functional states rather than converting them to land uses providing higher economic yield such as development or agriculture (McCullers 2013). Maintaining land as ecologically functional forests and grasslands promotes biodiversity, nutrient cycling, carbon storage, water filtration, and other critical ecosystem services that benefit society at large.

The most broadly recognized social benefit of prescribed fire is its use in reducing hazardous fuels. Fires in Yosemite National Park were most limited in spatial extent and severity where a "let-it-burn" policy had been adopted for naturally occurring wildfires (van Wagtendonk et al. 2012). Larger, more severe fires occurred in adjacent areas with a long history of fire suppression. Fire fuel models corroborate this outcome, with large reductions in fire intensity and average fire size in models that incorporate hazardous fuel reduction with prescribed fire (Fernandes and Botelho 2003). Similarly, prescribed burning lowered the incidence and extent of wildfires in Australian eucalypt forest (Boer et al. 2009). This reduction in hazardous fuels can lower the number and intensity of subsequent wildfires in the area, facilitating suppression efforts and limiting structural losses (Fernandes and Botelho 2003). There are also some costs inherent in using prescribed fire. Prescribed fire causes smoke, which can present safety and health risks (Hardy et al. 2001). It can lead to substantial reduction in visibility and a loss of life and property if not properly controlled, as was the case in Florida in January 2008 when 70 vehicles collided due to reduced visibility from fog mixed with smoke from an escaped prescribed fire (McCullers 2013). Smoke also causes respiratory health issues in communities near large fires (Bowman and Johnston 2005). Such risks can, however, be reduced by timing prescribed fire to limit the amount of smoke reaching nearby communities and by taking precautions to reduce accidents caused by smoke on roadways (Hardy et al. 2001). Another potential option for dealing with risks caused by smoke is to develop legislation which separates smoke-caused damages from fire-caused damages and mandates different liability standards for each. A statute can require a finding of gross negligence for recovery from certified burners if damages are the result of fires and simple negligence in cases of smoke-related injury or property damage. In addition to the risks posed by the generation of smoke, the potential for prescribed fires to escape and cause losses of lives and property is the largest cost associated with its use. Ninety-nine percent of prescribed fires are successfully restricted to the intended area of burning, but the rare escapes can be catastrophic (Ryan et al. 2013). For instance, the Lower North Fork Fire in Colorado in 2012 was started by a spot fire from a nearby prescribed burn and resulted in three fatalities and $11.3 million in property damages (The Colorado Legislative Council 2012).

Statutorily prescribed legal liability standards and regulations for prescribed burners seek to find an efficient and effective balance between the societal costs and benefits of prescribed burning. Gross negligence standards shift some costs of burning associated with escapes from the burner to the adjacent property owners. This reduced cost of burning provides an incentive encouraging prescribed burning on private land (Yoder et al. 2004). Our study reports an additional 9 percent (± 4 percent) of total hectares of forest, pasture, and rangelands were burned in counties with gross negligence liability standards in 2013. When applied across the counties included in this study, a switch to gross negligence liability for the simple negligence states would result in an average additional 7388 ha burned per county per year. Gross negligence also functions as an incentive to follow statutory regulatory requirements and receive prescribed burn training. This lowers the risk of escape and the attendant costs for both burners and adjacent property owners as those applying fire are better trained to properly conduct safe and effective prescribed burns. In addition, the lower liability standard can incentivize the creation of defensible space and fire-wise construction because adjacent landowners are exposed to a larger portion of the costs attendant to prescribed burning escapes than the burners under gross negligence liability standards.

Yoder (2008) analyzed the relationship of liability standards and regulations to the occurrence and severity of escaped prescribed fire in the United States. He defined severity as a measure of the cost of suppression plus an estimate of damage costs resulting from the fire. He found that gross negligence states had 62 percent more escapes than simple negligence states, but damage and suppression costs were not higher. Yoder's analysis does not include data on the total number or acreage of prescribed fires. Therefore, the higher number of escaped fires could potentially be the result of higher numbers of prescribed fires conducted in gross negligence states. Regardless, the finding of no difference in damage or suppression costs suggests that gross negligence standards are not leading to vastly greater losses than simple negligence standards.

In the absence of gross negligence standards, prescribed burn associations might provide a non-legislative mechanism for limiting liability associated with prescribed fire use by private landowners. These associations are cooperatives of landowners with a common goal of using fire to manage private lands and they are established to share the costs of prescribed burning (Toledo et al. 2012, Twidwell et al. 2013c). They provide shared labor and equipment on burn days, serve as a conduit for established knowledge related to prescribed burning, provide safety training for new members, and can potentially spread the costs of liability insurance among members (Toledo et al 2014). In Oklahoma and Texas, prescribed burn associations have even driven legislation that allows for burning by certified prescribed burn managers during burn bans to meet land management objectives (Twidwell et al. 2013c). However, the effectiveness of burn cooperatives to reduce liability concerns associated with prescribed burning is limited by risk-driven legislative and regulatory requirements as burn associations are still subject to the same level of liability as individual burners (Twidwell et al. 2013c).

In 2012, the Tennessee House of Representatives voted almost unanimously (with one vote opposing) for the passage of the Tennessee Prescribed Burning Act. The act had been drafted by the Tennessee Wildlife Federation and the Tennessee Prescribed Fire Council, with the hopes of developing a Certified Prescribed Burn Manager training program. The bill offered limited liability in the form of a gross negligence liability standard to CPBMs as an incentive to complete the training and use additional statutorily circumscribed precautions, such as developing a burn prescription and having a CPBM on site for the duration of the burn. It was drafted following the example of the Right to Burn Acts in other Southeastern states in order to promote safe use of prescribed fire to reduce hazardous fuels and increase ecosystem health. The bill faced a legislative battle in the Senate, however. It was attacked on grounds that gross negligence would leave burners unaccountable for damages. The debate was fueled by front-page news of a catastrophic escaped prescribed fire in Colorado just days before the Senate hearing on the Prescribed Fire Act. Supporters of the act failed to effectively counter with the importance of the act for increasing safety in prescribed burning through incentivizing training programs and in the end opted to settle for a simple negligence standard in order to move forward with the CPBM training program and have language regarding the value of prescribed fire for the ecosystems of Tennessee in the state statutes. Many supporters of the original bill felt the less stringent, gross negligence liability standard was essential to achieving the stated purpose of the statute – to promote the use of prescribed burning for range and forest health, fuel reduction, and perpetuation of Tennessee's plant and animal populations (T.C.A. § 68-102-146). Tennessee is not the only state struggling to develop appropriate statutes for reducing constraints on landowners who desire to include fire in their suite of management tools. Discussions of optimal liability and regulatory schemes for prescribed burning should be informed with data regarding the effects of these legal variables on land mangers' decisions.

Our results show that private landowners are more likely to use prescribed fire for managing their properties and burn a greater proportion of private land in counties where their state has adopted gross negligence liability standards compared with landowners in counties who are subjected to state-mandated simple negligence legal standards. Interestingly, regulatory requirements, such as adequate firebreaks, personnel, and equipment, written burn plans, and CPBMs on site do not decrease the amount of burning on private land. In fact, these types of regulations, in conjunction with lower liability, will make prescribed fire more available to landowners and managers while providing some safety assurances for neighbors. Taken together with Yoder's (2008) finding of no additional damage or increased suppression costs in states with gross negligence standards, lawmakers struggling to determine the optimal legal framework for promoting burning should consider the benefits of a lower legal liability standard for those undertaking to manage fuel, forests, and rangelands with fire. Given the importance of fire to the maintenance of natural systems worldwide and our demonstration of the effects the legal landscape has on private land prescribed burning, liability-related disincentives to prescribed fire use will likely have a tremendous influence on the future structure and functioning of ecosystems (Twidwell et al. 2013c). Ecologists and land managers also need to be aware that policy regulations and liability concerns may create legal barriers that inhibit the use of prescribed fire. Such recognition will allow them to better engage and educate both the public and policy-makers regarding the essential role fire plays in these ecosystems. Opportunities to foster communication between related stakeholder groups should be promoted whenever possible. Indeed, a more comprehensive and thorough understanding of these legal-ecological feedbacks is essential to increase the availability of effective ecosystem management strategies in fire-prone ecosystems worldwide and to provide solutions to management issues that address both social concerns and ecological perspectives.

CHAPTER VI

SUMMARY AND CONCLUSIONS

Restoring and managing brush encroached rangelands, especially given the potential for drastic changes in disturbance regimes resulting from climate change and human influence in the upcoming decades will require a precise understanding of vegetation response to management and disturbance such as drought. Researchers must identify unique species responses but are confronted with the challenge of disentangling the effects of multiple ecological processes operating across a variety of spatial and temporal scales (Archer et al. 1994, Hughes et al. 2006, Ratajczak et al. 2012). The research outlined in this body of work improves our understanding of management- and disturbance-induced vegetation change at multiple spatial resolutions. It provides detailed information regarding biophysical controls on encroaching brush species and perennial grass population dynamics, which will inform our understanding of the scale and direction of grassland responses to agents of disturbance and management. This information can aid in the development of predictive models that incorporate multiple scales, more accurately reflecting vegetation community responses to both intervention and altered disturbance regimes (Peterson and Lipcius 2003, Harris et al. 2006). Accurate predictive models will allow us to prepare for and adapt to changing climatic conditions and target intervention efforts (Baker 1995). Woody brush management and restoration of encroached rangelands is extremely cost-prohibitive, especially when employing mechanical and chemical interventions (Kreuter et al. 2005, Van Liew et al.

2012, Twidwell et al. 2013c). Currently, the Natural Resource Conservation Agency (NRCS) spends two-thirds of its budget annually to manage brush on only 3% of the total land area under its jurisdiction (Twidwell et al. 2013a). Given the high cost of restoration, both prioritizing areas to target interventions, and increasing knowledge regarding the potential effectiveness of interventions from both a brush cover and a grass community perspective are invaluable to restoring brush encroached rangelands.

The suite of experiments outlined in this dissertation provides information which can begin to fill those gaps for brush management on the South Texas Plains. In addition, I establish a framework for developing restoration targets in other systems and begin to elucidate the potential for natural disturbances to interact with management interventions. However, woody brush encroachment on rangelands is a complex social-ecological issue (Allen et al. 2011, Twidwell et al. 2013c, Anadón et al. 2014, Toledo et al. 2014). Therefore, understanding the ecology of responses to different interventions at scales relevant to land managers is only the first step. Understanding social barriers to effective ecosystem management that prevent adoption of ecologically effective interventions is equally important. One cost-effective method for achieving brush reduction and control in encroached rangelands is through the use of prescribed fire (Van Liew et al. 2012). However, even managers who are aware of the benefits and desirous of inexpensive means for achieving invasive brush management objectives have avoided using prescribed fire largely due to concerns over potential legal liability. Our legal research provides an assessment for determining the effectiveness of current laws and regulations relative to prescribed burning for the restoration and conservation of fire-dependent ecosystems. An understanding of landowner response to particular regulations and liability standards can be used to formulate prescribed burning legislation that will promote the safe and effective use of prescribed fire by private land managers.

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APPENDIX A

Mortality (individuals/year/treatment) for each perennial grass species present in subplots averaged across replicates for each treatment.

Code	Species	Functional Group*	Scientific name	Mortality: Control	SE	Cut Herbicide	SE	Mechanical	SE	Fire	SE
AROL	oldfield threeawn	A	Aristida oligantha								
ARPU	purple threeawn	P	Aristida purpurea	0.1038	0.0253	0.1452	0.0141	0.2304	0.0313	0.1712	0.2703
BOAR	needle grama	A	Bouteloua aristidoides								
BOHI	hairy grama	P	Bouteloua hirsuta	0.0479	0.0111	0.1565	0.0065	0.1745	0.0237	0.1176	0.3726
BOIS	kr bluestem	PI	Bothriochloa ischaemum var. songarica	0.0028	0.0016	0	0	0	0	0	0
BOLA	silver bluestem	P	Bothriochloa laguroides	0.0194	0.0112	0.0139	0.0026	0	0	0.0222	0.0813
BORI	tx grama	P	Bouteloua rigidiseta var. rigidiseta	0.0029	0.0017	0.0028	0.0004	0.0205	0.0028	0.0019	0.0096
CESP	sandbur	P	Cenchrus spinifex	0.0361	0.0121	0.0611	0.0049	0.5454	0.0742	0.2213	0.5018
CHCU	hooded windmill	P	Chloris cucullata	0.0361	0.0087	0.2121	0.0049	0.1619	0.022	0.1912	0.4858
DIAN	kleburgs bluestem	PI	Dichanthium Annulatum	0	0	0.0083	0.0001	0	0	0	0
DICO	fall witchgrass	P	Digitaria cognata	0	0	0.0111	0.0002	0.1743	0.0237	0.0222	0.0902
DIPA	tx cottontop	P	Digitaria patens	0	0	0.0139	0.0001	0.0057	0.0008	0.0038	0.0137
ERBA	mediterranean lovegrass	AI	Eragrostis barrelieri								
ERIN	plains lovegrass	P	Eragrostis intermedia	0.0083	0.0035	0.014	0.0011	0.1432	0.0195	0.0627	0.1707
ERLU	mourning lovegrass	P	Eragrostis lugens	0	0	0	0	0.0252	0.0034	0.0065	0.0256
ERSEC	red lovegrass	P	Eragrostis secundiflora	0.0028	0.0016	0.026	0.0004	0.0099	0.0013	0.0174	0.0715
ERSER	tx cupgrass	P	Eriochloa sericea	0	0	0.0083	0.0001	0	0	0.0056	0.0289
ERSES	tumble lovegrass	P	Eragrostis sessilispica	0.0611	0.0166	0.0535	0.0083	0.1011	0.0138	0.0807	0.1719
HECO	tanglehead	P	Heteropogon contortus	0.1167	0.0339	0.0896	0.0159	0.4427	0.0602	0.1532	0.3110
HIBE	curly mesquite	PR	Hilaria belangeri	0.5583	0.1427	0.841	0.076	1.3307	0.1811	1.0560	1.8200

NALE	tx wintergrass	P	Nassella leucotricha	0.0139	0.0056	0.0236	0.0019	0	0	0.0111	0.0349
PABI	pink papasgrass	P	Pappophorum bicolor	0.2795	0.0452	0.3541	0.038	0.5848	0.0796	0.4326	0.4765
PACA	canary grass	AC	Phalaris caroliniana								
PACO	kleingrass	P	Panicum coloratum	0.0028	0.0016	0	0	0.0439	0.006	0.0311	0.1519
PADI	dallis grass	PI	Paspalum dilatatum	0.0194	0.0097	0.0028	0.0026	0.0058	0.0008	0.0131	0.0582
PAHA	halls panic	P	Panicum hallii var. hallii	0.0479	0.0143	0.2183	0.0065	0.3987	0.0543	0.2263	0.4502
PANO	bahia grass	PR	Paspalum notatum	0	0	0.0056	0.0012	0	0	0	0
PAOB	vine mesquite	PR	Panicum obtusum	0	0	0.0444	0.0001	0.0069	0.0009	0	0
PECI	buffel grass	PI	Pennisetum ciliare	0	0	0.0174	0.0007	0.0204	0.0028	0.0074	0.0385
PENE	bentspike bristlegrass	PI	Pennisetum nervosum	0	0	0	0	0.0088	0.0012	0	0
PLMU	tobosa	PR	Pleuraphis mutica	0	0	0.0028	0.0001	0	0	0.0019	0.0096
SELE	plains bristlegrass	P	Setaria leucopila	0.2056	0.0345	0.2504	0.028	0.6709	0.0913	0.4171	0.4896
SERE	reverchon bristlegrass	P	Setaria reverchonii	0	0	0	0	0.0432	0.0059	0	0
SESC	sw bristlegrass	P	Setaria scheelei	0.0222	0.0085	0.0056	0.003	0.061975406	0.0084	0.0185	0.0574
TRAL	white tridens	P	Tridens albescens	0	0	0	0	0.0083	0.0011	0.0056	0.0289
URCI	fringed signalgrass	PR	Urochloa ciliatissima	1.0112	0.1972	0.4912	0.1376	1.3889	0.189	1.2158	1.7444

*P=perennial warm season tufted native bunchgrass, PI=perennial warm season tufted introduced bunchgrass, A=annual, AC=annual cool season,
PR=perennial warm season native rhizomatous