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Effects of Mode of Cedar (Juniperus ashei) Removal on Grassland Species Composition in the Texas Hill Country

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Introduction

Earth's species are in a constant state of fluctuation. In response to their biotic and abiotic environments species adapt and alter their range and habitats. Globally, arid grassland ecosystems are being invaded by woody tree species (Archer et al. 1988). In the past 200 years, such an invasion in Texas and the American Southwest has altered the landscape from dominant C_4 perennial bunchgrass vegetation to closed canopy forests of mainly *Juniperus* sp. and *Prosopis* spp. (Van Auken 2000; Smeins 1980; Smeins and Fuhlendorf 1997). Invasion is defined as aggressive behavior of species expanding into new habitat. In this case, the woody species encroaching into grassland habitat are all native, but are expanding their ranges as a result of human management practices which simultaneously decrease periodic wildfires and herbaceous cover. Woody encroachment is of great concern to ranchers due to a decreasing area for pastures. In addition many landowners value plant diversity and the accompanying services stable grasslands provide for native wildlife.

Unfortunately when closed canopy forests were removed in an attempt to promote grassland species another invasion occurred. King Ranch Bluestem (a.k.a. KR bluestem, *Bothriochloa ischaemum* (L.) Keng var. *songarica*), is a highly invasive, non-native, C_4, perennial grass. The species is found throughout Texas semi-arid rangelands due to its introduction as a rangeland revitalization tool. KR bluestem and its invasion is under investigation by many researchers because it threatens ecosystem stability (Duffy 2009; Gamfeldt et al. 2008; Schmidt et al. 2008;). A functioning ecosystem is thought to be maintained by biodiversity because with more species there is higher redundancy in ecosystem processes (Gamfeldt et al. 2008).

My study is designed to assess change in herbaceous communities following *Juniperus ashei* clearing on the Edwards Plateau of Central Texas. I quantify differences in herbaceous cover and diversity in grassland ecosystems as a function of modes of *J. ashei* removal (chainsaw, bulldozer, or Cedar Eater), and post-removal land management (seeding with native grass mixtures and lopping of *J. ashei* seedlings.). The study's ultimate goal is to provide Central Texas landowners with practical information for how to improve grassland habitat quality for wildlife while preventing non-native species invasions.

Historical Vegetation Distributions

Vegetation of Central Texas has alternated between dominant woodland, savannas, and scrub grasslands since 33,500 years before present (Archer 1990; Bryant and Scafer 1977; Perkins 1977). We know that the contemporary flora of Central Texas reached its current distribution in the last 3,000 years, which is concurrent with a trend of increasing aridity in the region (Bryant and Shafer 1977; Gould 1975). Others claim that aridity, and thus woody and herbaceous vegetation distributions, has been fluctuating, rather than steadily increasing, during the Holocene (Archer 1994; Van Devender 1987). The rapid changes in woody density observed in the past 150-200 years are concerning. We therefore need more studies to assess the role of human activity on woody species encroachment. It is possible that the observed vegetation change is part of a natural cycle, however if human forces are causing the change we should do our part to reverse and eliminate the effects.

The beginnings of vegetative change came in 1541 with the western expansion of European settlers and their cattle (Perkins 1977). In 1541, Francisco Vasquez de Coronado drove the first recorded herds across Texas. Five hundred cattle, accompanied by 1000 horses, and 5,000 rams and ewes, traversed a triangle ranging from the Llano Estacado, San Angelo, and the Red River east of Childress. Alonso de Leon later constructed Texas' first missions in response to French exploration in East Texas. Following construction in 1960 Leon continued to import cattle herds to the missions and modern longhorn lineages arose from these Christian hubs in East Texas, Waco, and San Antonio. Cattle continued to thrive in, and be imported to, the region throughout the 1700's but were only semi-domesticated and tended to range far afield. Wide open grasslands primed for cattle consumption, demand for beef, cross country trade, and Texas' wealth of cheap land found Texas with more cattle than any other state in the Union (Perkins 1977). Looking at history, the number of non-native organisms was itself a concerning trend, but additional changes to ranching practices compounded the preexisting issues.

The invention of barbed wire and increasing human population altered the density and movement of cattle herds and may have contributed to changes in vegetation described during the late 1800’s. Prior to the invention of barbed wire, wooden fencing was used to keeping livestock out of agricultural fields (Netz 2009). Cattle and their cowboys were thus semi-nomadic, following water and fresh food. Shipping wood from the east was an expensive and precious endeavor. Barbed wire thus filled an economic need for cheap and durable fencing material. Fences now required less wood, were less susceptible to prairie fires, and livestock did not break them down (Netz 2009).

Following the financial woes of the civil war, immigration of humans and cattle to the open plains of Texas was staggering. After 1876, two hundred thousand cattle per year were moved into Texas for new ranches (Netz 2009). Such fierce competition over ranching resources necessitated widespread fencing of cattle (rather than allowing cattle to range relatively freely). The resulting effects of enclosing large herds are difficult to separate from concurrent reductions in wildfires (Archer 1994; Smeins 1980; Taylor et al. 1993); however, the change from semi-nomadic to stationary concentrated herds resulted in increases in grazing intensity (Walker 1995; Warren et al. 1986). With increased ungulate concentrations, came reductions in herbaceous fitness, herbaceous coverage, changes to decomposer communities, increased bare-ground cover, incident radiation, soil runoff, and nutrient leaching (Mikola et al. 2009; Sayer 2005; Walker 1995; Warren et al. 1986). These dramatic environmental changes have convinced scientists that the current shift in woody plant density and cover is not a natural phenomenon, but is instead due to indirect effects of anthropogenic activity.

Mechanisms of Woody Species Encroachment

Given an understanding of the environmental changes occurring during the onset of woody species encroachment, we can turn our attention to the specific mechanisms of encroachment. This study focuses on expansion of *J. ashei* (a.k.a. cedar) on the Edwards Plateau in Central Texas; however, substantial research has been done on the encroachment into rangeland ecosystems by *Prosopis* spp. (mesquite). While the mechanisms driving encroachment of these two species into semi-arid grassland ecosystems are not identical, we include discussion of mesquite invasion here because it can inform us about how and why *J. ashei* invades and how it affects herbaceous plant species compositions.

Historically, *J. ashei* has dominated steep and rocky slopes where the species can effectively utilize deep groundwater resources and disperse its heavy round seeds via gravity (Alofs and Fowler 2013; Idso 1992; Foster 1917; Smeins and Fuhlendorf 1997; Wink and Wright

1973). Reduced competition of herbaceous species on highly sloped and rocky regions favors *J. ashei* establishment where the species has a natural ability to cope with highly eroded soils (Alofs and Fowler 2013; Smeins and Fuhlendorf 1997; Foster 1917). Part of this ability stems from the species' deep taproots which not only serve to anchor individuals but also allows access to deep groundwater trapped in the karst limestone bedrock (McCole and Stern 2007; O'Connor et al. 2013). Slopes also allow immediate seed dispersal for *J. ashei's* heavy round seeds which role downhill via gravity (Smeins and Fuhlendorf 1997). Additionally, seed predation by whitetail deer, and many birds during high drought years can provide a second dispersal mechanism into a wider variety of habitats (Smeins and Fuhlendorf 1997). Seeds can reach open grassland by either of these two dispersal mechanisms, but may or may not be able to establish depending on the health of the recipient grassland community.

In locations of healthy grassland, *J. ashei* establishment and growth is limited by herbaceous species competition, and *J. ashei* maturation is minimized by frequent wildfires. Owens and Schliesing (1995) investigated variation in seed germinability of *J. ashei* between different types of soil and vegetation conditions. They found that germination of seeds in open grasslands is minimal, with no individuals surviving for longer than two years. Jackson and Van Auken et al. (2004) come to a similar conclusion that individuals of *J. ashei* do not establish on ungrazed prairie dominated by herbaceous species. The majority of plant competition in grassland habitats occurs belowground, with species competing over very limited reserves of water and nutrients in the top few inches of soil (Van Auken 2000). With shallow and fibrous roots that effectively absorb surface moisture, grasses are strong competitors in these environments. The same resource niche is occupied by *J. ashei* seedlings which have yet to develop large taproots, so they are in direct competition for resources (especially water) when *J. ashei* is establishing in grasslands (Knapp and Soule 1998). The competitive pressure of herbaceous competition thus limits *J. ashei* germination, establishment, and growth (Fowler 2002; Van Auken et al. 2004).

Should *J. ashei* species survive past infancy (which is more frequent along a mature *J. ashei's* canopy dripline), wildfires hotter than 100°C are fatal to seedlings less than 3cm in diameter (about 4-5 years old) (Yancey 1982). Once older than 4 or 5 years, fires do not spread as effectively beneath the *J. ashei* canopy and do not reach an individual's sensitive central stem (Knapp and Soule 1998, Smeins and Fuhlendorf 1997). Thus, to reduce the occurrence of large diameter *J. ashei* individuals, fires more frequent than every 4-5 years seem to be necessary (Yancey 1982). In cases where *J. ashei* are beneath the canopy of other trees such as Texas Live Oak (*Quercus fusiformis*) or Post Oak (*Quercus stellata*), *J. ashei* are able to survive higher temperature fires (up to 125°C for trees with a 3.8cm diameter; Yancey 1982); however, the intensity of fires is dependent on the biomass and health of the grassland. Fire may not kill *J. ashei* seedlings in degraded grasslands.

Woody species encroachment theories most frequently claim that reduced fine fuel loads (grass cover) lead to reduced fire intensity and fire coverage. This allows long-lived and hardy woody species to establish and grow (Brown and Archer 1999). *J. ashei* encroachment may occur when grassland fitness is reduced by ungulate grazing, and a concomitant reduction in wildfire frequency or intensity occurs. Cattle preferentially graze herbaceous cover as opposed to shrubby or woody plants (it is far less palatable), and thus influence the distribution and biomass of species (Fowler and Simmons 2009). The resulting effects on the herbaceous community include a reduction in grass cover and fitness, reduced intensity and coverage of wildfires, higher irradiance at the soil level, decreased soil-water penetrance and increased soil moisture more suitable to *J. ashei* growth (Jackson and Van Auken 1997; Schmalz et al. 2013; Van Auken 2000; Van Auken et al. 2004). Perhaps the most important of this list is the impact of wildfire reduction on *J. ashei* seedlings. Efforts to reduce *J. ashei* expansion should begin with limiting *J. ashei* seedling germination and survival because the seedling stage is the plant's most vulnerable life phase (Archer 1994; Brown and Archer 1999). Fires hot enough to kill *J. ashei* seedlings in one study had herbaceous biomasses ranging from 1542-1792 kg/ha (Yancey 1982). While the fuel load greatly impacts the heat of fire produced, fuel moisture is also partially responsible for the heat and uniformity of the fire (Yancey 1982). As *J. ashei* mature, they further limit the competitive ability of herbaceous species by increasing shade and producing physical litter barriers (Ansley et al. 2006; McCole and Stern 2007; Van Auken et al. 2004). Finally, new *J. ashei* seedlings are facilitated by the mature parent tree's microhabitat. Thus each mature individual becomes a hub of woody invasion (Jackson and Van Auken 1997; Van Auken and Jackson, 2004). Overgrazing creates a positive feedback loop in which *J. ashei* become more competitive as herbaceous cover is reduced, and maturation further facilitates *J. ashei* encroachment while limiting the fitness of the remaining grassland. We see this mechanism is specific to *J. ashei* growth based on the species' susceptibility to fire and reduced fitness via herbaceous competition. It is unclear if this mechanism is applicable to other invasive woody plants, such as *Prosopis* species.

Site history and experimental protocol are essential concerns when investigating *Prosopis* spp. encroachment. For example, Brown and Archer (1989) observed *Prosopis* *glandulosa* var. *glandulosa* under different levels of manipulated defoliation. None, moderate, and heavy defoliation treatments were all tested but were also replicated in pastures that had been protected from grazing, as well as sites that had been grazed long-term. They found that germination following moderate and heavy defoliation was not significantly different from one another, but that the difference between no defoliation and moderate/heavy treatments was. Others have found that herbaceous defoliation only reduces *P. glandulosa* germinability a small degree (Schmidt and Stubbendieck 1993; Brown and Archer 1999); however, once seedlings had established, their survival, development, and physiological activity were comparable across all treatments. Additionally, germination comparisons between previously protected sites and recently grazed sites showed little change in seedling survival, development, or physiological activity. Thus, the long-term grazing was not a facilitator of woody encroachment in this study. Other studies have contradictory findings, where *Prosopis velutina* density increased as a result of cattle exclusion (Browning and Archer 2011). The importance of records documenting past grazing, fire, climate data, and locations of woody clearing is essential to these studies (Archer 1994; Browning and Archer 2011; Weltzin et al. 1997). One must also be cognizant of the different species being studied as specific differences in species can have significant effects.

While occasionally cited as a mechanism of woody plant encroachment, increased concentrations of CO_2 cannot explain the wide variability of woody encroachment patterns worldwide. This hypothesis hinges on the extra energy invested into the production of malate in order to prevent photorespiration (an extremely energy expensive process in which oxygen rather than CO_2 is fixed by RUBISCO). In theory, as CO_2 concentrations increase, photorespiration should decrease. Thus, the advantages to C_4 plants are minimized when CO_2

concentrations reach a certain threshold (Idso and Quinn 1983; Idso 1992; Smeins and Fuhlendorf 1997); however, an increase could confer advantage to woody C3 species and allow them to encroach into grassland habitat. On the other hand, such an advantage is also expected to benefit C3 grasses, and this trend is not observed (Archer et al. 1995). Additionally, C3 woody plants continue to invade C3 grasslands on a global scale (Archer et al. 1995). Finally, sites with little variation in climate still show high variability of woody species establishment based on anthropogenic forces (Archer et al. 1995). Thus, increased atmospheric CO2 alone is an insufficient explanation to support claims of global woody encroachment.

Kangaroo rats, prairie dogs, ants, and soil microbes can also be major components of ecosystem and species composition changes. For example, the extirpation of Kangaroo rats (*Dipodomys* sp.) in pasture plots led to the increases in grass height, litter accumulation, and a decrease in bare ground cover (Archer 1994; Brown and Heske 1990). While this study shows how *Dipodomys* sp. decreased herbaceous fitness, *Cynomys ludovicianus* (black-tailed prairie dogs) colonies in South Texas Savannahs maintained grassland ecosystems while limiting encroachment. (Weltzin et al 1997).*C. ludovicianus* are granivores, and frequently eat and destroy *Prosopis* spp. seeds (Weltzin et al 1997). *P. glandulosa* seed pod predation around prairie dog colonies increased as a direct result of prairie dog extirpation. Colonies also influenced soil hydrology due to burrowing, and increased *P. glandulosa* seed herbivory via increased ant populations in burrows (Weltzin et al. 1997). Seen as a rangeland pest, 98% of colonies in the U.S. were exterminated at the beginning of the 1900's. Such widespread extirpation had far reaching implications on vegetation and may be responsible in part for the observed increases in woody species density (Weltzin et al. 1997).

Fungal and bacterial associations in herbaceous leaves are also important factors of species fitness and interaction. Endophytes can provide a competitive advantage by both increasing the host's fitness and inhibiting or facilitating the fitness of other individuals (Perkins and Nowak 2013; Wearn et al. 2012). Commonly associated with herbaceous species endophytes can impact ecosystem dynamics. Rudgers et al (2007) explain how endophytes make antiherbivore compounds that protect grasses from predation. Thus endophytes indirectly increase woody species competition, which slows woody encroachment (Rudgers et al. 2007). While this has not been studied in *J. ashei* or *Prosopis* spp. woodlands, the relationship ought be considered when analyzing the mechanisms of woody species encroachment.

Rare weather and disturbance events can have pervasive impacts on ecosystems in arid climates (Browning and Archer 2011); however, this trend is difficult to predict because wet or dry weather may have different year to year effects. Wet years usually allow herbaceous species to compete effectively with *J. ashei* and other woody species, while drought allows woody species to sometimes gain ground over grasslands (despite *J. ashei* mortality in extreme drought) (Ansley et al. 2006; Smeins and Fuhlendorf 1997). Oppositely, wet years and seasonally variable rains can increase opportunities for woody species and forests. In a study of Western Junipers (*J. occidentalis*) in Oregon, the traditional explanations for the observed woody encroachment were failing to explain the species' 57% cover increase over twenty three years (Knapp and Soule 1998). Researchers propose alternative hypotheses. Increases in *J. occidentalis* seed rain following wet years in the early 1900's could explain the site's current density. Other studies have quantified the effect of similar climate anomalies by examining the age of standing woody species. Age classes are compared to historical climate records and researchers investigate anomalies in the historical record that could explain current age distributions (Browning and Archer 2011). Once again we recognize the importance of keeping proper site history and yearly records recording site history and climate or disturbance events.

Herbaceous Invasion: A Problem Following Woody Encroachment.

It has been established that woody encroachment itself is a threat to grassland biodiversity. Reduction and fragmentation of herbaceous habitat can compromise grassland ecosystem stability via reductions in biomass and biodiversity. Other problems arise as landowners remove *J. ashei.* Clearing mature *J. ashei* stands appears to result in increases in non-native invasive grasses but the mechanism of species invasion and is a subject of much controversy under constant debate. My study is designed to investigate if the practices of *J. ashei* removal and post removal land management act as determinants of grassland community species composition. I measure factors that may allow insight into how landowners can minimize opportunities for invasion while promoting the success of native species. My goal is to provide Central Texas landowners with useful advice on how to improve their grassland quality while preventing non-native species invasions.

Invasion by non-native species can occur in a number of ways including reductions in biomass, decreased species richness and diversity, and losses of key competitive species (Alofs and Fowler 2013). Clearing *J. ashei* could create conditions ideal for invasion. My study examines how the *J. ashei* understory compares to open grassland in terms of vegetative and herbaceous cover. Invasibility describes the ease with which new species are able to establish

(Levine and D'Antonio 1999). If the *J. ashei* understory is relatively depauperate, then clearing may create opportunities for invasion once canopy shade is removed (Alofs and Fowler 2013; Levine and D'Antonio 1999; Levine et al. 2002). Managers seek the most effective way to minimize these risks. Knowledge as to the drivers of invasion will help to inform these management decisions. Therefore one should know if increases in biomass, diversity, or certain species alter invasibility. For example, one might expect that planting herbaceous seedlings immediately following clearing might decrease invasibility. While not specifically testing the practice of bio-control, my study does estimate the effects of land management on cover and biodiversity. Low cover or diversity values resulting from management practices could indicate a system's susceptibility to invasion (Alofs and Fowler 2013; Levine and D'Antonio 1999; Levine 2002; Lyons 2006).

King Ranch bluestem (a.k.a. Yellow Bluestem, *Bothriochloa ischaemum*), is a highly invasive non-native perennial C_4 bunchgrass that threatens stability of grassland ecosystems across south Texas. Used by ranchers to improve degraded rangelands and by the Texas Department of Transportation to stabilize soil along roadsides, the spread of this invasive occurred quickly and broadly (Alofs and Fowler 2013). Heavy invasions of KR bluestem can form near monocultures with cover values close to 95% (Alofs and Fowler 2013; Fowler and Simmons 2009). Ironically, where KR bluestem is prolific on the Eastern Edwards Plateau, complete cover by *J. ashei* is the only habitat which prevents the growth of KR bluestem (Fowler and Simmons 2009; Gabbard and Fowler 2006); however, once *J. ashei* canopies are removed, KR bluestem can easily dominate primary successional vegetation (pers. obs.). The prior assumes that KR bluestem can disperse its seeds into these habitats. Potential dispersal routes include wind dispersal and seeds caught in machinery treads or carried inside a vehicle (Veldman and Putz 2010). It remains to be determined whether the invasion of KR bluestem into healthy systems is an inevitable reality based on time, or if healthy systems are able to resist homogenization and coexist with KR bluestem.

I hypothesize that the type of *J. ashei* removal and accompanying land management practices influence the composition of returning grasslands on the Edward's Plateau. Landowners commonly remove *J. ashei* as a means to improve hunting quality, wildlife habitat, or increase biodiversity. The methods for removal, the post-removal management of *J. ashei* slash, and treatments applied to the area (such as seeding with native grass blends or hand lopping *J. ashei* seedlings) vary greatly. My study seeks to confirm that *J. ashei* encroachment reduces the richness and diversity of herbaceous species, and that its removal promotes the return of herbaceous grasslands. Secondly, I investigate if chainsaw, bulldozer, and Cedar Eater (or equivalent machinery) removal methods affect observed herbaceous cover, diversity, and relative species abundances and if these are determined by mode of *J. ashei* removal. Finally, the same herbaceous responses are correlated with different post-removal land management practices including the sowing of native grass seed mixtures and hand lopping of *J. ashei* seedlings. Our goal is to provide Central Texas landowners with practical and accessible advice regarding best practices for the management of our native habitats.

Materials and Methods

Study Site

Thirteen properties in Kerr and Bandera counties on the Edwards Plateau in the Hill Country of central Texas were utilized in this study (Fig. 1). The Hill Country lies on the eastern region of the Edwards Plateau and is characterized by tall, rugged hills and thin topsoils covering primarily layers of limestone and some granite (Foster 1917). Most of the region's thin dry soil is underlain with Karst limestone, especially on the steep hillsides and canyon walls which resulted from the dissolution of ancient calcareous layers (Foster 1917; Jackson and Van Auken 1997; USDA web soil survey: websoilsurvey.sc.egov.usda.org). Karst limestone is characterized by connected compartments that store water (Dreybrodt 1990; McCole and Stern 2007). In this semi-arid ecosystem these compartments can be an important source of water for plant roots (McCole and Stern 2007). Exposed karst features may also produce natural springs which can create lush habitats amidst generally xeric vegetation. Annual rainfall is approximately 25-38 cm/yr on the western side of the Edwards Plateau, transitioning in a gradient to the east which receives approximately 25-84 cm/yr (Foster 1917; Van Auken et al. 1980). The climate is also characterized by mean annual temperatures of 20°C, low humidity, intense sunlight, and rapid evapotranspiration (Foster 1917; Van Auken et al. 1980).

Two habitats make up the bulk of the Hill Country landscape. These are closed-canopy woodlands and open, savanna woodland. The closed canopy woodlands are dominated by *Juniperus ashei* (Cedar) and *Quercus fusiformis* (Texas Live Oak) with a mixture of *Quercus falcate* (Spanish Oak), *Prunus serotina* (Escarpment Black Cherry), *Quercus laceyi* (Lacey Oak), *Quercus sinuate var. breviloba* (White Shin Oak), *Dispuros texana* (Texas Persimmon), and

Arbutus xalapensis on steep canyon slopes (Texas Madrone) (Jackson and Van Auken 1997; McClean 1985; Wynd 1944; Yancey 1982). Canopy cover is dense in the closed-woodland areas, ranging from 50 to 100 percent. The dense shade allows for little growth of open grassland species (Wayne and Van Auken 2010). Instead *Carex planostachys* (cedar sedge) dominates along with *Bouteloua curtipendula* (Sideoats Grama) (Wayne and Van Auken 2010). Open woodland sites in this study had been cleared of woody plants (mostly composed of *J. ashei*) using various methods of removal, although in some cases landowners left standing large mature *J. ashei* and *Qercus* sp. In open sites, a variety of grasses, cacti, and other plants have reemerged including, *Bouteloua curtipendula* (Sideoats Grama), *Bouteloua rigidiseta* (Texas Grama), *Sporobolus compositus* (Tall Dropseed), *Eragrostis intermedia* (Plains lovegrass), *Digitaria cognata* (Fall Witchgrass), *Tridens muticus* (Slim Tridens), and *Schizachyrium scoparium* (Little Bluestem). *Opuntia engelmannii* (Texas Prickly Pear), *Glandularia bipinnatifida* (Prairie Verbeena), *Salvia fannaceae* (Meely Cupsage), and *Sida* spp. The non-native KR bluestem (King Ranch Bluestem) has invaded much of the grassland of the Hill Country, particularly along roadsides and in areas with disturbed soils.

Reconnaissance of all thirteen properties was performed in mid May to evaluate their usefulness for the study. Cleared sites with similar topography, soils, elevation, minimal slope, low vehicular traffic, and of appropriate size (at least 50 meters long) were identified and noted on aerial maps (Fig. 1). Of the thirteen, ten of the properties held more promising site locations or had multiple cleared sites on a single property making data collection more efficient. Regardless of reconnaissance evaluations, a landowner survey was sent to all landowners to assess land use history of each location (Appendix A). Any properties without clearing, without clear records of *J. ashei* clearing, or any surveys that were not returned were excluded. Of the original thirteen properties, eight were selected for use in gathering categorical data through written surveys. Each landowner filled out a land use history survey indicating the location of sites of previous cedar removal, the method of that removal, any post-removal management practices (including if/how woody debris was placed, any seeding following woody removal, and if landowners performed selective clearing of Juniper saplings). Areas previously used for livestock grazing were noted on a map as well as the time of such grazing and type of livestock. Areas and previously used for agriculture were noted on the same map along with the time of such activities. Landowners were also asked if the site was used for animal husbandry (indicating a high density of livestock in barns and pens) and if so for which species. Any crops planted on the property were also noted. Landowners additionally indicated the locations of any bird or deer feeders, deer blinds, and past fires. No sites are currently or recently used for grazing or agriculture, and have not experienced recent fires. Five Landowners returned surveys and data was collected on these properties for a total of 13 pairs(Fig. 1).

Each cleared site was paired with a parallel dense cedar stand (a.k.a. cedar brake) used as a control. Sites were selected to maintain the least possible distance between *J. ashei* removal treatments and control pairs, thus minimizing changes in soil, slope, aspect, and other environmental or topographic variables. Any *J. ashei* removal treatments which were not flat were paired with controls of equal slope and aspect changes. Each transect was 50 meters in length. Sites were purposefully chosen away from confounding factors such as dirt roads, washes, slopes, non-representative anomalies in the landscape (such as campfire rings, litter, sites of feral hog rooting, etc.), or areas with heavy traffic. Soil types between sites were similar, all consisting of limestone parent rock, low water holding capacity, similar pH, and a few various minor components (Table 1). Data were collected from 29th of September to 10th November 2013, during peak herbaceous flowering.

Transect Procedures

A line transect method specifically intended to sample the representative cover of herbaceous vegetation was used for data collection. Designed by Whittaker, this method is part of a larger protocol designed to survey herbaceous, shrub, and forest vegetation over large areas (Daubenmire 1959; Ghorbani et al 2011; Goldsmith and Harrison 1976; McDougall and Morgan 2005; Schmida 1984). Areas of cedar removal used for transect establishment and data collection were identified using satellite maps and communication with individual land owners. Once the placement area for two adjacent transects in intact and removed cedar areas was decided, the location orientation of the transect was mapped for future research at the same location

Beginning at meter five, percent cover data was collected from a 1 x 1 m plot frame every five meters for 50 meters, alternating sides of the transect.   Percent cover data were collected on bare ground, cryptobiotic crust, rock, cedar leaf litter, non-cedar leaf litter, fine mulch, deadfall (defined as wood chunks larger than two cm), herbaceous litter, standing dead biomass, total herbaceous cover, native and non-indigenous herbaceous cover, individual herbaceous species cover, woody canopy cover, and individual cover for each species found within the plot's boundaries. I also noted whether the plot had any type of soil disturbance or anomaly (such as an ant bed, hog rooting, or animal feces). These data provided an estimate of species richness and the individual species percent cover data were used to calculate a

Shannon-Wiener Index for diversity of each transect. Unknown species were given a specimen number and a sample located outside the plot was taken for Trinity University's herbarium records. Most unknown plants were juvenile individuals and accounted for little biomass in any plot. References used for identification included Grasses of the Texas Hill Country (Loflin and Loflin 2006), Brush and Weeds of Texas Rangelands (Rector and Hanselka 2008), Wildflowers of the Texas Hill Country (Enquist 1987), Shinners and Mahler's Illustrated Flora of North Central Texas (Diggs et al. 1999), and consultations with botanist Dr. Floyd R. Waller who helped me identify unknown specimen. A parallel transect immediately adjacent to the first was then run inside the cedar brake as a control for environmental variables differing between sites. At the eight properties, thirteen transect pairs were established for a total of 26 transects (cedar brake and controls) and 257 plots.

Statistical Analysis

Individual plot cover percentages describing vegetative cover, percent native grass cover, percent invasive grass cover, and herbaceous cover were averaged per transect. Averaging allowed increased statistical power. Species richness quantified the number of species found within each transect. Shannon-Wiener diversity index was calculated using species proportions averaged across all ten plots per transect.

Predictors were selected from the land use history surveys. Modes of *J. ashei* removal were all aggregated into one predictor with 4 categories: chainsaw, bulldozer, Cedar Eater, or control. The second predictor examined post-removal land management. This predictor asked if the cleared area had been seeded, with 3 categories: seeded, not seeded, and control. Thirdly, post removal management by killing any *J. ashei* seedlings with hand loppers had 3 categories: lopped, not lopped, and control. Finally, transects were clearing in 7 different years. This with an additional control category made 8 total categories for the year cleared predictor.

Tests appropriate for categorical predictors and continuous response variables, with consideration for response variable normality, were used to determine significance. Anderson Dally and Shapiro-Wilk test were performed on response variables using R Studio. PNG and SWD were normally distributed while VC, PIG, THC, and SR were not. Based on normality, appropriate multi-factorial analyses were performed. For normal responses PNG and SWD, I conducted a factorial ANOVA to glean a broad picture of significance, followed by Tukey's Honest Significant Difference test for each predictor-response pair. This determined the direction of any observed significance for all possible combinations of available data. Non-parametric responses were tested using multiple Kruskal-Wallis tests, examining relationships between a single predictor and a single response. The Wilcoxon Signed-Rank Post-Hoc test was used to parse out individual contributions of each category toward the observed variation in response variables (Navidi 2011)

Results

Removal Mode:

Significant variation existed between removal treatments and percent native grass cover (factorial ANOVA $p=0.029$) and Shannon-Wiener Diversity (factorial ANOVA $p=.0004$). Removal mode varied significantly in vegetative cover, herbaceous cover, invasive grass cover, and species richness (multiple Kruskal Wallis tests). Herbaceous cover was significantly higher in chainsaw versus control transects, but was not significantly different between controls and any other type of removal (Fig. 2a). Herbaceous cover in bulldozer treatments was close to being significantly different from controls (p=0.069) (Fig. 2a). Vegetative cover was also significantly higher in chainsaw clearing than controls, with no other treatments being significantly different from each other or controls (Fig. 2b). Vegetative cover in bulldozer treatments was close to being significantly different from controls (p=0.069) (Fig. 2b). Native grass cover was significantly higher in both chainsaw and bulldozer treatments compared to controls, but not between Cedar Eater and controls, nor between clearing modes (Fig. 2c). Invasive grass cover was significantly higher than controls only in transects cleared by bulldozer (Fig. 2d). Diversity was significantly higher in chainsaw and bulldozer treatments compared to controls, but not significantly different between control and cedar eater (although it came close p=0.061). Nor was there a significant difference in diversity between clearing modes (Fig. 2e). No significant changes were found in comparison with species richness (Fig. 2f).

Post Removal; Seeding:

Seeding treatments did not vary significantly in percent native grass cover (factorial ANOVA p=0.637) nor Shannon-Wiener diversity (factorial ANOVA p=0.641). Seeding categories did vary significantly in vegetative cover, herbaceous cover, invasive grass, and species richness (multiple Kruskal Wallis tests). Herbaceous cover was significantly higher in both seeded and non-seeded transects than control, but was not significantly different between one another (Fig. 3a). The same trend was seen in vegetative cover (Fig. 3b). Native grass cover did not differ significantly between either seeding treatment and controls, nor between seeded or non-needed transects (Fig. 3c). Invasive grass cover was significantly higher in seeded transects than controls, but non-seeded transects did not significantly differ from controls (Fig. 3d). Invasive grass cover in seeded and not seeded transects did not differ significantly from one another. Diversity did not differ between controls and either treatment, nor between treatments (Fig. 3e). Species richness was significantly higher in not seeded transects than in controls (Fig. 3f). Seeded transects' species richness were not significantly different from controls or not seeded transects (Fig. 3f).

Post Removal; Hand Lopping:

Lopping treatments varied significantly in percent native grass cover (factorial ANOVA p=0.0002), but did not change significantly in Shannon-Wiener diversity (factorial ANOVA p=0.221). Significant change was associated with lopping treatments in herbaceous cover, vegetative cover, percent invasive grass, but not in species richness though the p values were very close to .05 significance (multiple Kruskal Wallis tests). Herbaceous cover was significantly higher in lopped treatments compared to control, while not-lopped transects compared to controls were trending toward increased herbaceous cover (p=0.069). While the trend suggests herbaceous cover increases the change between factors did not reached the .05 significance level (Fig. 4a). Herbaceous cover of lopped and not lopped transects was not significantly different from one another. Not lopped and lopped transects had higher vegetative cover than controls, but lopped and not-lopped treatments did not differ significantly from one another (Fig. 4b). Native grass cover was significantly higher in lopped transects than controls, and lopped transects were also significantly higher than not-lopped transects (Fig. 4c). Not-lopped transects were not significantly different from controls although the data trended toward an increase in cover (p=0.085). Invasive grass cover was significantly higher in not-lopped transects than controls, but did not differ between lopped transects and controls (Fig. 4d). Lopped and not-lopped transects did not differ from one another in invasive grass cover. Neither lopped nor not lopped transects were significantly different in diversity than controls, nor are the two treatments different from each other (Fig. 4e). Species richness did not differ between controls and lopped or not-lopped treatments, but both were trending toward increases in richness ($p=0.052$ for lopped-control, and $p=0.060$ for not lopped-control)(Fig. 4f).

Year Cleared:

Years cleared had insufficient replicate transects to give meaningful results comparing years cleared and response variables. All linear regressions yielded similar results. Herbaceous and vegetative cover were closest to being significant ($p=0.096$ and $p=0.078$) and had strong negative correlations to the year cleared ($R^2=0.5414$ and $R^2=.5806$)(Fig. 5a; Fig. 5b). Native grass cover, and invasive cover were negatively correlated to the year cleared (R^2 values above 0.5), and had non-significant p values (Fig. 5c; Fig. 5d). Diversity and species richness also had negative trends; however, the correlations were not as strong ($R^2= 0.37$ and $R^2=0.15$ respectively), and p values were not significant (Fig. 5e; Fig. 5f).

Discussion

Year of *J. ashei* Clearing:

While results from this study had insufficient replicates to suggest a conclusive trend, the consistent negative slope of my regressions for all response variables calls for further investigation. Sites cleared further in the past trend toward increases in herbaceous, vegetative, native, and invasive cover, with concurrent increases in diversity and species richness. Many studies stress the role of rare or large weather events in community dynamics and plant life history strategies (Ansley et al. 2006; Brown and Archer 2011; Knapp and Soule 1998; Smeins and Fuhlendorf 1997). Given more data, weather events and yearly precipitation values could be analyzed to provide more specific insight into their role in grassland ecosystems and restoration.

Removal Mode:

While Cedar Eater removal does reduce the cover of *J. ashei*, the resulting grassland has low cover, diversity, and richness values that are not ideal for restoration purposes. Recalcitrant mulch from Cedar Eater removal likely prevented seedling recovery and thus results in low vegetative and herbaceous cover values (Sayer 2005). Neither diversity nor species richness show any significant increase from the control's low biodiversity baseline although diversity comes close. With all observed sites having been cleared in the past 7 years, estimations for the long term impacts of Cedar Eater's woody debris cannot be estimated in this study; however, the large relative abundance of brown to green matter will likely result in decreased decomposition for many years. This unavailable carbon could result in reduced microbe activity thus increasing decomposition time (Wolkovich et al. 2010).

While bulldozer removal is linked to increases in native grass cover and diversity, the treatment is also linked to increases of non-native invasive cover. Additionally, bulldozing treatments were close to being significantly different from controls when observing herbaceous and vegetative cover values. Firstly, these sites exhibit a high degree of standard error. This is due in part to the small sample effect (bulldozer n=3, Cedar Eater n=2, Chainsaw n=7). It is also probably due to the varying intensity of said invasion. For example several properties were almost entirely homogenized by KR bluestem, but others had less homogenous communities. Thus percent cover values were highly variable and spanned as much as a 50 percentage points in cover. Despite the high variation, the change from control to bulldozer is still significant. I hypothesize that bulldozers create opportunities for invasion on two fronts. First, it has been shown that vehicles and tire treads can carry invasive seeds into previously non-invaded areas (Veldman and Putz 2010). Secondly, we know that woody removal indirectly facilitates invasion because KR bluestem cannot grow beneath the woody canopy (Alofs and Fowler 2013). By removing *J. ashei* one allows KR bluestem new habitat in which to expand. This novel territory is initially low in interspecific herbaceous competition before species can recover. This allows a low diversity, low biomass, and thus unstable environment which can be easily invaded (Alofs 2010; Alofs and Fowler 2013; Levine and D'Antonio 1999; Lyons 2006; O'Connor et al. 2013). While this treatment is monetarily inexpensive, grassland quality suffers.

Chainsaw removal seems to be the most reliable and consistent option for restoring grassland quality following woody encroachment. Vegetative, herbaceous, native cover, and diversity were all significantly higher in chainsaw transects than controls. Invasive cover did not differ significantly from. Unfortunately, it seems that any removal of the canopy increases opportunity for invasion because not all chainsaw transects are invasive free (Alofs and Fowler 2013). Finally we can be confident in these results because chainsaw sites have more sampled sites (n=7).

These findings support the hypothesis that differences in removal mode influences the species composition of resulting grasslands, where bulldozer removal increases invasive cover. Few studies have attempted to quantify grassland change potentially caused by different types of *J. ashei* removal; however, the work is essential to understanding effective land management. Ansley et al. (2006) did compare types of chaining (a highly disturbing method where two heavy machines drag large chains between them to tear down wooded areas) and concluded there was little difference between two chain heights; however, compared to standing *J. ashei* canopies the cleared areas were significantly improved in herbaceous cover. While Ansley et al.'s study measures herbaceous cover values, it does not break down values into native or invasive categories. This is essential in making informed management decisions.

<u>Post Removal: Seeding</u>

Results of native and invasive grass cover following post removal herbaceous seeding suggests that this management approach is an unnecessary and potentially harmful treatment. Seeding did not have a significant impact on herbaceous or vegetative cover, native grass cover, species diversity or species richness. This is encouraging evidence that herbaceous seed banks are likely intact despite long years of *J. ashei* coverage (Ribiero et al 2010). Additionally, broadcast seed distribution techniques can be easily predated by many species (Ribiero et al. 2010). What is concerning is the increase in non-native invasive King Ranch Bluestem cover following seeding. I hypothesize that the mechanism for this invasion is contamination of the "native seed" mixtures with KR bluestem. Most seed mixtures advertise the purity of the mixture, indicating how much of the seed is potentially non-native. The problem is that any percentage, no matter how small, could allow KR bluestem to establish and spread through time. A solution to this problem is plug planting. Not only does this increase the success of germination under controllable conditions (Lyons pers. comm.), but also allows managers to identify and weed out any invasive plants which inadvertently appear.

The effects of fine thin mulching may influence the success of seeding treatments. At one site, a landowner had cleared a significant area followed by seeding with a native seed mix. In one section, he had spread a thin layer of fine *J. ashei* mulch over the cleared soil and this section had high native cover and diversity. Where a large pile of unspread mulch still stood, no vegetative growth occurred. Where spreading mulch had been neglected in conjunction with seeding treatments, KR bluestem had formed a very dense low diversity population. I was unable to quantify any changes due to mulching because it was sometimes impossible to differentiate *J. ashei* leaf litter from finely chopped *J. ashei* mulch. In this case, there is some force at work which I was unable to observe. I suspect that fine mulching helps protect the herbaceous seed bank from erosion, and may increase the water holding capacity of the soil. Additionally, large mulch piles or thick mulch coverings can hinder vegetative recovery.

<u>Post Removal: Lopping</u>

Killing *J. ashei* seedlings using hand loppers seems to be a strong and easy way to maintain the natural role of grass fires without the associated risks. Both lopping and not lopping are linked to higher vegetative and herbaceous cover values. This is expected and supports our initial hypothesis that any type of clearing is better than no clearing (although we know that some methods are better than others). It is surprising that total herbaceous cover only differs significantly between control and lopped transects (Fig. 4a); however, the P value comparing control and not lopped transects was very close to significant (p=0.069). Thus any change between lopped and not lopped transects was not robust.

Lopping also is effective for native grass regeneration, but we are uncertain of lopping's effect on invasive cover. Native cover increases make sense following lopping considering most

Texas native grasses are shade intolerant (Fowler 2005). Our results show this is a robust change, with lopping being significantly higher than both not lopped and control transects. Thus, shade created by *J. ashei* canopy (even a small area beneath young trees) can be detrimental to native grass cover (Alofs and Fowler 2013); however, explaining observed increases in invasive cover in not lopped transects is more difficult. One does not expect shade intolerant invasive species to increase when *J. ashei* are not lopped and maturing readily. I contend that lopping may not be independent of other clearing or management practices. It is possible that lopping indicates specific landowner priorities such as hunting, aesthetic appeal, or habitat restoration. Thus it follows that landowners desiring a more aesthetic landscape may invest more effort in clearing and management practices that enhance grassland restoration. Others may only desire to clear open space for deer blinds and feeders, which requires a relatively small investment. Such differing values may lead to differing quality of care for each site, which could have effects on native and invasive cover that this study is unable to quantify.

<u>Practical Recommendations:</u>

Taking time to remove *J. ashei* with a chainsaw followed by periodic lopping treatments seems to be the most effective tool for grassland restoration. We suggest that controlling the maturation of *J. ashei* with hand lopping treatments mimics the natural role of wildfires and may help maintain grasslands for years to come. While relatively inexpensive and quick removal methods, such as bulldozing and Cedar Eater, do remove the *J. ashei* canopy, both have large negative impacts. Effort should be taken to develop affordable clearing businesses and practices which specifically consider grassland quality following removal. The chance of invasion by non-native species seems to increase when bulldozers are used. Furthermore the thick mulch created by Cedar Eaters is too recalcitrant to be beneficial for timely grassland regeneration. While seeding treatments theoretically were a good idea, it appears to be unnecessary due to existing herbaceous seed banks, and may increase opportunities for non-native invasion. Future studies ought to address mulching practices in this environment. Additionally, prescribed fire is used in many countries and states to help maintain herbaceous diversity and reduce high fuel load buildup. Landowners could benefit greatly from a list of practical restoration techniques focusing on prescribed burns.

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Table 1: Soil types per transect site. US Soil Survey referenced to collect typical pH balances for the major type of soil, soil's water holding capacity, and major and minor soil components at the site of interest.

Site	pH (0-5 inches)	Available H_20	Map Unit	Map Unit Secondary
1A	7.5	0.6	Eckrant	kerrville Association
1A	7.5	0.6	Eckrant	kerrville Association
1B	7.5	0.6	Eckrant	kerrville Association
1B	7.5	0.6	Eckrant	kerrville Association
3A	7.5	0.6	Eckrant	Rock Outcrop
3A	7.5	0.6	Eckrant	Rock Outcrop
3C	7.5	0.6	Eckrant	Rock Outcrop
3C	7.5	0.6	Eckrant	Rock Outcrop
5A	7.5	1.1	Eckrant	comfort association
5A	7.5	1.1	Eckrant	comfort association
5B-1	7.5	1.1	Eckrant	comfort association
5B-1	7.5	1.1	Eckrant	comfort association
5B-2	7.5	1.1	Eckrant	comfort association
5B-2	7.5	1.1	Eckrant	comfort association
5C	7.5	1.1	Eckrant	comfort association
5C	7.5	1.1	Eckrant	comfort association
6A	7.5	1.1	Eckrant	comfort association
6A	7.5	1.1	Eckrant	comfort association
7B	8.2	2.6	kerville	minor components
7B	8.2	2.6	kerville	minor components
8A	7.5	1.1	Eckrant	comfort association
8A	7.5	1.1	Eckrant	comfort association
8B	7.5	1.1	Eckrant	comfort association
8B	7.5	1.1	Eckrant	comfort association

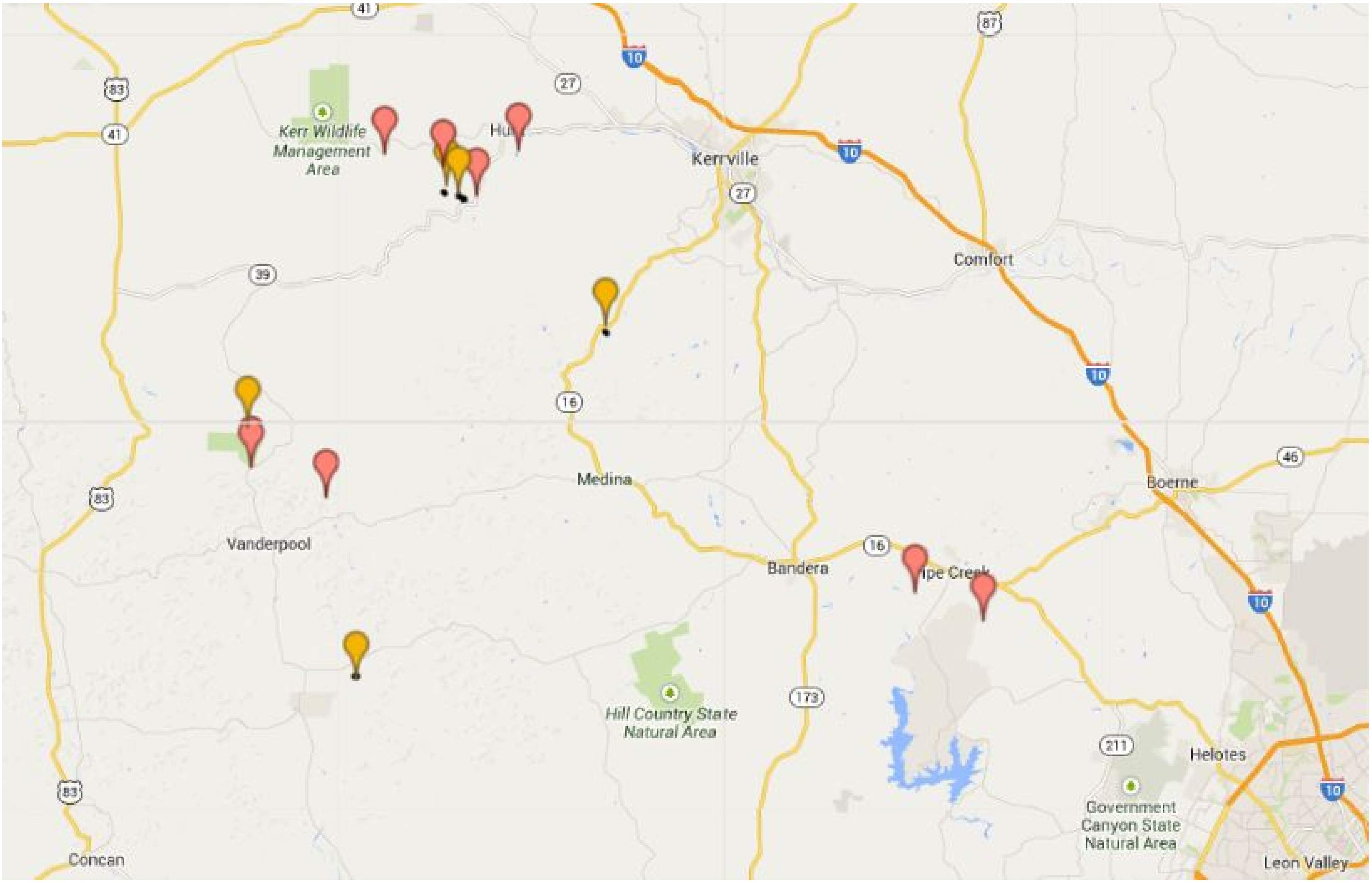

Figure 1: Map of 13 reconnoitered properties in Kerr and Bandera counties. Pink markers were reconnoitered but did not have data collected on. Gold markers are where data collection occurred.

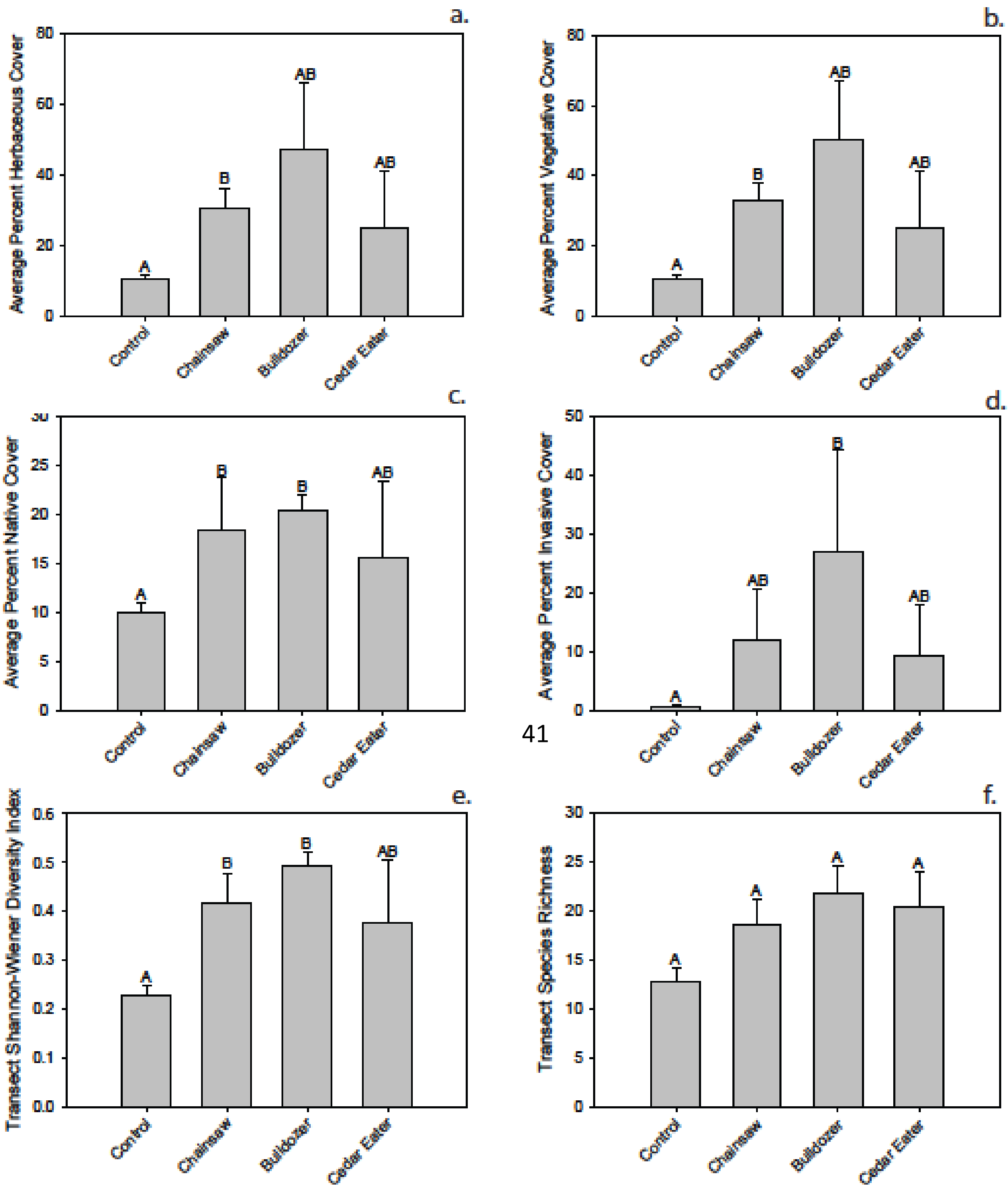

Figure 2: Responses by removal mode. Error bars represent ± 1SE. Different letters indicate values were significantly different ($p<0.05$) when using appropriate tests for normally and not normally distributed data. a). Removal modes by observed variation in transects' average percent herbaceous cover . b). Removal modes by transects' average vegetative cover. c). Removal modes by transects' average native grass cover. d). Removal modes by transects' average invasive grass cover. e). Removal modes by transects' Shannon-Wiener Diversity Index. f). Removal modes by transects' species richness.

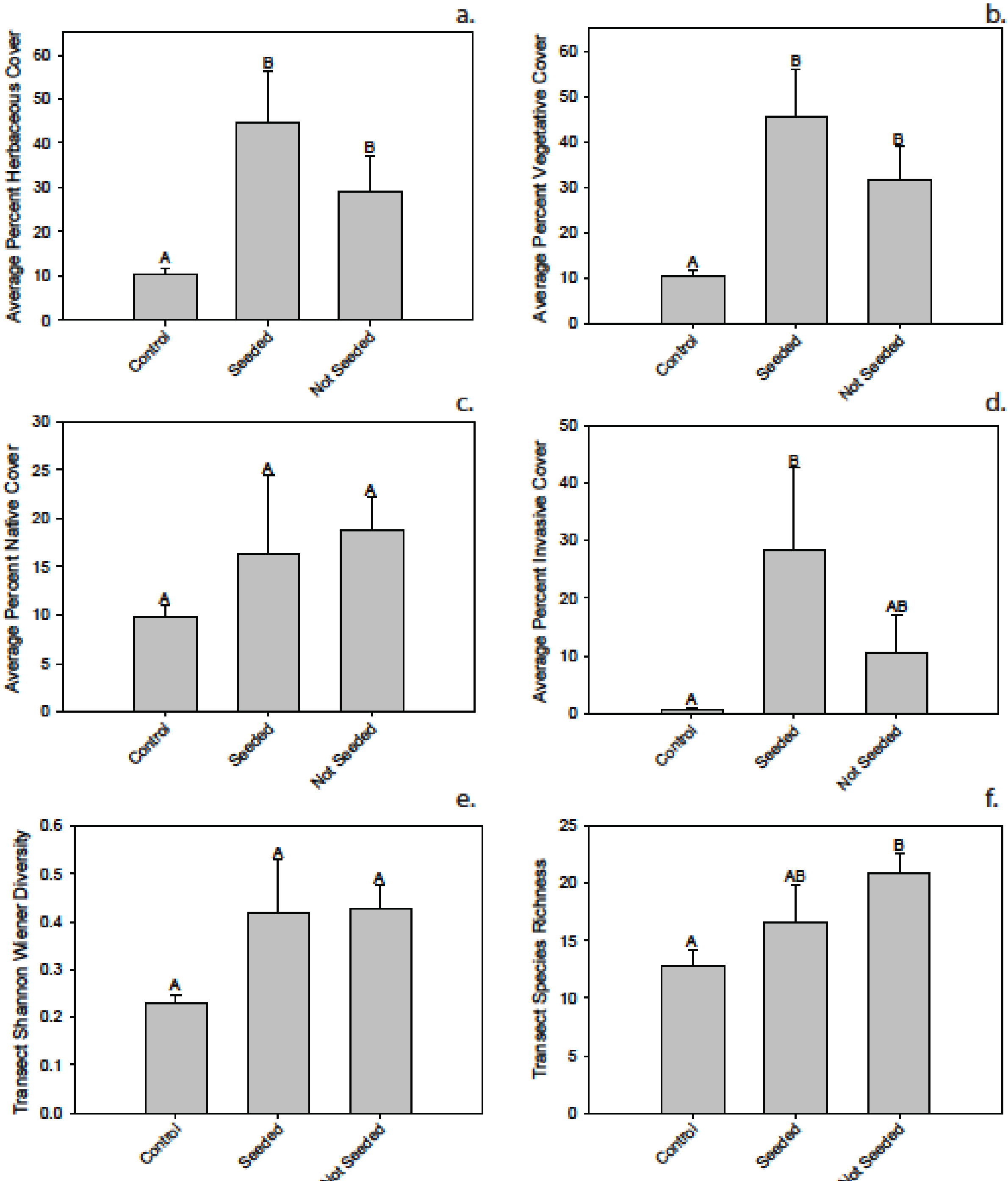

Figure 3: Responses by seeding treatments. Error bars represent $\pm$ 1SE. Different letters indicate values were significantly different ($p<0.05$) when using appropriate tests for normally and not normally distributed data. a). Seeding by observed variation in transects' average percent herbaceous cover. b). Seeding by transects' average vegetative cover. c). Seeding by transects' average native grass cover. d). Seeding by transects' average invasive grass cover. e). Seeding by transects' Shannon-Wiener Diversity Index. f). Seeding by transects' species richness.

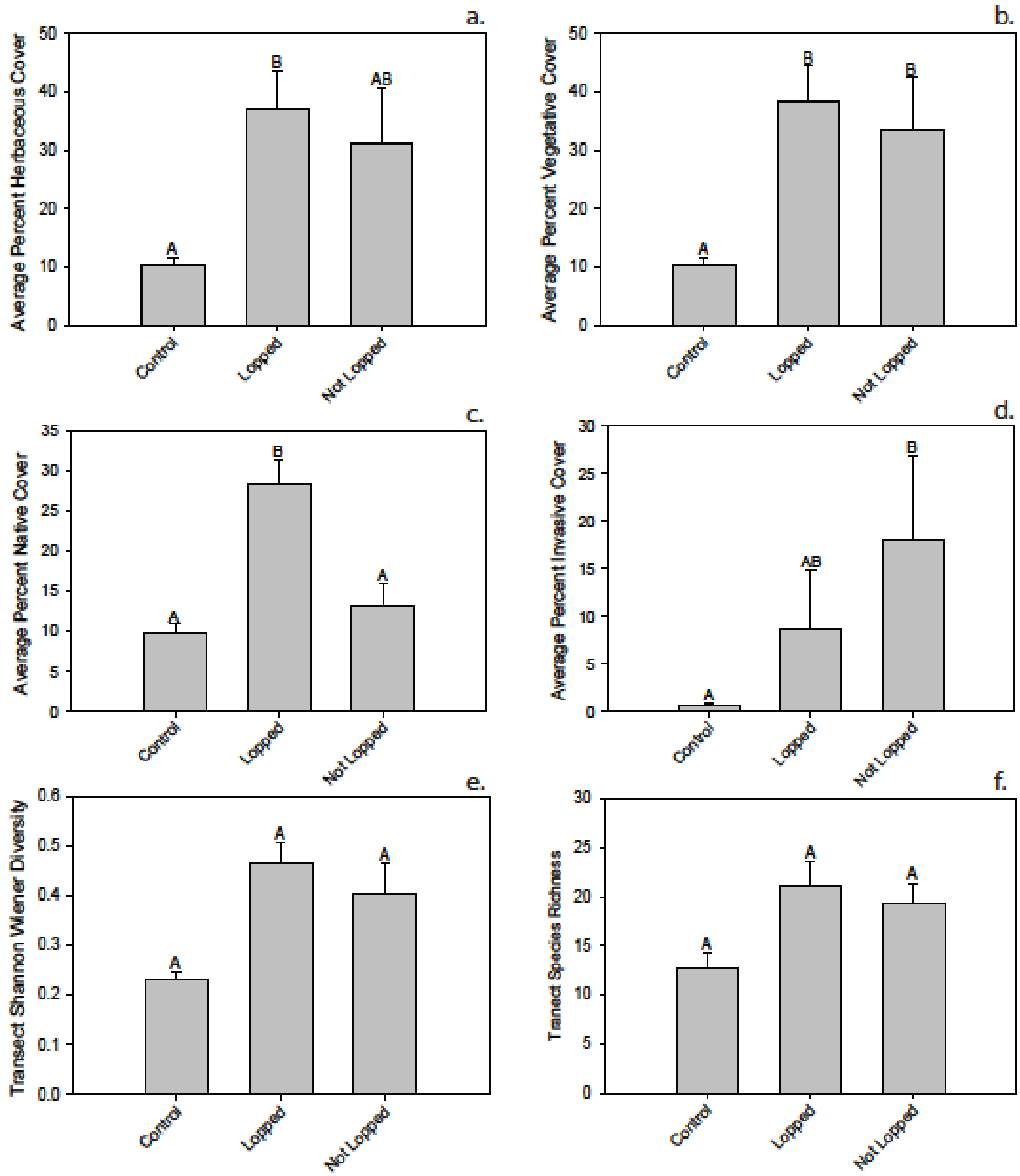

Figure 4: Responses by lopping treatments. Error bars represent ± 1SE. Different letters indicate values were significantly different ($p<0.05$) when using appropriate tests for normally and not normally distributed data. a). Lopping by observed variation in transects' average percent herbaceous cover . b). Lopping by transects' average vegetative cover. c). Lopping by transects' average native grass cover. d). Lopping by transects' average invasive grass cover. e). Lopping by transects' Shannon-Wiener Diversity Index. f). Lopping by transects' species richness.

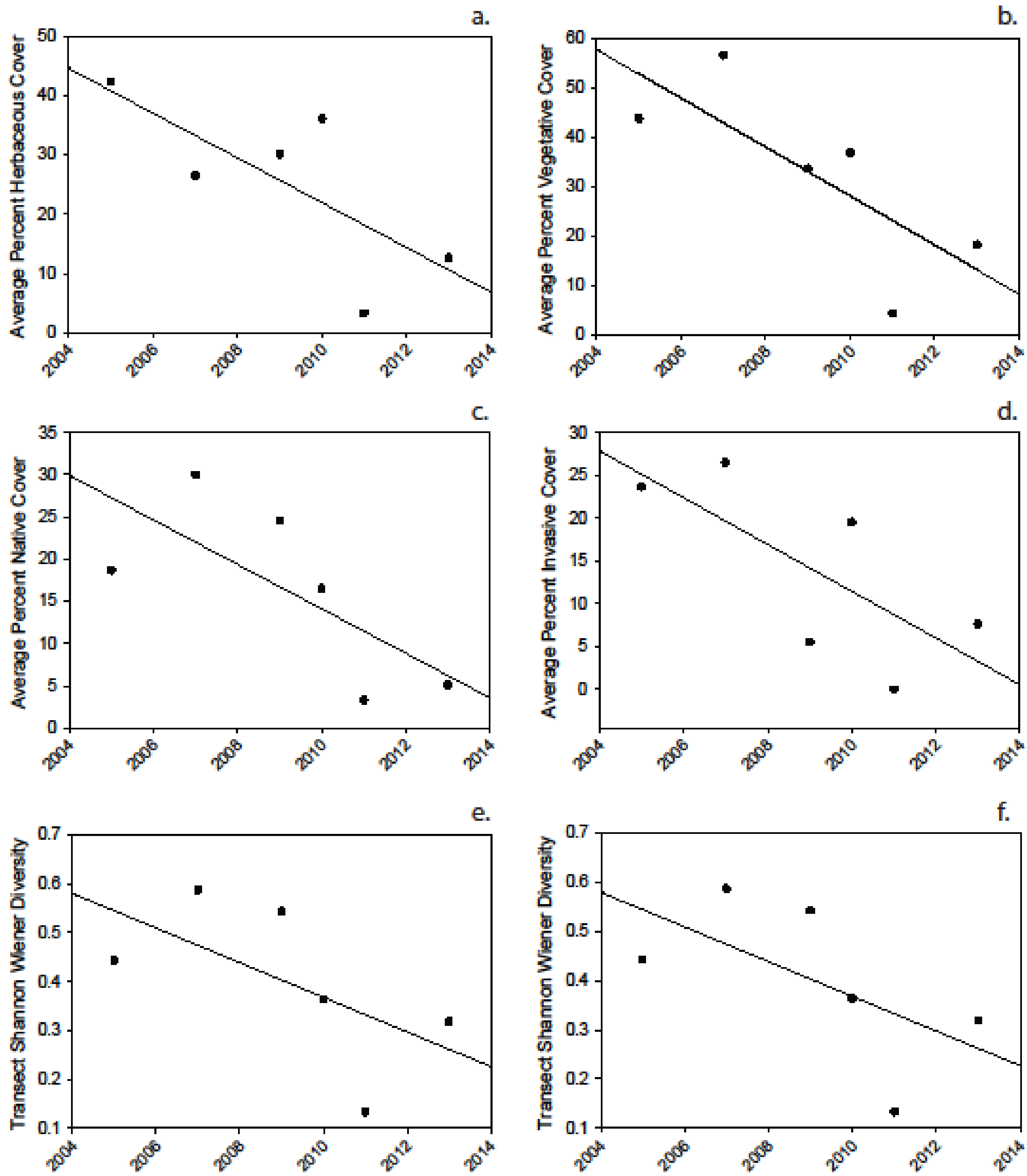

Figure 5: Linear regressions of responses compared to year of *J. ashei* clearing. a). Herbaceous cover by year cleared . b). Vegetative cover by year cleared. c). Native cover by year cleared d). Invasive cover by year cleared. e). Transects' Shannon-Wiener diversity by year cleared. f). Transects' species richness by year cleared

Appendix 1: Landowner Survey

Please mark areas with the following on the map of your property provided, and then fill out the questionnaire accordingly. If you do not have the digital capabilities to mark the attached map, we can mail you a copy to annotate by hand. Return address is listed at the end of this document.

1) Areas cleared of cedar. Please draw a polygon around the area and label each area A, B, C, etc.
2) Areas that have been grazed (at any point in the properties history). Please draw a polygon around the area and label each area 1, 2, 3, etc.
3) Areas used for agricultural purposes. Please draw a polygon around the area and label I, II, III, etc.
4) Locations of past fires. (Please mark with an PF)
5) Locations of deer blinds and deer feeders. (Please mark with a DB-blind or DF-feeder)
   a) Do you use bird feeders that deer, turkey, or hog opportunistically frequent?

<u>For locations cleared of cedar please answer the following:</u>

Cleared Cedar Site A:

- i) When was the site initially cleared?
- ii) Has it been cleared since?
- iii) If yes, how frequently? (List dates if possible).
- iv) How was the cedar initially removed? Please circle all that apply, or elaborate.
  - (1) Chaining
  - (2) Cedar Eater, or equivalent machinery, that mulches cedars down to ground level with minimal soil disturbance.
  - (3) Hydraulic Shearing: blade cuts 5-20 inch diameter tree at base for minimal ground disturbance.
  - (4) Bulldozing
  - (5) Grubbing (Root removal by machine without bulldozing)
  - (6) Brush Hog Attachment on Tractor (removes greenery from small cedars)
  - (7) Greenery Removal by Chainsaw
  - (8) Root Removal by Shovel
  - (9) Other (please elaborate): ____
- v) Following the initial removal, what was done with the wood?
  - (1) Trees left where felled?
  - (2) Trees placed in piles and not burned
  - (3) Trees placed in burn piles and burned
  - (4) Trees physically removed from site
  - (5) Trees mulched in place (mulch may be large and slow to decompose)
  - (6) Trees mulched and spread thin
  - (7) Trees mulched into piles of chips
  - (8) Other (please elaborate): ____

vi) Do you conduct cedar seedling removal using loppers?

vii) Following the initial removal, did you change the approach in subsequent removals? If yes, please explain.

<u>For locations with past grazing please answer the following:</u>

Grazed Site 1: Has the site been grazed in the past?

If yes, when was it last grazed?

If yes, what types of livestock were present?

(If you have indicated more than three sites that have been grazed, please answer the questions below on an additional piece of paper. Be sure that the numbers on the map corresponds to the number on the survey questions.)

For locations used for agricultural purposes:

Ag Site I: How long ago was the site used for agricultural purposes?

Was the site used for animal husbandry (e.g., barns, pens)? If yes, what types of animals?

Was the site used for agriculture? If yes, what types of crops?

(If you have indicated more than three agricultural sites, please answer the questions below on an additional piece of paper. Be sure that the numerals on the map corresponds to the numerals on the survey questions.)