RESTORING NATIVE PLANT AND ARTHROPOD COMMUNITIES IN GULF COASTAL PRAIRIES FOLLOWING PLANT INVASION AND DROUGHT

by

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ABSTRACT

Plant invasions are a threat to biodiversity, as changes in plant community characteristics resulting from invasion can affect other organisms, such as arthropods. The effects of invasions may interact with other disturbances and alter the efficacy of restoration strategies. We sought to understand the effects of Old World bluestem grasses (OWBs, *Bothriochloa*, *Dichanthium* spp.), which have become dominant in prairie ecosystems and reduce the quality of habitat for wildlife. In an attempt to reduce OWBs, we applied treatments to modify soil conditions to a state which favors native plants and arthropods. We conducted our research in 2011, which coincided with extreme drought and provided us with the opportunity to test the efficacy of soil modification under varying conditions. First, we explored the effects of plant invasion and drought on native plant and arthropod communities by comparing characteristics of plots dominated by native plants to plots dominated by OWBs. As drought subsided, we observed a shift from an arthropod community driven by detritivores to one driven by herbivores associated with plant invasion. Arthropod communities were dominated by invasive species. Second, we explored the efficacy of soil modification and seeding treatments to reduce OWBs in the presence and absence of drought based on a field experiment and a more controlled microcosm experiment. Although changes in soil chemistry from soil treatments were short-lived, we observed reduced dominance of OWBs in areas treated with soil disturbance and seeding in both experiments and we observed no differences between experiments when we alleviated the effects of drought. Finally, we examined the concomitant effects of our soil modification and seeding treatments on arthropod communities in the field experiment. We observed fewer arthropods in treated plots than undisturbed OWB monocultures, but soil and seeding treatments increased arthropod diversity and reduced dominance of invasive arthropods relative to undisturbed OWB monocultures. Based on our findings, simple soil disturbance in combination with seeding of native plants may increase diversity of native plants and arthropods where invasive plants are dominant in the short term, but monitoring over longer time frames may reveal additional benefits from soil modification.

CHAPTER ONE

INTRODUCTION TO THESIS

Introduction

The spread of invasive plant species threaten biodiversity at a global scale (Chornesky and Randall 2003; Reichard et al. 2005; Vitousek et al. 1996; Wilcove et al. 1998). Anthropogenic activities, such as global trade and land-use, have increased the likelihood and frequency of invasions into novel environments (Bryson and Carter 2004; Hobbs et al. 2009; Reichard et al. 2005). Changes in climatic conditions also may promote the establishment and spread of invasive plants in new locations (Bradley et al. 2009). As the rate of change in environmental stressors increases, we are likely to observe concomitant changes to likelihood and impact of invasions (Chornesky and Randall 2003; Hobbs and Huenneke 1992; IPCC 2007; Mack and D'Antonio 1997; Tylianakis et al. 2008; Vitousek et al. 1996).

Invasive plants typically simplify characteristics of plant communities by reducing diversity of both species and structure (Brooks et al. 2004; Gaertner et al. 2009; Levine et al. 2003; Vilà et al. 2011). Invasive plants also can alter ecological processes, such as fire and nutrient cycling (Blumenthal et al. 2009; Brooks et al. 2004; D'Antonio and Vitousek 1992; Ehrenfeld 2003; Reed et al. 2005; Vitousek 1990), which can promote feedback loops that increase dominance of invasive plants (Chornesky and Randall

2003). Under these conditions, traditional restoration tools, such as prescribed fire or herbicide, may no longer alter competitive relationships to favor native plants.

Researchers recently have stressed the importance of understanding soil properties in altered landscapes to increase restoration success (Heneghan et al. 2008; Vogelsang and Bever 2009). Taking this idea one step further, modifying the physical and chemical properties of the soil may alter competitive relationships and support establishment of native plant species. Some native plant species tolerate a wider range of soil pH levels than nonnative plants, and may grow, germinate, and acquire nutrients more quickly in acidic or alkaline soils (Elliot et al. 2013; Longhurst et al. 1999; Owen and Marrs 2000; Tibbett and Diaz 2005). Plant species that can tolerate toxic acidifying ions (such as Al^{+3} and $Fe^{+2\text{-}3}$) and phosphorus-limited soils, for example, may compete with invasive plant species that are less tolerant of acidic conditions (Farrell et al. 2011; Tibbett and Diaz 2005). Restoration projects that reduce soil pH with additions of elemental sulfur, ferric compounds, and organic matter have reduced the spread of weedy plants and increased establishment of native plants (Farrel et al. 2011, Novak et al. 2009, Owen and Marrs 2000, Tibbett and Diaz 2005). Adding lime also has increased soil pH in degraded landscapes, resulting in increased calcium and manganese availability, facilitated the establishment of both endangered and indicator plant species, and reduced recruitment of invasive plants (Dorland et al. 2005, Elliott et al. 2013, Kirkham et al. 2008, Longhurst et al. 1999).

Although altering soil pH may favor some plant species over others, some nonnative plants can alter soil pH (Alerding and Hunter 2013; McGrath and Binkley 2009). As such, native plant species may be favored only when resources in soils are altered to match natural and historic conditions (Blumenthal et al. 2003). Changes in nutrient availability, such as increased soil nitrogen from agriculture or nitrogen-fixing plants, may increase the competitive ability of invasive plants (Abraham et al. 2009; Alpert 2010; Alpert and Maron 2000; Blumenthal 2009; Huenneke 1990; Siemann and Rogers 2007; Sigüenza et al. 2006; Suding et al. 2004; Vitousek et al. 1996). Nitrogen may be reduced in the soil by adding organic carbon, such as sugar or wood components, which encourages soil microorganisms to consume both abundant carbon and nitrogen (Alpert 2010). These microorganisms compete with nonnative plant species for available nitrogen, which may result in a reduction in abundance of invasive plants (Alpert 2010; Blumenthal et al. 2003, 2009). Success of carbon additions are dependent on species and environment (Alpert 2010), but adding carbon has reduced dominance of some species of invasive grasses and woody plants (Alpert and Maron 2000, Blumenthal 2009, Corbin and D'Antonio 2004, LeJeune et al. 2006).

Increasing the presence of vesicular-arbuscular mycorrhizae (VAM) also may increase plant establishment and nutrient acquisition and alter competitive relationships between native and nonnative plants (Archer and Pyke 1991, Bunn et al. 2009, Callaway et al. 2003). Soil biota can be augmented by adding inocula to promote establishment of mycorrhizal fungi (e.g., Heneghan et al. 2008, Vogelsang and Bever

2009), which may be essential for the establishment of native plants (St. John 1980). Mycorrhizae and other mutualistic symbionts are affected by soil disturbance and may not survive if they have not coevolved a mutualistic relationship with the plant species (Archer and Pyke 1991, Biondini et al. 1985). Inoculations of mutualistic symbionts may be necessary to aid in establishment of native plant communities.

Restoring the native plant community may facilitate the restoration of other communities affected by invasive plants. Changes in the plant characteristics associated with plant invasion may alter the quality of habitat for arthropods (Crist et al. 2006; Gratton and Denno 2006; Litt and Steidl 2010; Litt et al., in press; Pearson 2009; Standish et al. 2004; Tallamy 2004; Wolkovich 2010). Because arthropods provide valuable ecosystem services, such as pollination and decomposition, and are an important food resource for avian and mammalian species (Archer and Pyke 1991; Brussaard 1997; Burger et al. 2003; de Bruyn 1999; Folgarait 1998; Potts et al. 2010; Snyder and Hendrix 2008; Wiens and Rotenberry 1979; Wilson et al. 1987), the increase in presence or abundance of arthropods following native plant restoration may increase ecosystem health.

Old World bluestems (OWBs, *Bothriochloa* and *Dichanthium* spp.) are a group of nonnative grasses introduced from Africa, Asia, Eurasia, and Australia (Celarier 1958; USDA-NRCS 2014) that have become dominant in the central and southern Great Plains of the United States. OWBs were introduced to the United States in the early twentieth century as potential cattle forage, due to advantages in productivity, nutrient

acquisition, and grazing tolerance (Berg 1993; Coyne and Bradford 1985; Dabo et al. 1988; Dewald et al 1988; Nixon 1949). However, OWBs are of lower forage value than previously thought; mature plants become less palatable in comparison to native range plants (Berg and Sims 1995; Dabo et al. 1988; Dewald et al. 1988; Gillen and Berg 2001). OWBs were and still are planted throughout the Great Plains to reduce soil erosion on reclamation sites and highways (Berg 1993; Harmoney et al. 2004), contributing to its spread.

OWBs are warm-season, perennial, C4 bunchgrasses that can grow via stolons or produce copious amounts of seed; some species display apomictic behavior (Coyne and Bradford 1985; de Wet and Harlan 1970; Nixon 1949; Schmidt and Hickman 2006). Increased dominance of OWBs can alter fire regimes and nutrient cycling (Reed et al. 2005), and reduce diversity of native plant and wildlife communities (Cord 2011; Gabbard and Fowler 2007; Hickman et al. 2006; Sammon and Wilkens 2005; Schmidt et al. 2008; Woodin et al. 2010).

Traditional management strategies, such as fire and herbicides, have not reduced OWB populations successfully. The effect of prescribed burns on reducing OWB dominance has been variable (Berg 1993; Ruckman et al. 2011; Simmons et al. 2007; Twidwell et al. 2012), and the density and productivity of OWBs may increase following burns (Berg 1993; Gabbard and Fowler 2007). Applications of herbicides to OWBs generally are broad-scpectrum, which may impede revegetation of native plants (Harmoney et al. 2007; Ruckman et al. 2011; Ruffner and Barnes 2012), and OWBs can

recover within one year of appliec herbicides (Harmoney et al. 2004, 2007; Mittelhauser et al. 2011). Application of fire and herbicides has produced variable results at reducing dominance of OWBs (Harmoney et al. 2004, 2007; Mittelhauser et al. 2011; Ruckman et al. 2011; Ruffner and Barnes 2012; Simmons et al. 2007; Twidwell et al. 2012). Soil modification has not been tested as a restoration tool to reduce OWBs, but could serve as an alternative to traditional management strategies.

In 2011, an extreme drought event occurred throughout the introduced range of OWBs (NDMC-UNL 2014) and persisted for several years in the southern portion of its range. Drought conditions can exacerbate the effects of plant invasion on native plant communities (Boulant et al. 2008; Castillo et al. 2007; Crous et al. 2012; Everard et al. 2010; Miller 1994; Schumacher et al. 2008), and given that OWBs are drought-tolerant (White and Dewald 1996), the combination of plant invasion and drought conditions may inhibit restoration success. The drought also provided us with the opportunity to test how effective soil modification treatments could be in the presence of multiple disturbances (i.e., drought and plant invasion).

We conducted a field study to investigate the relationships of plant invasion and drought and their effects on native plant and arthropod communities. We also examined the efficacy of soil modification to reduce dominance of OWBs and favor the establishment of native plant and arthropod communities. In the second chapter, we explore the effects of plant invasion and drought on native plant and arthropod communities by comparing characteristics of two plant communities, one dominated by

OWBs and the other dominated by native plants. In the third chapter, we explore the efficacy of soil modification and seeding treatments to reduce OWBs in the presence and absence of drought based on a field experiment and a more controlled microcosm experiment. In our final chapter, we examine the efficacy of soil modification and seeding on restoring native plant and arthropod communities in the field experiment. By determining the effectiveness of soil modification for restoration of native plants and arthropods in landscapes dominated by OWBs, we may provide alternative management options for landowners and inform future research on OWBs and other warm-season grasses in prairie ecosystems of the southern and central Great Plains.

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CHAPTER TWO

SHIFTS IN COMPOSITION OF PLANT AND ARTHROPOD COMMUNITIES FOLLOWING PLANT INVASION AND DROUGHT

Contributions of Authors and Co-Authors

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CHAPTER TWO

SHIFTS IN COMPOSITION OF PLANT AND ARTHROPOD COMMUNITIES FOLLOWING PLANT INVASION AND DROUGHT

Abstract

Invasive plants can alter the structure and composition of native plant communities, with concomitant effects on arthropods. However, plant invasion may not be the only disturbance affecting native plant and arthropod communities, and multiple disturbances can have compounding effects. We conducted a field study to compare characteristics of plant and arthropod communities between communities dominated by nonnative Kleberg bluestem (*Dichanthium annulatum*) and communities dominated by native plants during drought from 2011-2013. We hypothesized that native plant communities would have more species of plants and arthropods than Kleberg bluestem, and the effects of drought would reduce richness of plants and arthropods at a greater rate in Kleberg than in communities of native plants. We sampled characteristics of vegetation, soil, and arthropod communities every summer during extreme (2011), moderate (2012) and non-drought (2013) conditions. Overall, native plant communities had more plant species/m^2 and forb cover than Kleberg regardless of drought severity, which may provide more plant hosts for herbivorous arthropods. Although native plant communities had more species/m^2 of arthropods than Kleberg during extreme drought, the number of species was comparable as drought subsided. Native plant communities also had more arthropods/m^2 than Kleberg during extreme drought, but this pattern reversed as drought subsided. Where invasive plants were dominant, arthropod communities dominated by detritivores were replaced with herbivores; arthropod communities were dominated by invasive arthropods. Plant invasion was associated with the dominance of an invasive hopper (*Balclutha rubrostriata*) and mites (Mochlozetidae) as drought conditions subsided, which in turn may have increased abundance of mite predators in Kleberg bluestem communities. Increased litter cover and available nutrients in native plant communities may be associated with an increase in pillbugs (*Armadillidium vulgare*), and the composition of plant litter may have increased habitat quality for decomposer arthropods. Ants were more abundant in native plant communities as drought conditions decreased, and were associated with increased richness of plants and arthropods as food. Understanding the mechanisms behind how native plants and arthropods are affected by multiple disturbances can provide insight for implementing restoration tools where invasive plants are dominant.

Introduction

Invasive plant species have altered ecosystem processes and caused economic losses (Bryson and Carter 2004; Chornesky and Randall 2003; D'Antonio and Vitousek 1992; Ewel et al. 1999; Vitousek 1990). Human activity has increased the likelihood of biological invasions and other landscape changes (Hobbs et al. 2009; IPCC 2007; Tylianakis et al. 2008; Vitousek 1990). Once established, invasive plants reduce richness of native plant communities and change vegetation structure (Gaertner et al. 2009; Hejda et al. 2009; Levine et al. 2003; Vilà et al. 2011), which may affect other organisms at different trophic levels, especially arthropods (Gratton and Denno 2006; Litt and Steidl 2010; Litt et al., in press; van Hengstum et al. 2014).

Changes in structure and composition of vegetation communities may alter availability and quality of habitat for arthropods (Lenda et al. 2013; Wolkovich et al. 2009; Wolkovich 2010). Many arthropods require specific plant species for food or reproduction sites (Bernays and Graham 1988; Burghardt et al. 2010; Tallamy 2004; Williams et al. 2011) and novel plants may not be recognized as habitat by native arthropod species (Brown et al. 2002; Burghardt et al. 2010; Grabas and Laverty 1999; Tallamy 2004; Williams et al. 2011). Changes in plant composition also may alter vegetation structure, such as plant cover, density, and height, which may affect behavior or movement of arthropods (Crist et al. 2006; Pearson 2009; Samways et al. 1996; Schirmel et al. 2011; Standish 2004; Wolkovich et al. 2009; Wu et al. 2009).

However, responses of arthropods to changes in plant composition or structure resulting from invasive plants may be taxa-dependent (Litt et al., in press).

Arthropods most likely to be altered by changes in plant composition include herbivores and pollinators (Litt et al., in press). Herbivorous arthropods may not utilize invasive plants due to novel physical and chemical defenses (Burghardt and Tallamy 2013; Burghardt et al. 2010; Carrol et al. 1998; Fortuna et al. 2013; Graves and Shapiro 2003; Tallamy 2004, Tallamy et al. 2010). Native pollinators may not recognize or be able to acquire nectar and pollen from novel hosts (Brown et al. 2002; Grabas and Laverty 1999; Williams et al. 2011). Alternatively, pollinators may prefer invasive plants that have higher densities of flowers or nectar loads, with negative consequences for native plants (Bjerknes et al. 2007; Chittka and Schürkens 2001; Woods et al. 2012).

Although some arthropods do not feed on living vegetation, plant invasions still may have indirect effects (Lenda et al. 2013; Litt et al., in press; Wolkovich 2010). Invasive plants can alter litter composition and modify soil moisture, mineralization rates, and soil pH, which may have adverse effects on the composition and abundance of decomposer arthropods (Alerding and Hunter 2013; Kappes et al. 2007; Mayer et al. 2005; McGrath and Binkley 2009; Wolkovich et al. 2009). Shifts in abundance, composition, or behavior of prey as a result of plant invasions can have concomitant effects on communities of predaceous arthropods (Alerding and Hunter 2013; Gratton and Denno 2006; Litt and Steidl 2010; Schreck et al. 2013; Tang et al. 2012). Furthermore, changes in arthropod communities following plant invasion can affect the

quality and availability of habitat for other trophic levels (Burghardt et al. 2008; van Hengstum et al. 2014; Litt et al., in press).

Plant invasion may be only one of many stressors influencing communities of native plants and arthropods; concurrent stressors can interact to have novel effects (Paine et al. 1998; Vitousek et al. 1996). Drought, for example, may reduce the ability of native plants to compete with invasive plants that tolerate drought conditions (Boulant et al. 2008; Castillo et al. 2007; Crous et al. 2012; Everard et al. 2010; Miller 1994; Schumacher et al. 2008). Drought may reduce abundance or richness of arthropod communities through direct mortality (Schultz et al. 2006; Xu et al. 2009) or changes in availability of habitat or food (Buchholz et al. 2013; Frampton et al. 2000; Kindvall 1995; Scheirs and De Bruyn 2005). Few studies have examined the combined effects of drought and plant invasion on arthropod communities, and understanding how native communities respond to multiple stressors in the environment may provide insights for conservation (Paine et al. 1998).

Old World bluestem grasses (OWBs, *Bothriochloa* and *Dichanthium* spp.) are a group of warm-season bunchgrasses from Africa and Eurasia that were introduced in the early twentieth century as a potential cattle forage (Celarier 1958; Nixon 1949). OWBs currently are planted to reduce soil erosion on reclamation sites and highways, contributing to its spread (Harmoney et al. 2004). OWBs may alter fire regimes, nutrient cycling, and soil chemistry (Dirvi and Hussain 1979; Reed et al. 2005), as well as the composition of native plant and animal communities (Cord 2011; Gabbard and Fowler

2007; Hickman et al. 2006; Sammon and Wilkens 2005; Schmidt et al. 2008; Woodin et al. 2010).

We developed a field-based study to investigate shifts in composition of plant and arthropod communities between landscapes dominated by OWB grasses and landscapes dominated by native plants. We sought to build on previous work (e.g., Cord 2011, Woodin et al. 2010) by focusing specifically on changes in the composition of arthropod communities at the functional group and species-level (Oliver and Beattie 1996). In 2011, a severe drought event occurred throughout the introduced distribution of OWBs (NDMC-UNL 2014), which persisted for several years in the southern portion of this area. This drought event provided us with the opportunity to explore relationships between plant invasion and drought severity on plant and arthropod communities. OWBs are drought tolerant (White and Dewald 1996), and plant invasion may exacerbate drought-induced stress on native plant and arthropod communities. We predicted that OWB-dominated landscapes would have fewer plant species, as well as lower species richness and abundance of herbivorous arthropods, with concomitant effects on decomposer and predaceous arthropods. We predicted that vegetation and arthropod characteristics (e.g., richness) would be affected more negatively by drought in OWB-dominated areas than in areas dominated by native plants. Finally, we predicted that the number of plant and arthropod species would increase as drought conditions subsided.

Methods

Study Area

We conducted our research at the Welder Wildlife Refuge (N 28.121155, W 97.442808), a 3,157-ha refuge located 12 km northeast of Sinton, Texas. The wildlife refuge represents an intermediate between the Gulf Coastal Prairie and Rio Grande Plain vegetative zones (Box 1961). We sampled within a plant community at the southernmost border of the refuge, which was classified historically as a mesquite-buffalograss community (Box 1961). The mesquite-buffalograss community consisted of buffalograss (*Buchloe dactyloides*), 3 awn grasses (*Aristida* spp.), Hall's Panicum (*Panicum halli var. halli*), and plains bristlegrass (*Setaria leucopila*), and dominant forbs which included croton (*Croton* spp.), horsemint (*Monarda citriodora*), and Mexican hat (*Ratibida columnifera*; Box 1961). The dominant woody vegetation included honey mesquite (*Prospis glandulosa*) and Texas huisache (*Acacia smallii*), with trace amounts of other acacias (Box 1961). However, this plant community now is dominated by Kleberg bluestem (*Dichanthium annulatum*), which became dominant in the refuge within the past decade (Goertz 2012). The Kleberg bluestem community is a monoculture (85-90% of total vegetation cover), with trace amounts of forbs (e.g., *Cienfuegosia drummondii*, *Ratibida columnifera*, and *Solanus elagnifolium*). The Kleberg bluestem community is bordered by a 2-m wide, disked firebreak on one side and a fence on the other.

We also sampled within a plant community that was dominated by native grasses, but without OWBs, which was 1.3 km from the Kleberg bluestem community. The native plant community included grasses such as: Hall's panicum, paspalum (*Paspalum* spp.), and knotroot bristlegrass (*Setaria ramiseta var. firmula*), and annual forbs such as ragweed (*Ambrosia cumanensis*), sumpweed (*Iva annua*), and sulphur mallow (*Cienfuegosia drummondii*). Woody vegetation adjacent to the native plant community included honey mesquite and Texas huisache, with trace amounts of blackbush acacia (*Acacia rigidula*).

We established 10, 6 x 9-m plots within the study area, five in the Kleberg bluestem community and five in the native plant community. We established plot size based on a concurrent study (Chapters 3 and 4). Plots in the Kleberg bluestem community were selected at random and were at least 1.5 m apart. We selected plots in the native plant community in areas that lacked woody vegetation; plots were at least 32 m apart.

Vegetation Sampling

We measured vegetation density, canopy cover, and maximum height by species on two 1-m^2 quadrats per plot, June-August 2011-2013. We placed quadrats at random within each plot for each sampling period, and were at least 1 m from plot boundaries to avoid edge effects. Plants were identified to species using Everitt et al. (2011) for grasses, Everitt et al. (2002) for woody plant species, and Everitt et al. (1999) for

herbaceous plant species, and cross-referenced with type specimens in the Welder Wildlife Foundation herbarium.

We measured species density of plants by counting the number of plants in each quadrat. We considered grasses as separate individuals if the crowns from stolons occurred more than 10 cm from the original crown base. Biennial or perennial species that appeared dead were counted due to uncertain dormancy responses to drought conditions. We estimated horizontal canopy cover (≤ 1-m tall) by species, as well as cover of bare ground and litter (vegetative material separate from living vegetation or growing structures attached to the ground). We then grouped plant species into specific cover classes that included grasses, forbs (herbaceous plants), and woody plants. Finally, we measured the height of the tallest plant of each species in each quadrat. We averaged vegetation variables for quadrats within each plot. We used species richness and canopy cover of plants as measures of community richness and composition, and plant density and height as measures of vegetation structure.

Soil Sampling

In May of each field season, we collected 1 L of soil from each quadrat to determine soil chemistry. We collected soil up to a depth of 15 cm in each quadrat and combined samples from quadrats within plots. Soil samples were analyzed by Texas Plant and Soil Labs (Edinburg, TX) to quantify soil pH, organic matter (% O.M.), and available nutrients (NO_3 and P_2O_5) using an extractable CO_2 method (McGeorge and Breazeale 1931; Texas Plant and Soil Labs 2012). We used these soil characteristics to

assess differences in soil chemistry between plant communities, and better understand any changes in the plant and arthropod communities (Brussaard 1997; Levine et al. 2003).

Arthropod Sampling

We sampled arthropods within the same 1-m^2 quadrats in each plot as we did when sampling vegetation. Although a variety of methods are used to sample arthropods, each method is taxonomically biased to some degree (Greenslade 1964; Southwood 1982, Standen 2000). In an attempt to sample the arthropod community completely, we used three techniques: pitfall traps, vacuum sampling, and Berlese-Tullgren funnels. We started sampling arthropods 24 hours after we completed vegetation sampling and waited at least 24 hours between each technique to allow the arthropod community to recover.

Pitfall sampling is an effective technique to capture terrestrial arthropods, such as beetles (Greenslade 1964; Triplehorn and Johnson 2005), harvestmen (Sabu et al. 2011), and arachnids (Bowen et al. 2004; Goetze et al. 2001; Uetz and Unzicker 1975; Work et al. 2002). We placed two pitfall traps (266-ml plastic cups) randomly within each quadrat, ensured that pitfall traps were flush with the soil surface and filled traps halfway with propylene glycol (Prestone Low Tox® Antifreeze/Coolant). We left the traps undisturbed for 24 hours, after which we collected the contents of all traps.

Vacuum sampling, or D-vac sampling, is a useful method for sampling arthropods in vegetation or on the wing in grasslands (Brook et al. 2008; Standen 2000). We used a

vacuum sampler (Model 122, Rincon-Vitoca Insectaries, Ventura, CA) to sample each quadrat for 90 seconds and transferred specimens to a plastic bag. We removed specimens attached to the net with an aspirator (BioQuip model 1135A, Rancho Dominquez, CA). To prevent or reduce predation, we placed cotton balls soaked with ethyl acetate in the plastic bag.

Berlese-Tullgren funnels generally are considered an efficient method for sampling diversity of soil-dwelling arthropods (Sakchoowong et al. 2007; Smith et al. 2008; Triplehorn and Johnson 2005) and are more efficient in dry environments when compared to other extraction methods (Sabu et al. 2011). We used Berlese-Tullgren funnels (BioQuip model 2845) and decreased the diameter of the mesh filter (0.32 x 0.32 cm) from the original model to keep soil particles from falling into the collecting cup. We collected 473 ml of soil from each quadrat and placed the sample within the upper part of the funnel. Soil and funnels were exposed to sunlight for 48 hours to facilitate extraction.

We combined samples from all techniques within each quadrat of each plot to obtain more comprehensive estimates of the arthropod community (Southwood 1982). We froze or stored all specimens in 70% ethyl alcohol for later sorting and identification. We identified all arthropods to family based on Krantz and Walter (2009) for mites, Richardson (1905) for isopods, Stockwell (1992) for scorpions, Summers (1979) for centipedes and millipedes, and Triplehorn and Johnson (2005) for insects and spiders. When possible, we identified arthropods to morphospecies (Oliver and Beattie 1996,

hereafter referred to as species) for greater taxonomic resolution. Specimens that could not be identified beyond family (e.g., all Acari, most Araneae) were not considered as separate species for analysis if other species had been identified from the same family.

We also assigned all arthropods to a single functional group that represented the role of each taxa in an ecosystem (Appendix A). We classified herbivores as arthropods that consume living vegetation as a majority of their diet. We classified pollinators as arthropods that consume pollen or nectar as a majority of their diet, or pollinate plants by consuming flowering parts of the plant (Triplehorn and Johnson 2005). We classified decomposers as arthropods that either consume dead animal or plant matter as a majority of their diet, or consume microorganisms (i.e., bacteria and fungi) and concentrate available nutrients in excrements (Brussaard 1997; Clarholm 1985). We classified predators as arthropods that consume other arthropods during at least part of their life cycle, and we also included parasitoids in this group. We designated ants (family Formicidae) as their own functional group, as ants perform multiple roles in ecosystems (Brussaard 1997; Folgarait 1998; Triplehorn and Johnson 2005; Wilson 1987). We did not assign immature or larval specimens to functional groups that had different life strategies than their adult morphs (e.g., Lepidoptera), due to a lack of taxonomic resolution; these specimens comprised <1% of all individuals sampled (Appendix B).

We used species richness and abundance of all arthropods and of each functional group as coarse measures of community structure and composition. We also

examined presence and abundance of species; presence indicated that the plot provided habitat and abundance provided a measure of habitat quality.

Precipitation

We obtained precipitation data from a nearby weather station at the headquarters of the Welder Wildlife Refuge, approximately 7.2 km from the study area. We quantified monthly precipitation from October 1956 (from the start of the water year, October 1) until September 2013 and we compared annual precipitation during our study to the long-term annual mean to assess the severity of drought. Lags between rain events and arthropod responses are common (Frampton et al. 2000; Tanaka and Tanaka 1982); we quantified precipitation 2-4 weeks prior to start of each sampling period to better understand changes in the arthropod community (Frampton et al. 2000; Tanaka and Tanaka 1982). We used the Palmer Drought Severity Index (NCDC-NOAA 2014) as a measure of drought severity for each field season (June-August) in the study.

Data Analysis

We examined differences in vegetation, soil, and arthropod characteristics between plant communities using generalized linear mixed models. We included plant community (native and Kleberg) and year (as a proxy for drought) as independent factors in all models and explored evidence for a two-way interaction (community * year). We removed the interaction term from models when $P > 0.1$, but retained all

simple effects in final models. When appropriate, we accounted for repeated measures and considered three possible covariance structures: no within-group covariance, compound symmetric, or first-order autoregressive, selecting the most appropriate covariance structure based on lowest AIC values. When necessary, we transformed response variables to meet assumptions. We used the appropriate distribution and link function for each response variable; we used a binomial distribution and logit link to analyze differences in presence and a Poisson distribution and log link to analyze differences in abundance. We used a quasi-likelihood method to test for overdispersion in the Poisson model when necessary (Ramsey and Schafer 2002; Zuur et al. 2009). All analyses were completed using the lme4, MASS, and nlme packages in R (Bates et al. 2014; Pinheiro et al. 2013; R Core Development Team 2013; Venables and Ripley 2002).

We analyzed differences in bare ground and litter cover in 2011, as more than half of all values were zero in 2012 and 2013; we made informal comparisons in 2012 and 2013 based on means and 95% confidence intervals. We did not examine differences in woody plant cover, as woody plants were < 1% of all individual plants sampled during the three-year study (Appendix C). We examined changes in presence for species that occurred in 25 – 80% (24 – 64) of 80 total plot samples (i.e., 10 plots * 8 sampling periods), and changes in abundance for species that occurred in at least 50% of total plot samples. Therefore, we analyzed presence of 19 taxa (including four ants, nine decomposers, and six predator species), and abundance of seven taxa (including two ants, three decomposers, one herbivore, and one predator species). We explored

only simple effects of plant community and year in models for presence of arthropod species due to issues with convergence.

Results

Precipitation

Total rainfall for the water year (October 1—September 30) measured 32.3 cm for 2011, 62.5 cm for 2012, and 69.1 cm for 2013, which was 36%, 69%, and 76% of the long-term average (90.2 cm), respectively. Most precipitation did not occur during our sampling periods (Fig. 2.1). We observed the most precipitation between field seasons in 2013 (2011 = 1.1 cm, 2012 = 12.0 cm, 2013 = 26.0 cm). We categorized magnitude of drought (based on PDSI) during each year of the study as extreme (<-4.00), moderate (-3.99 to -3.00), and none (-1.99 to 1.99) for 2011, 2012, and 2013, respectively (NCDC-NOAA 2014).

Vegetation

We identified a total of 24 plant species in the native plant community, including six species of grasses, one sedge, 14 forbs, and three woody plants (Appendix C). We identified 17 plant species in the Kleberg bluestem community, which included three species of grasses, 12 forbs, and two woody plants (Appendix C). Dominant species differed by plant community; Kleberg bluestem represented 94% of all individuals sampled in the community, whereas seacoast sumpweed (*Iva annua*), Hall's panicum (*Panicum halli* var. *halli*), tickseed (*Coreopsis tinctoria*), and western ragweed (*Ambrosia*

cumanensis) collectively represented nearly 70% of all individuals sampled in the native plant community (Appendix C).

Nearly all of the characteristics of vegetation composition and structure we measured differed between plant communities and the magnitude of some differences changed over time (Table 2.1). On average, native plant communities had 2.6 more species/m^2 (95% CI = 1.3 – 3.8; Fig. 2.2) than Kleberg bluestem communities. Native plant communities had more forb cover during extreme (9.2%, 4.1 – 21.92), moderate (94.3, 36.7 – 99.57), and non-drought conditions (16.3, 6.4 – 42.0), relative to Kleberg bluestem communities (Fig. 2.3). Plant density was similar in both plant communities during extreme and moderate drought, but lower (34.1 plants/m^2, 27.9 – 40.2) in native plant communities than Kleberg bluestem when drought subsided (Fig. 2.2). Plants in native plant communities also were shorter during extreme drought (26.2 cm, 25.6 – 26.4) and when drought subsided (33.8, 30.2 – 37.5; Fig. 2.2). Litter cover did not differ between plant communities during extreme drought, but generally was higher in native plant communities as drought severity decreased, relative to Kleberg bluestem communities (Fig. 2.3). Kleberg bluestem communities had more grass cover (74.1%, 37.1 – 89.2) relative to native plant communities during moderate drought (Fig. 2.3); all grass cover (100%) in Kleberg bluestem communities was comprised of nonnative grasses (Appendix C).

Soils

All of the soil characteristics measured differed between plant communities and the magnitude of some differences changed over time (Table 2.2). Soils in native plant communities generally were more acidic (1.9 units pH, 95% CI = 1.7 – 2.2) and had 1.6% more organic matter (0.1 – 3.9) than soils in Kleberg bluestem communities (Fig. 2.4). Soils in the native plant community also had more available nitrogen (NO_3) and phosphorus (P_2O_5) relative to Kleberg bluestem communities, but the differences depended on drought severity (Table 2.2). Differences in available nitrogen and phosphorus between communities decreased over time, as drought severity decreased (Fig. 2.4); available phosphorus differed little between communities when drought conditions subsided (Fig. 2.4).

Arthropods

We captured a total of 14,181 arthropods (*n* = 6,975 for Kleberg, *n* = 7,206 for Native), representing 30 orders, 157 families, and 271 species (Appendix B). Arthropod communities in native plant communities were comprised mainly of woodlice (*Armadillidium vulgare*), which represented 47.5% of all individuals captured (Table 2.3), whereas Kleberg bluestem communities mainly consisted of Mochlozetid mites (32.1%) and leafhoppers (*Balclutha rubrostriata*, 26.2%). We were unable to analyze pollinator arthropods from the study, as pollinators represented <1% of all arthropods collected (Appendix B).

Native plant communities generally had more species of arthropods than Kleberg bluestem communities, but the differences were most pronounced during extreme drought (8.0 species/m^2, 95% CI = 5.3 – 11.6, Table 2.4; Fig. 2.5). During extreme drought, native plant communities also had 131.1 more arthropods/m^2 (88.0 – 205.9) than Kleberg bluestem communities, but patterns shifted as drought severity decreased (Table 2.4; Fig. 2.6). Native plant communities had 36.3 fewer arthropods (33.6 – 50.0) during moderate drought and 46.4 fewer arthropods (44.1 – 48.5) when drought subsided (Fig. 2.6), relative to Kleberg bluestem communities.

Herbivores

Native plant communities had 0.8 more species/m^2 of herbivores (0.5 – 4.3), but herbivores were much more abundant in Kleberg bluestem communities, especially as drought severity decreased (Table 2.4; Figs. 2.5 and 2.6). Herbivores comprised 61% (4,234 arthropods) of total abundance in Kleberg bluestem communities, in comparison to 12% (872) in native plant communities (Appendix B). Mochlozetid mites represented 53% (2,238) of all herbivorous arthropods in Kleberg bluestem communities, compared to 13% (111) in native plant communities (Appendix B). A species of leafhopper (*Balclutha rubrostriata*) represented 43% (1,830) of all herbivorous arthropods in Kleberg bluestem communities, and was not collected in native plant communities (Appendix B); both *B. rubrostriata* and mites were the main contributors to differences in herbivore abundance between communities (Tables 2.5 and 2.6; Fig. 2.6).

Decomposers

We did not detect differences in species richness of decomposer arthropods between plant communities, but abundance differed (Table 2.4; Figs. 2.5 and 2.6). Native plant communities had more decomposers during extreme drought (126.7, 90.8 – 176.7) and when drought conditions subsided (127.4, 103.5 – 156.9), relative to Kleberg bluestem communities (Fig. 2.6). Decomposer arthropods comprised 61% (4,384 arthropods) of total abundance in the native plant community, in comparison to 16% (1,093) in Kleberg bluestem communities (Appendix B). Pillbugs (*Armadillidium vulgare*) represented 71% (3,420) of decomposer arthropods in native plant communities, compared to 10% (110) in Kleberg bluestem communities (Appendix B); pillbugs were the main contributors to differences in abundance between communities (Table 2.6; Fig. 2.6). Of the nine decomposer taxa studied, three were observed more frequently in Kleberg bluestem communities (e.g., *Blattella vaga*), two were observed more frequently in native plant communities (e.g., *Melanophthalma* spp.), and four did not respond to changes in plant community (Tables 2.7 and 2.8).

Predators

Species richness and abundance of predaceous arthropods differed between plant communities and over time, but the patterns were complex (Table 2.4; Figs. 2.5 and 2.6). During extreme drought, native plant communities had 5.3 more species (3.3 – 8.4) of predaceous arthropods relative to Kleberg bluestem communities, but richness did not differ as drought severity decreased (Fig. 2.5). Predators were more abundant

(12.0, 8.0 – 17.9) in native plant communities during extreme drought, but became more abundant in Kleberg bluestem communities (22.1, 19.2 – 25.3) when drought subsided (Fig. 2.6).

Of the six predator species studied, only two taxa were observed more frequently in native plant communities (*Eumicrosoma* spp. and *Vonones* spp.); plant community did not affect the presence of the other four taxa (Tables 2.7 and 2.8). Three taxa were observed more frequently as drought conditions decreased (e.g., *Haplothrips* spp.), whereas the presence of the other three species did not change (Table 2.8). Anystid mites were common in both plant communities; these mites represented 29% (365 arthropods) of all predators collected in Kleberg bluestem communities and 22% (190) in native plant communities (Appendix B). Although the abundance of Anystid mites did not differ between plant communities during extreme drought, Kleberg bluestem communities had more mites than native plant communities during moderate drought (7.4, 5.8 – 9.3) and when drought subsided (5.3, 4.5 – 6.4; Table 2.6).

Ants

Native plant communities had 0.6 more ant species/m^2 (0.5 – 0.7) relative to Kleberg bluestem communities (Table 2.4; Fig. 2.5). Ant abundance did not differ between plant communities during extreme drought, but native plant communities had 7.3 more ants (4.7-11.2) under moderate drought conditions and 14.1 more ants (9.8-20.2) when drought subsided, relative to Kleberg bluestem communities (Table 2.4; Fig. 2.6). Fire ants (*Solenopsis* spp.) represented 71.7% (745 arthropods) of ant abundance in

native plant communities and were the overall contributors to differences in ant abundance between plant communities (Appendix B). In contrast, fire ants represented 60% (233) of ant abundance in Kleberg bluestem communities (Appendix B).

Of the four ant species studied, one species was observed more frequently in native plant communities (*Solenopsis geminata*), one species in Kleberg bluestem communities (*Tapinoma sessile*), and presence of two species did not respond to changes in plant communities (*Forelius pruinosus* and *Solenopsis invicta*; Tables 2.7 and 2.8). The presence of two species increased in plant communities as drought severity decreased (*F. pruinosus* and *S. geminata*), whereas the other two species did not change (Tables 2.7 and 2.8). Native plant communities had more native fire ants (*S. geminata*, 12.2 ants/m^2, 3.3 – 44.9) and invasive fire ants (*S. invicta*, 5.1, 3.0 – 8.5) than Kleberg bluestem communities as drought subsided (Tables 2.5 and 2.6).

Discussion

Both plant invasion and drought can alter composition and abundance of arthropod communities (Alerding and Hunter 2013; Buchholz et al. 2013; Burghardt and Tallamy 2013; Cord 2011; Frampton et al. 2000; Fortuna et al. 2013; Graves and Shapiro 2003; Pearson 2009; Tallamy et al. 2010; Wolkovich 2010), and the combination of these multiple stressors can have novel effects on communities (Paine et al. 1998; Turner 2010; Vitousek et al. 1996). The effects of drought on native plant and arthropod communities can supersede the effects of plant invasion. Although we observed fewer

species of plants in both communities during extreme drought, the magnitude of difference in plant diversity between these two communities changed as drought severity decreased. We observed changes in the composition of arthropod functional groups that were associated with plant invasion, but these differences were not apparent during extreme drought. The direction and magnitude of the response by arthropods to plant invasion and drought often were driven by specific, dominant species.

Herbivores

Plant-feeding arthropods generally decrease in diversity and abundance with increased dominance of invasive plants (Litt et al., in press) because arthropods have not evolved with these novel plants (Bernays and Graham 1988; Burghardt et al. 2010; Niemala and Mattson 1996; Tallamy 2004). Specialist herbivores, such as true bugs (Hemiptera), butterflies (Lepidoptera), thrips (Thysanoptera), and some beetles (Triplehorn and Johnson 2005) may be affected most negatively. Like Cord (2011), we found that communities of herbivorous arthropods were less diverse in communities of OWBs, but we also observed a greater abundance of herbivores in monocultures of Kleberg bluestem during moderate and non-drought conditions. Herbivorous arthropods provide an important food resource for many grassland birds (Wiens and Rotenberry 1979) and changes in the abundance or composition of herbivorous arthropods following plant invasion may alter habitat quality for species representing other trophic levels.

Woodin et al. (2010) reported plant-feeding arthropods were abundant in OWB grasses, specifically leafhoppers (Cicadellidae), stink bugs (Pentatomidae), and seed bugs (Lygaeidae). We found that a single species of leafhopper (*Balclutha rubrostriata*) represented 26% of all arthropods collected in Kleberg bluestem communities (Appendix B), which is comparable to Woodin et al. (2010). *Balclutha rubrostriata* is an invasive species whose native range overlaps with OWB grasses and has been associated with OWBs in its introduced range (Morgan et al. 2013; Zahniser et al. 2010). We collected nearly all *B. rubrostriata* (~99%) when Kleberg bluestem grasses were flowering; *B. rubrostriata* may use flowering Kleberg bluestem as a food source in its introduced range. Based on our data, if invasive herbivores can utilize the invasive plant, herbivores can be more abundant where an invasive plant species is dominant than in native plant communities (Tallamy 2004; Tallamy et al. 2010). Mochlozetid mites also were substantially more abundant in Kleberg bluestem communities relative to native plant communities. Both *B. rubrostriata* and Mochlozetid mites represented 58% of all arthropods collected in Kleberg bluestem communities and changes in the composition and abundance of arthropods observed in OWB monocultures were driven by these herbivores.

Decomposers

Detritus and fungal-feeding arthropods may benefit from increases in plant litter and decomposition from microbes associated with plant invasion (Gratton and Denno 2006; Kappes et al. 2007; Levin et al. 2006; Litt et al., in press; Wolkovich 2010). In

addition, litter from invasive plants may have different chemical properties that may alter soil conditions and benefit certain arthropod taxa (Alerding and Hunter 2013; Standish 2004). We found abundant litter in both plant communities during extreme drought, but soils in native plant communities had more available nitrogen and phosphorus that may increase the nutritional content of litter for decomposers.

We found that the differences in abundance of decomposers between plant communities were driven by pillbugs (*Armadillidium vulgare*). *Armadillidium vulgare* is a detritivore that can increase rates of decomposition and mineralization in soil communities, but also is an invasive species from Europe that can replace native detritivores (David and Handa 2010; Ellis et al. 2000; Frouz et al. 2008; Singer et al. 2012). *A. vulgare* is sensitive to changes in soil pH and prefers near-neutral soils (van Straalen and Verhoef 1997; Zimmer et al. 2000), despite being collected in acidic soils found in our native plant community. *A. vulgare* was nearly three times more abundant in communities of native plants relative to Kleberg bluestems during extreme drought, despite the same amount of litter cover. Plant tissues of OWBs have high C:N ratios (Reed et al. 2005) and litter may be less palatable for detritivores, which suggests that although *A. vulgare* selects both plant communities as habitat, quantity and composition of litter has a greater influence on habitat quality than soil properties. Increased abundance of pillbugs in native plant communities also may increase habitat quality for native arthropods and other wildlife that forage for *A. vulgare* in the litter layer (Fisher and Cover 2007; Paris 1963; Řezáč and Pekár 2007). Therefore, changes in

litter composition and abundance following plant invasion and drought may affect other trophic levels.

Predators

Changes in vegetation structure from plant invasion may affect predatory arthropods indirectly (Gratton and Denno 2006; Pearson 2009). For example, predators that depend on vegetation structure for prey capture (e.g., web-building spiders) may increase with increased plant density or height (Pearson 2009). Increased plant cover or density also may impede movement for predators or serve as refugia for prey, making capture of prey species more difficult (Crist et al. 2006; Samways et al. 1996; Wolkovich et al. 2009; Wu et al. 2009). Given that predatory arthropods vary in habitat and prey preferences, responses to plant invasion may be species-specific (Litt et al., in press). We documented increased plant density and cover in Kleberg bluestem communities relative to native plant communities, comparable to Cord (2011) and Woodin et al. (2010), which may impede movement for predaceous arthropods.

Changes in vegetative characteristics following plant invasion may not only affect the ability of predators to locate and capture prey, but also may affect the diversity and availability of prey (Gratton and Denno 2006; Hansen et al. 2009; Simao et al. 2010). We suspect this might explain why some parasitoids (e.g., *Eumicrosoma* spp.) were observed more frequently in native plant communities than in Kleberg bluestem communities. If plant invasions reduce diversity of herbivorous arthropods, invaded plant communities may provide habitat only for specialist predators (Gratton and Denno

2006; Hansen et al. 2009). Although most thrips (Thysanoptera) are herbivorous, some species are predaceous (Triplehorn and Johnson 2005). We sampled two groups of thrips (*Haplothrips* and *Scolothrips* spp.) more frequently as drought severity decreased, which may have resulted from the concurrent increase in abundance of mite prey (Triplehorn and Johnson 2005). Anystid mites also prey on other mites (Krantz and Walter 2009) and the increase in abundance of predatory mites in Kleberg bluestem communities also may be due the abundance of other mites.

Ants

Because ants play many ecological roles, these species may show diverse responses to plant invasions (Litt et al., in press; Wolkovich et al. 2009). Most of the ants we collected are omnivorous (Buczowski 2010; Ness 2003; Rudgers et al. 2003; Taber 2000; Tennant 1991) and changes in plant composition resulting from invasion may not be an important factor for determining habitat. For example, *Tapinoma sessile*, one of the most widely-distributed species of ants in North America (Fisher and Cover 2007), can tolerate a variety of environmental conditions and stressors (Buczowski 2010), and may explain why *T. sessile* was present in both plant communities, although observed more frequently in Kleberg bluestem communities. Although densities of red-imported fire ants (*Solenopsis invicta*) may be lower in OWBs (Sternberg et al. 2006), omnivory may explain why *S. invicta* did not respond to changes associated with plant community or drought in our study. Decreased drought severity resulted in increased richness of both plants and arthropods, which may have affected food availability for ants. Both

Forelius pruinosus and native fire ants (*S. geminata*) were observed more frequently as drought subsided, and changes in the composition of the ant community may be more influenced by drought than by invasive plants.

Conclusions

We present evidence that plant invasion and drought interact to alter richness and composition of vegetation communities, with concomitant interactive effects for arthropods. We found that differences in vegetation characteristics between native plant and Kleberg bluestem communities were more pronounced as drought conditions subsided, suggesting that effects of drought may supersede effects of plant invasion. However, differences in arthropod communities between native and invasive plant communities depended on changes in functional groups as drought severity decreased. Arthropod communities shifted from being dominated by detritivores in the native plant community to dominated by herbivores in the Kleberg bluestem community, contrary to general patterns reviewed in Gratton and Denno (2006), van Hengstum et al. (2014), and Litt et al. (in press). Herbivorous arthropods increased as drought severity decreased in Kleberg bluestem communities, whereas detritivores fluctuated over time due to changes in litter cover. However, both plant communities were dominated by nonnative arthropods and invasions at multiple trophic levels (i.e., plants and arthropods) may have profound consequences for biodiversity. Arthropod communities provide important ecosystem services, such as pollination, decomposition, and seed dispersal (Brussaard 1997; Folgarait 1999; Price et al. 2011; Triplehorn and Johnson

2005; Wilson 1987), and changes in arthropod communities associated with multiple invasions or disturbances could alter the integrity of these ecosystem services. Arthropods also provide a substantial food resource for other trophic levels, and shifts in the composition of the arthropod community as a result of multiple disturbances will likely have negative consequences on native grassland fauna (Doxon et al. 2011; Frampton et al. 2000; Hickman et al. 2006; Wilson 1987). Understanding how changes in plant and arthropod communities in association with invasion and other disturbances alter other trophic levels may help inform conservation practices where these disturbances are present.

Tables

Table 2.1. Factors affecting vegetation characteristics with plant invasion and drought, southern Texas, summers 2011-2013.

Vegetation Variable	Community		Year		Community*Year	
	$F_{1,8}$	*P*	$F_{2,68}$	*P*	$F_{2,66}$	*P*
Richness	94.76	<0.001	16.09	<0.001		
Plant Density	6.72	0.032	12.59	<0.001	6.85	0.002
Plant Height	74.63	<0.001	16.29	<0.001	15.59	<0.001
Bare Ground[a]	15.27	0.005				
Litter[a]	0.76	0.410				
Grass	21.46	0.002	9.86	<0.001	12.79	<0.001
Forb	300.16	<0.001	14.29	<0.001		

[a] 2011 data only

Table 2.2. Factors affecting soil characteristics with plant invasion and drought, southern Texas, 2011-2013.

Soil Variable	Community		Year		Community*Year	
	$F_{1,7}$	*P*	$F_{2,9}$	*P*	$F_{2,7}$	*P*
pH	361.75	<0.001	11.25	0.004		
% O.M.	37.16	<0.001	1.82	0.216		
NO_3	57.89	<0.001	12.39	0.005	20.97	0.001
P_2O_5	36.05	<0.001	40.12	<0.001	3.82	0.076

Table 2.3. Five most common arthropod species collected in native plant and Kleberg bluestem communities (*n* = 300 samples), southern Texas, summers 2011-2013.

Community	Order	Family	Species	% of individuals
Native	Oniscidea	Armadillidiidae	*Armadillidium vulgare*	47.5
	Collembola	Entomobryidae	*Entomobrya* spp.	5.3
	Sarcoptiformes	Euphthiracaridae		5.1
	Hymenoptera	Formicidae	*Solenopsis geminata*	5.4
	Hymenoptera	Formicidae	*Solenopsis invicta*	4.9
Kleberg	Sarcoptiformes	Mochlozetidae		32.1
	Hemiptera	Cicadellidae	*Balclutha rubrostriata*	26.2
	Trombidiformes	Anystidae		5.2
	Blattodea	Blattellidae	*Blattella vaga*	3.0
	Hymenoptera	Formicidae	*Solenopsis invicta*	2.6

Table 2.4. Factors affecting arthropod characteristics with plant invasion and drought, southern Texas, summers 2011-2013.

Arthropod Variable		Community		Year		Community*Year	
		$F_{1,8}$	*P*	$F_{2,66}$	*P*	$F_{2,68}$	*P*
Richness							
	Total	23.05	0.001	33.90	<0.001	4.30	0.018
	Herbivores	7.43	0.026	18.13	<0.001		
	Decomposers	0.06	0.817	11.45	<0.001		
	Predators	29.65	<0.001	12.89	<0.001	9.03	<0.001
	Ants	6.28	0.037	3.40	0.039		
Abundance							
	Total	0.39	0.645	788.06	<0.001	348.60	<0.001
	Herbivores	51.63	<0.001	646.84	<0.001	94.51	<0.001
	Decomposers	106.43	<0.001	169.45	<0.001	332.72	<0.001
	Predators	9.65	0.013	55.76	<0.001	46.94	<0.001
	Ants	23.98	<0.001	179.56	<0.001	36.29	<0.001

Table 2.5. Factors affecting abundance in arthropod species with plant invasion and drought, southern Texas, summers 2011-2013.

				Community		Year		Community*Year	
Functional Group	**Order**	**Family**	**Species**	$F_{1,8}$	*P*	$F_{2,68}$	*P*	$F_{2,66}$	*P*
Herbivores	Sarcoptiformes	Mochlozetidae		146.46	<0.001	174.00	<0.001	116.53	<0.001
Decomposers	Collembola	Entomobryidae	*Entomobrya* spp.	10.98	0.011	7.67	0.001	40.31	<0.001
	Oniscidea	Armadillidiidae	*Armadillidium vulgare*	313.72	<0.001	430.38	<0.001	54.62	<0.001
	Orthoptera	Gryllidae	*Gryllus* spp.	5.10	0.054	8.79	<0.001	4.45	0.015
Predators	Trombidiformes	Anystidae		12.08	0.008	2.25	0.113	29.86	<0.001
Ants	Hymenoptera	Formicidae	*Solenopsis geminata*	26.90	<0.001	92.19	<0.001	6.66	0.002
			Solenopsis invicta	3.11	0.116	81.42	<0.001	6.50	0.003

Table 2.6. Abundance of arthropod species (means and 95% CIs) in Kleberg bluestem and native plant communities during drought.

Functional Group	Order	Family	Species	Community	Abundance (arthropods/m^2) 2011	2012	2013
Herbivores	Sarcoptiformes	Mochlozetidae		Kleberg	1.9 (1.1-3.3)	41.1 (27.5-61.2)	84.3 (56.7-125.3)
				Native	3.2 (1.6-6.5)	1.8 (1.0-3.3)	1.0 (0.5-1.9)
Decomposers	Collembola	Entomobryidae	*Entomobrya* spp.	Kleberg	0.5 (0.2-1.2)	5.4 (2.5-11.8)	1.4 (0.6-3.3)
				Native	13.6 (5.5-33.6)	6.1 (2.7-13.9)	8.0 (3.3-19.3)
	Oniscidea	Armadillidiidae	*Armadillidium vulgare*	Kleberg	4.2 (2.9-6.2)	3.2 (2.2-4.9)	0.5 (0.2-1.1)
				Native	119.3 (75.7-188.1)	20.6 (13.5-31.6)	119.2 (55.3-256.7)
	Orthoptera	Gryllidae	*Gryllus* spp.	Kleberg	1.2 (0.6-2.4)	0.6 (0.3-1.5)	0.7 (0.3-1.6)
				Native	2.8 (1.2-6.6)	0.1 (0.0-0.6)	1.1 (0.4-3.2)
Predators	Trombidiformes	Anystidae		Kleberg	5.9 (4.4-8.1)	9.3 (7.0-12.5)	9.2 (7.0-12.4)
				Native	8.5 (5.6-13.0)	1.9 (1.2-3.2)	3.9 (2.5-6.0)
Ants	Hymenoptera	Formicidae	*Solenopsis geminata*	Kleberg	0.3 (0.1-0.8)	1.1 (0.4-3.3)	1.8 (0.6-5.0)
				Native	0.4 (0.1-1.7)	3.8 (1.0-14.3)	14.0 (3.9-50.9)
			Solenopsis invicta	Kleberg	3.7 (2.1-6.4)	1.0 (0.6-1.7)	4.5 (3.1-6.4)
				Native	3.5 (1.6-7.5)	1.7 (0.9-3.2)	9.6 (6.1-14.9)

Table 2.7. Factors affecting presence of arthropod species with plant invasion and drought, southern Texas, summers 2011-2013.

Functional Group	Order	Family	Species	Community		Year	
				$F_{1,8}$	*P*	$F_{2,68}$	*P*
Decomposers	Blattodea	Blattellidae	*Blattella vaga*	4.03	0.080	6.53	0.003
	Coleoptera	Anthicidae	*Acanthinus scitulus*	1.40	0.271	3.79	0.028
		Latridiidae	*Melanophthalma* spp.	4.09	0.078	0.00	1.000
	Collembola	Sminthuridae		6.66	0.033	4.70	0.012
	Diptera	Chloropidae	*Liohippelates* spp.	3.75	0.089	1.92	0.154
		Phoridae	*Megaselia* spp.	0.06	0.813	0.05	0.951
	Orthoptera	Gryllidae	*Gryllus* spp.	0.36	0.565	2.61	0.081
	Psocoptera	Liposcellidae	*Liposcelis* spp.	0.00	1.000	0.74	0.481
	Sarcoptiformes	Galumnidae		6.77	0.031	1.52	0.226
Predators	Hymenoptera	Scelionidae	*Eumicrosoma* spp.	4.32	0.071	2.23	0.115
	Opiliones	Cosmetidae	*Vonones* spp.	6.53	0.034	3.69	0.030
	Scorpionida	Buthidae	*Centruroides vittatus*	1.66	0.234	0.97	0.384
	Thysanoptera	Phlaeothripidae	*Haplothrips* spp.	2.16	0.180	6.83	0.002
		Thripidae	*Scolothrips* spp.	0.15	0.709	10.46	<0.001
	Trombidiformes	Erythraeidae		0.02	0.891	8.10	<0.001
Ants	Hymenoptera	Formicidae	*Forelius pruinosus*	0.05	0.829	14.79	<0.001
			Solenopsis geminata	5.24	0.051	3.82	0.027
			Solenopsis invicta	0.05	0.829	1.83	0.168
			Tapinoma sessile	5.51	0.047	1.94	0.151

Table 2.8. Presence of arthropod species (means and 95% CIs) for Kleberg bluestem and native plant communities during periods of drought, southern Texas, summers 2011-2013. When we detected that presence differed among years, we provide separate means and 95% CIs, otherwise we provide only one estimate that represents the mean (and 95% CI) for the entire study.

Functional Group	Order	Family	Arthropod Taxa	Community	Probability of Presence 2011	2012	2013
Decomposers	Blattodea	Blattellidae	*Blattella vaga*	Kleberg	0.97 (0.76-1.00)	0.70 (0.20-0.95)	0.26 (0.03-0.80)
				Native	0.33 (0.06-0.80)	0.04 (0.00-0.27)	0.01 (0.00-0.07)
	Coleoptera	Anthicidae	*Acanthinus scitulus*	Kleberg	0.45 (0.23-0.70)	0.16 (0.05-0.40)	0.14 (0.04-0.40)
				Native	0.65 (0.39-0.84)	0.30 (0.11-0.61)	0.26 (0.09-0.56)
		Latridiidae	*Melanophthalma* spp.	Kleberg	0.30 (0.18-0.46)		
				Native	0.53 (0.31-0.73)		
	Collembola	Sminthuridae		Kleberg	0.09 (0.01-0.42)	0.73 (0.22-0.96)	0.48 (0.09-0.90)
				Native	0.01 (0.00-0.04)	0.27 (0.04-0.78)	0.01 (0.00-0.05)
	Diptera	Chloropidae	*Liohippelates* spp.	Kleberg	0.15 (0.04-0.45)		
				Native	0.05 (0.02-0.14)		
		Phoridae	*Megaselia* spp.	Kleberg	0.31 (0.14-0.57)		
				Native	0.12 (0.05-0.26)		
	Orthoptera	Gryllidae	*Gryllus* spp.	Kleberg	0.73 (0.48-0.89)		
				Native	0.67 (0.45-0.83)		

(Table 2.8 continued)

Functional Group	Order	Family	Arthropod Taxa	Community	Probability of Presence 2011	2012	2013
Decomposers	Psocoptera	Liposcellidae	*Liposcelis* spp.	Kleberg	0.35 (0.16-0.60)		
				Native	0.35 (0.17-0.59)		
	Sarcoptiformes	Galumnidae		Kleberg	0.45 (0.22-0.70)		
				Native	0.15 (0.06-0.34)		
Predators	Hymenoptera	Scelionidae	*Eumicrosoma* spp.	Kleberg	0.36 (0.16-0.61)		
				Native	0.64 (0.40-0.83)		
	Opiliones	Cosmetidae	*Vonones* spp.	Kleberg	0.16 (0.05-0.40)		
				Native	0.54 (0.26-0.80)		
	Scorpionida	Buthidae	*Centruroides vittatus*	Kleberg	0.53 (0.29-0.75)		
				Native	0.67 (0.46-0.84)		
	Thysanoptera	Phlaeothripidae	*Haplothrips* spp.	Kleberg	0.15 (0.04-0.44)	0.68 (0.28-0.92)	0.82 (0.45-0.96)
				Native	0.05 (0.02-0.13)	0.39 (0.11-0.77)	0.58 (0.20-0.89)
		Thripidae	*Scolothrips* spp.	Kleberg	0.07 (0.01-0.38)	0.74 (0.45-0.91)	0.14 (0.02-0.63)
				Native	0.03 (0.01-0.09)	0.52 (0.11-0.91)	0.06 (0.01-0.40)
	Trombidiformes	Erythraeidae		Kleberg	0.05 (0.01-0.28)	0.65 (0.18-0.94)	0.25 (0.04-0.75)
				Native	0.05 (0.02-0.14)	0.68 (0.20-0.95)	0.28 (0.04-0.077)

(Table 2.8 continued)

Functional Group	Order	Family	Arthropod Taxa	Community	Probability of Presence 2011	2012	2013
Ants	Hymenoptera	Formicidae	*Forelius pruinosus*	Kleberg	0.04 (0.00-0.26)	0.08 (0.01-0.47)	0.76 (0.26-0.97)
				Native	0.06 (0.02-0.20)	0.03 (0.00-0.22)	0.84 (0.36-0.98)
			Solenopsis geminata	Kleberg	0.22 (0.07-0.50)	0.52 (0.22-0.81)	0.67 (0.33-0.90)
				Native	0.68 (0.35-0.89)	0.90 (0.68-0.97)	0.94 (0.79-0.99)
			Solenopsis invicta	Kleberg	0.86 (0.61-0.96)		
				Native	0.85 (0.65-0.94)		
			Tapinoma sessile	Kleberg	0.81 (0.54-0.94)		
				Native	0.39 (0.14-0.71)		

Figures

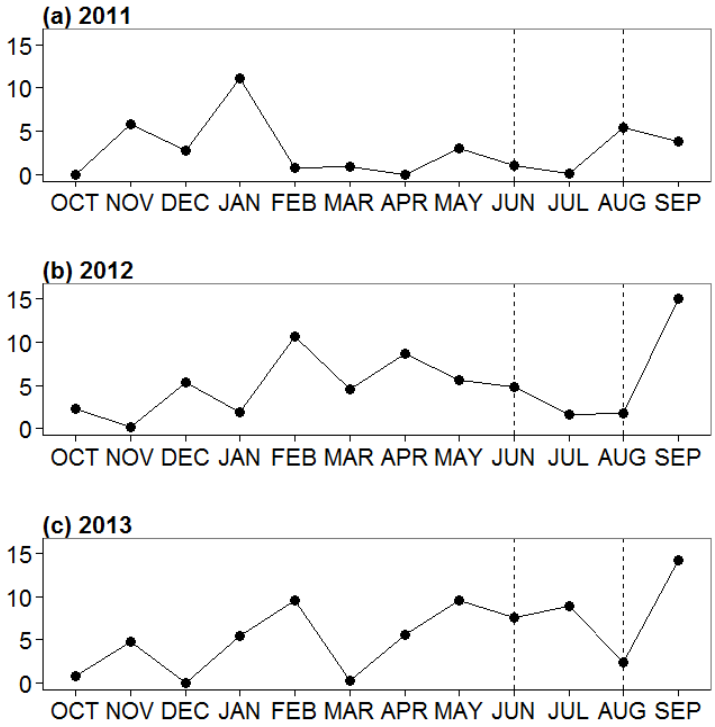

Figure 2.1. Total monthly precipitation for the Welder Wildlife Refuge, starting at the beginning of the water year (Oct 1), southern Texas, 2011-2013. The dashed lines represent the timing of our summer sampling periods.

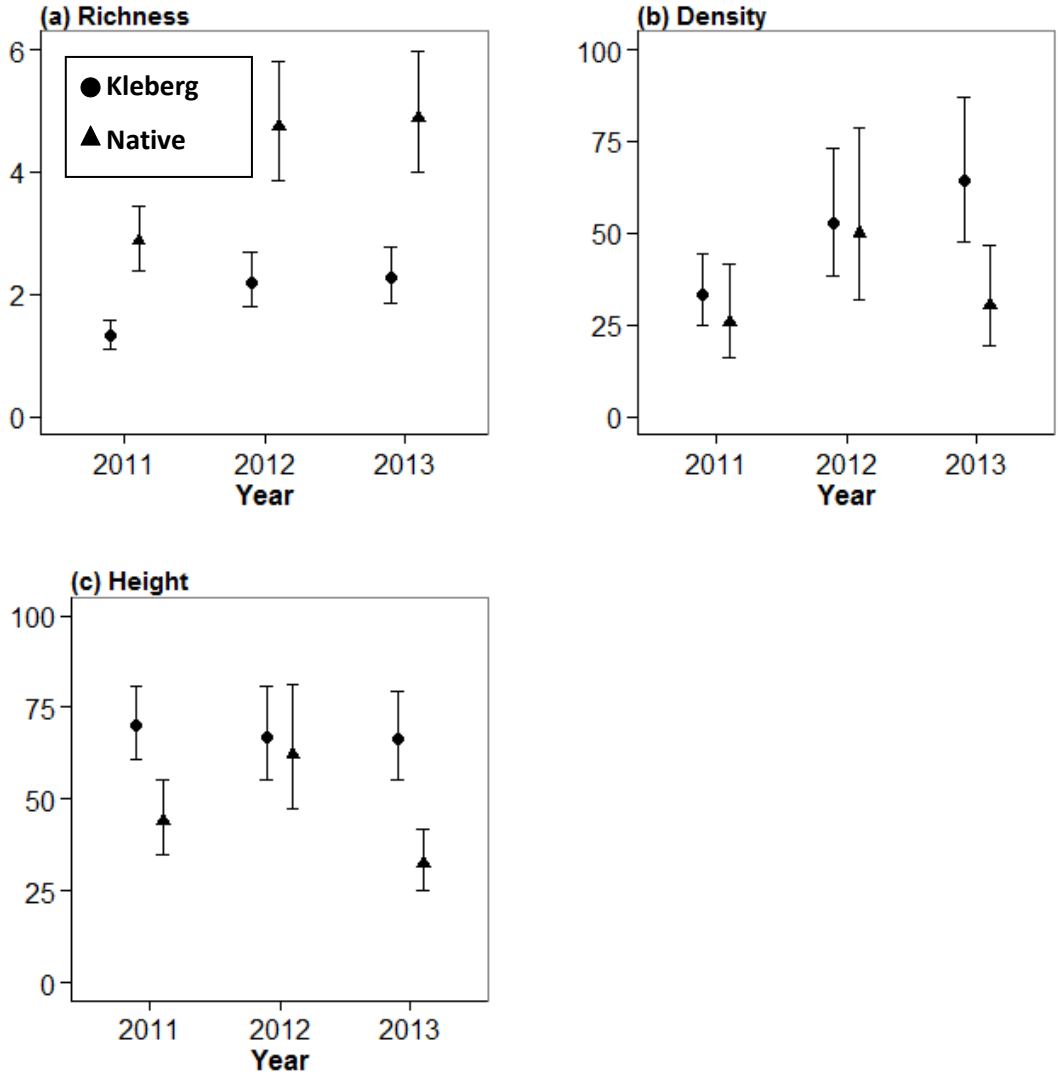

Figure 2.2. Vegetation characteristics (means and 95% CIs) in Kleberg bluestem and native plant communities during years of drought, including (a) species richness (plants/m^2), (b) density (plants/m^2), and (c) maximum height (cm).

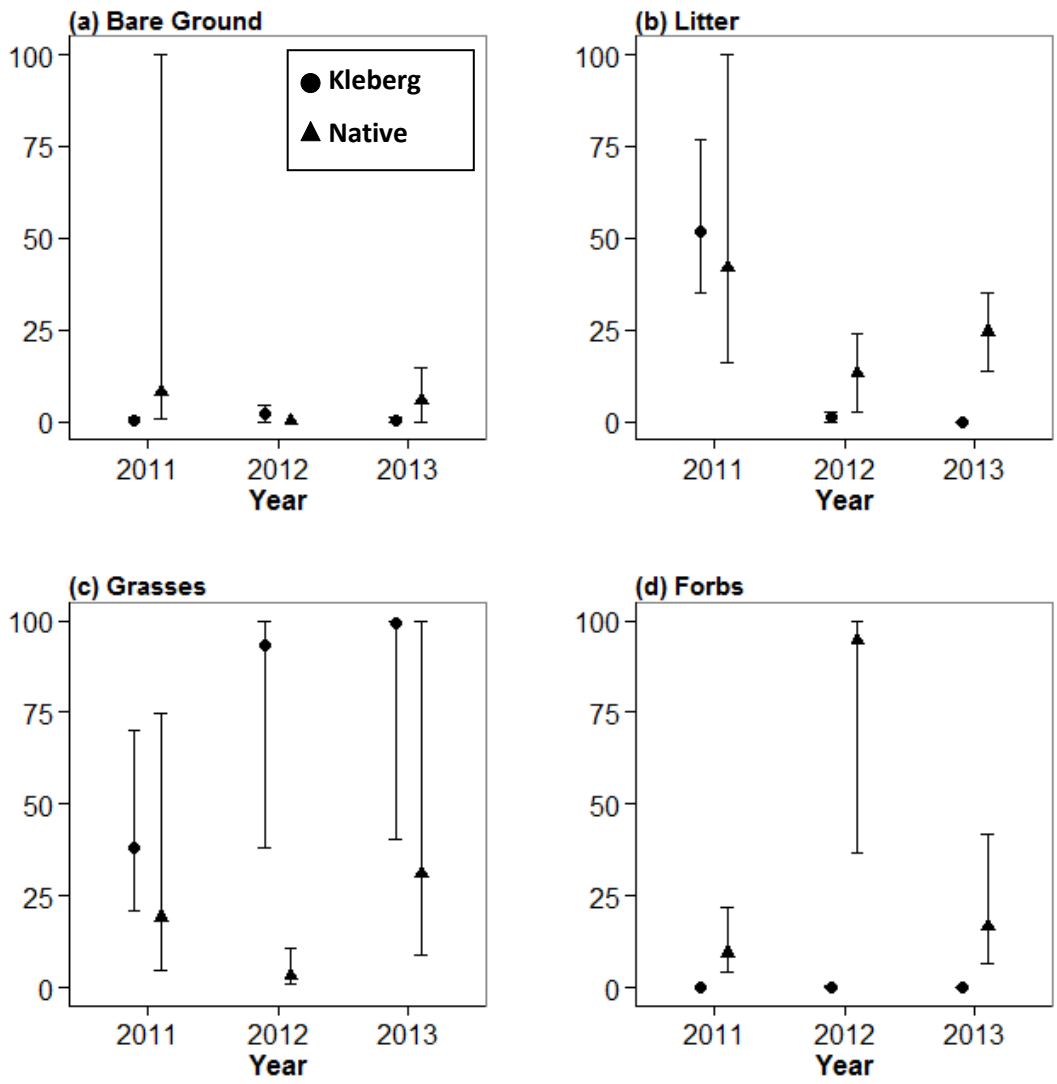

Figure 2.3. Canopy cover by cover class (%, means and 95% CIs) in Kleberg bluestem and native plant communities during years of drought.

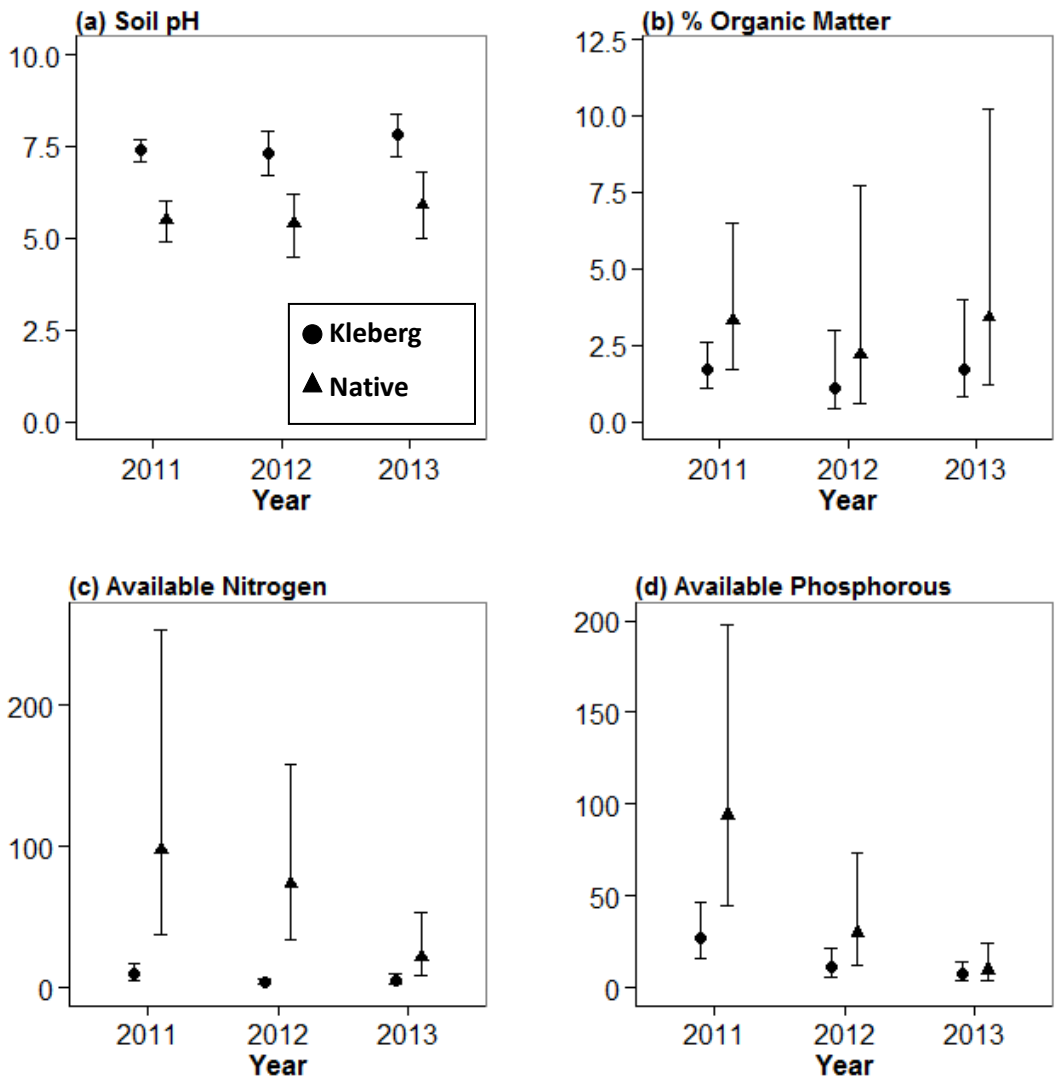

Figure 2.4. Soil characteristics (means and 95% CIs) in Kleberg bluestem and native plant communities during years of drought, including (a) pH, (b) % organic matter, (c) available plant NO_3 (kg/ha), and (d) available plant P_2O_5 (kg/ha).

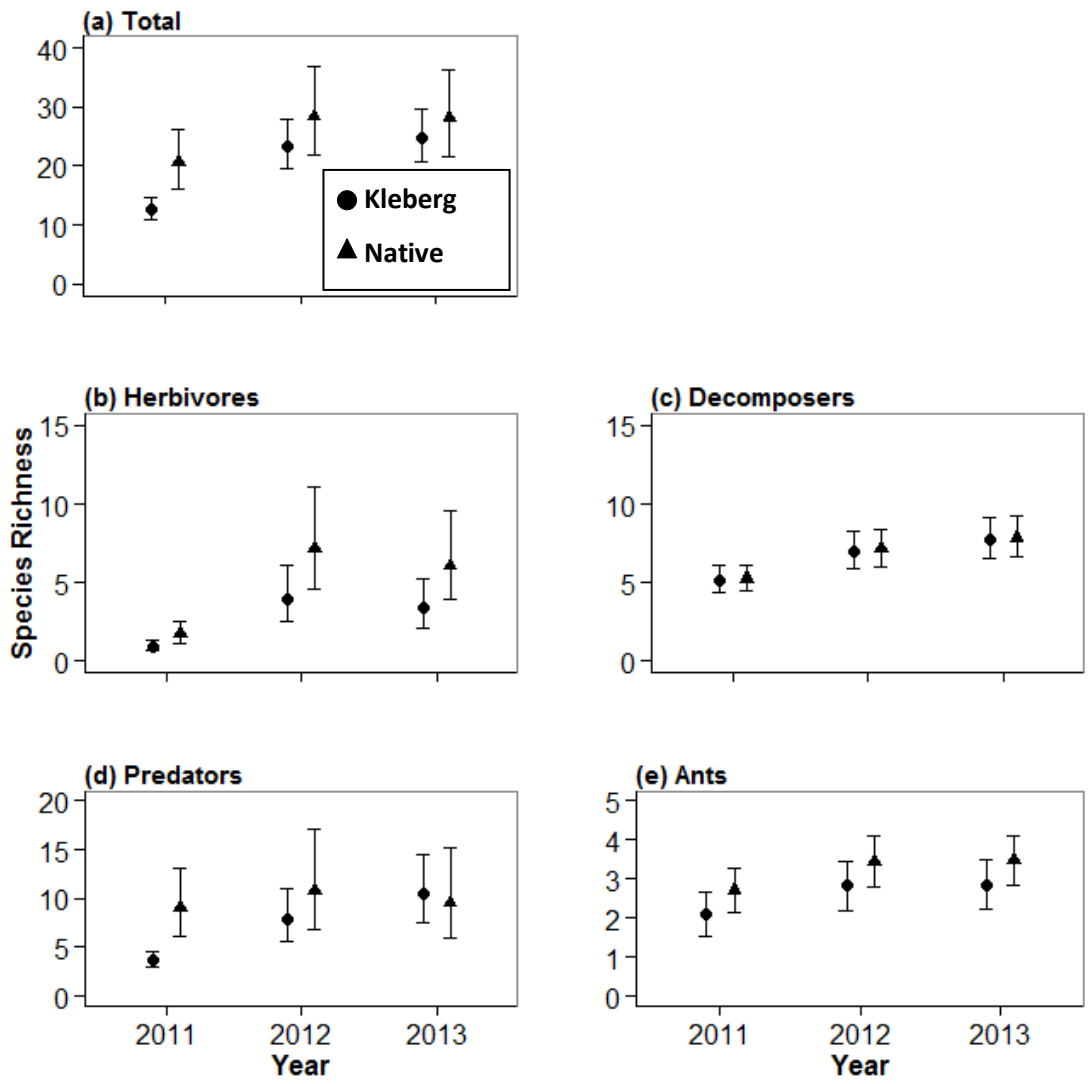

Figure 2.5. Species richness of arthropods (species/m^2, means and 95% CIs) in Kleberg and native plant communities during years of drought.

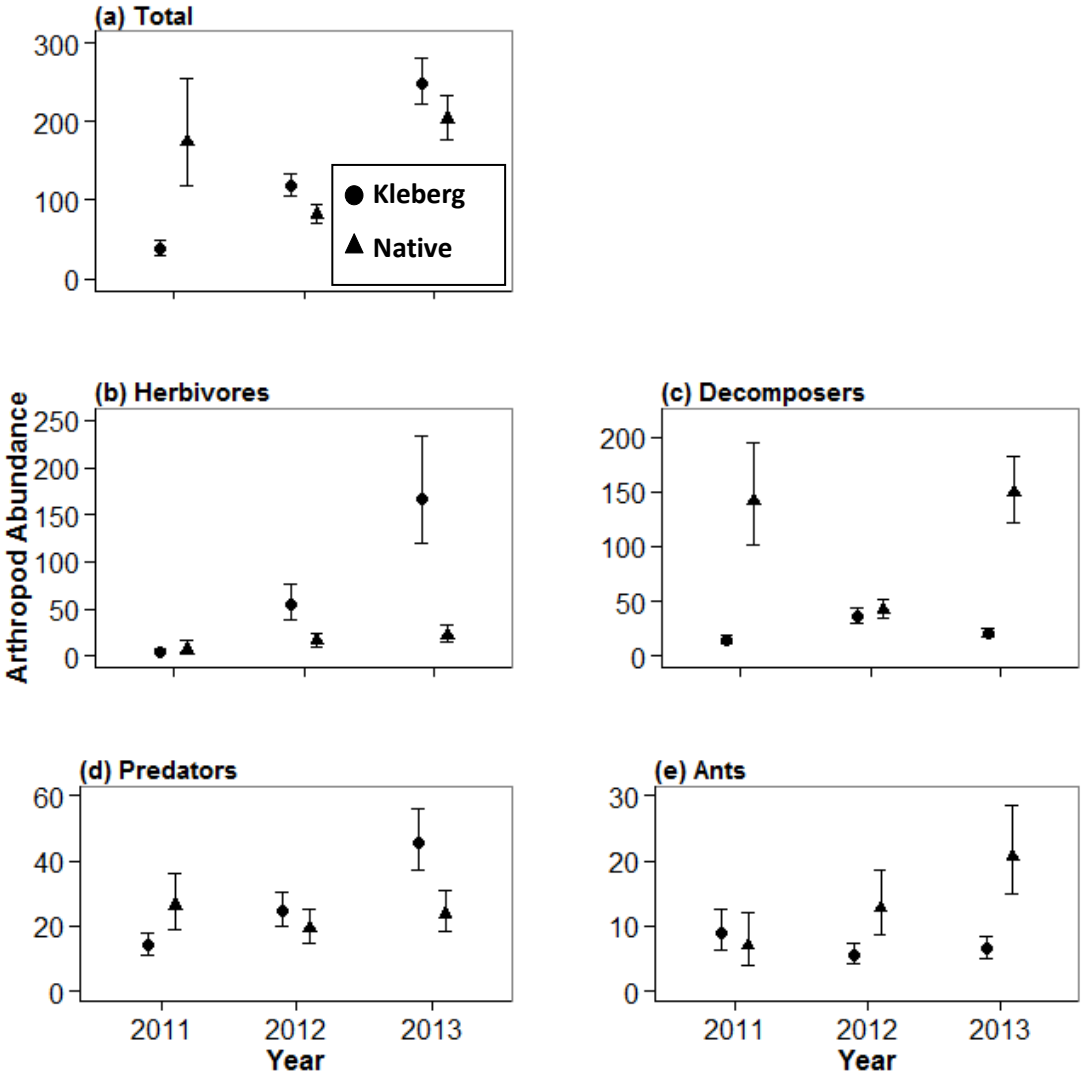

Figure 2.6. Abundance of arthropods (arthropods/m^2, means and 95% CIs) of Kleberg and native plant communities during years of drought.

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CHAPTER THREE

MODIFYING SOIL PROPERTIES TO RESTORE NATIVE PLANT COMMUNITIES FOLLOWING PLANT INVASION AND DROUGHT

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CHAPTER THREE

MODIFYING SOIL PROPERTIES TO RESTORE NATIVE PLANT COMMUNITIES FOLLOWING PLANT INVASION AND DROUGHT

Abstract

Invasive plants can alter native plant communities through changes in soil characteristics, creating feedback loops that may impede restoration. We proposed that altering soil characteristics to favor establishment of native plants may serve as an alternative restoration tool. We conducted a field experiment in areas dominated by Old World bluestems (OWBs, *Dichanthium annulatum*) to investigate the efficacy of soil treatments for reducing dominance of an invasive plant and restoring native plant communities. We conducted our study during an extreme drought that persisted for several years and provided us with the opportunity to examine the interactive effects of drought, plant invasion, and soil treatments. We also assessed the efficacy of soil treatments in the absence of drought using a microcosm under controlled conditions. We applied a total of 10 treatments to 50 plots in June 2011 for the field study, and 100 containers in June 2013 for the microcosm experiment. Treatments included simple soil disturbance, addition of sulfur, lime, carbon, or mycorrhizal fungi, and each of the previous treatments in combination with native seed. We sampled vegetation and soils every month in the summer 2011-2013 for the field study and once in summer 2013 for the microcosm experiment. No vegetation grew in treated plots during 2011 due to drought. Although lime treatments increased soil pH immediately, changes in soil chemistry were either short-lived or absent in both studies and the initial soil chemistry may have inhibited treatments. Seeding with native plants resulted in reduced cover and density of OWBs in both experiments, and increased richness and cover of native plants in the field study. However, we observed very few plants from the native seed mix in plots during the field study; differences in propagule pressure and seed predation between plots with and without seed may have contributed to differences in results. We propose that simple soil disturbance in combination with seeding may help reduce the dominance of OWBs in the short-term. Future research efforts should focus on determining the efficacy of soil treatments for restoring native plant communities in other soil types and monitored effects over longer time periods.

Introduction

Introductions of invasive plant species into native ecosystems have become a global concern (Bryson and Carter 2004; Chornesky and Randall 2003; Reichard et al. 2005; Vitousek et al. 1996). Anthropogenic activities, such as agriculture, grazing, and silviculture, can promote establishment and spread of invasive plants (Archer and Pyke 1991; Chornesky and Randall 2003; Mack and D'Antonio 1997; Hobbs and Huenneke 1992). In many cases, combined stress from anthropogenic effects and competition with invasive plants can result in displacement of native plant species (Levine et al. 2003; Mack and D'Antonio 1997; Paine et al. 1998).

Once dominant in a community, invasive plants can alter ecosystem processes through changes in soil characteristics (Callaway and Ridenour 2004; Corbin and D'Antonio 2004; D'Antonio and Vitousek 1992; Vitousek et al. 1996). For example, changes in the composition of litter may alter decomposition or mineralization rates (Vinton and Goergen 2006; Vitousek 1990). Increased litter cover may increase soil moisture, which could facilitate plant establishment (Wolkovich et al. 2009). Changes in abundance and composition of plant litter also can alter soil pH (Alerding and Hunter 2013; Kappes et al. 2007), which inhibit seed germination. Some species of invasive plants are capable of altering soil properties through allelochemicals, which may inhibit germination and growth of seedlings and colonization of mycorrhizal symbionts (Callaway et al. 2003, 2008; Callaway and Ridenour 2004; Koger and Bryson 2004; Stinson et al. 2006; Wolfe et al. 2008). Some invasive plant species, such as nitrogen

fixers, increase the availability of nutrients and promote establishment of other invasive plants (Alpert and Maron 2000; Werner et al. 2010; Vitousek 1990). Other invasive plants may provide refugia for native soil pathogens (Eppinga et al. 2006; Mangla et al. 2008).

Changes in soil properties resulting from plant invasions can create feedback loops that further promote invasion (Chornesky and Randall 2003; Vinton and Goergen 2006), where traditional management strategies, such as prescribed fire, may no longer function to restore native plant communities (Bryson and Carter 2004; D'Antonio and Vitousek 1992). Modifying biological or chemical properties of the soil could provide alternatives for restoration by creating conditions that favor native plant species. Altering soil pH by adding ash, lime, litter, or sulfur may increase germination or nutrient acquisition of some plants (Elliott et al. 2013; Farrel et al. 2005; Heneghan et al. 2008; Lawson et al. 2004; Longhurst et al. 1999; Owen and Marrs 2000; Tibbett and Diaz 2005). When native plants may tolerate lower nutrient levels than invasive plants and reducing nutrient availability to pre-invaded conditions can shift competitive interactions (Blumenthal et al. 2009). For example, adding a carbon source, such as sugar or wood, can reduce available nitrogen by increasing microorganism abundance, allowing native plants to better compete with invasive plants (Alpert 2010; Blumenthal et al. 2009). Inoculations of mutualistic symbionts also may be necessary to aid in the establishment of native plant communities in invaded landscapes (Archer and Pyke

1991; Biondini et al. 1985; Callaway et al. 2003; 2008; Stinson et al. 2006; Wolfe et al. 2008).

Old World bluestems (OWBs, *Bothriochloa* and *Dichanthium* spp.) are a group of grasses native to Africa, Asia, Eurasia, and Australia (Celarier 1958) that have become dominant in the central and southern Great Plains of the United States (USDA-NRCS 2014). Old World bluestems were introduced to the United States in the early twentieth century as a potential cattle forage, due to advantages in productivity, nutrient acquisition, grazing tolerance, and rapid establishment (Berg 1993; Coyne and Bradford 1985; Dabo et al. 1988; Dewald et al. 1985, Nixon 1949). However, OWBs are of lower forage value than previously thought, as mature plants are less palatable than native range plants (Berg and Sims 1995; Dewald et al. 1985; Dabo et al. 1988; Gillen and Berg 2001). OWBs were and still are planted by the USDA Conservation Reserve Program throughout Oklahoma and Texas to revegetate marginal farmland (Nixon 1949; Schmidt et al. 2008) and reduce soil erosion on reclamation sites and highways (Berg 1993; Harmoney et al. 2004), contributing to its spread.

OWBs grow well under disturbed conditions, respond well to nitrogen fertilizers, and allocate resources rapidly to foliar growth when stressed by grazing (Berg 1993; Coyne and Bradford 1985; Schmidt and Hickman 2006). Once established, these grasses may alter fire regimes, nutrient cycling, and soil chemistry (Dirvi and Hussain 1979; Reed et al. 2005), as well as the composition of native plant and animal communities (Cord

2011; Gabbard and Fowler 2007; Hickman et al. 2006; Sammon and Wilkins 2005; Schmidt et al. 2008; Woodin et al. 2010).

Traditional management strategies, such as fire and herbicides, have not reduced OWB populations successfully. Prescribed fire alone results in variable and marginal reductions in OWB productivity (Berg 1993; Ruckman et al. 2011; Simmons et al. 2007; Twidwell et al. 2012). Applications of herbicides may reduce dominance of OWBs initially, but populations generally recover within one year (Harmoney et al. 2004, 2007; Mittelhauser et al. 2011; Ruffner and Barnes 2012). Additionally, commonly-used herbicides are broad-spectrum, which may be problematic when attempting to restore native plant communities (Harmoney et al. 2007; Ruckman et al. 2011; Ruffner and Barnes 2012).

Plant invasions typically are only one of many stressors influencing native plant communities in a given landscape, and multiple stressors can interact to have novel effects (Darling and Cote 2008; Paine et al. 1998; Turner 2010). Drought events, for example, may exacerbate stress on native plant communities and invasive plant species may become dominant after drought events subside (Boulant et al. 2008; Castillo et al. 2007; Crous et al. 2012; Everard et al. 2010; Larios et al. 2013; Miller 1994; Schumacher et al. 2008). OWBs are drought-tolerant (White and Dewald 1996), which could increase resilience to restoration strategies. For example, severe drought could reduce soil moisture, which may impede the efficacy of herbicides (Harmoney et al. 2004). Soil

modification techniques have not been examined for OWBs, and may prove as effective tools on restoring native plant communities in the presence and absence of drought.

We developed a field-based experiment to test the efficacy of soil modification techniques for reducing dominance of OWBs (field study). We predicted that altering soil conditions would increase abundance and species richness of native plants and reduce abundance of OWBs, and we expected these changes to persist several years after treatment. In 2011, a severe drought event occurred throughout the introduced distribution of OWBs (NDMC-UNL 2014), which persisted for several years in the southern portion of this range. Although this drought event provided us with the opportunity to test the efficacy of soil modification technique under more extreme conditions in our field experiment, we initiated an additional experiment to determine the efficacy of soil modification in the absence of drought (microcosm experiment). We predicted that we would observe more species of native plants and reduced dominance of OWBs following soil modification treatments in the absence of drought in comparison to soil modification treatments under drought conditions.

Methods

Study Area

We conducted our research at the Welder Wildlife Refuge (N 28.121155, W 97.442808), a 3,157-ha refuge located 12 km northeast of Sinton, southern Texas. The wildlife refuge represents an intermediate between the Gulf Coastal Prairie and Rio

Grande Plain vegetative zones (Box 1961). Soils are Victoria Clay, a typic Ustert common to the refuge and several adjacent counties (USDA-SCS 1965). The soil is heavy, neutral to calcareous, composed of high amounts of calcium, manganese, and sodium, the combination of which binds nutrients like nitrogen and phosphorous from root systems (Brady and Weil 2004).

Field Study

We selected a study area at the southernmost border of the refuge, which was classified historically as a mesquite-buffalograss community (Box 1961). The study area now is dominated by Kleberg bluestem (85-90% of total vegetation cover), with trace amounts of forbs (e.g., *Cienfuegosia drummondii*, *Ratibida columnifera*, and *Solanus elagnifolium*) and is bordered by a 2-m wide, disked firebreak on one side and a fence on the other.

We established 55, 6 x 9-m plots within the study area, with 1.5-m buffers between plots. The plot and buffer sizes were determined to permit maneuverability of disking equipment around and through the plots. Prior to treatment application in April 2011, we collected soil samples, which were analyzed for chemical composition (Texas Soil and Plant Lab, Edinburg, TX). We estimated canopy cover of vegetation by species on two 1-m^2 quadrats placed randomly within each plot in June 2011.

Treatment Application

In June 2011, we removed all standing vegetation in the treatment plots and disturbed the soil by disking once with an off-set disk prior to treatment application. We explored a total of 10 modification treatments: soil disturbance alone, decrease in pH, increase in pH, decrease in available N, and increase in mycorrhizal fungi, as well as each in combination with a native seed mix. Soil disturbance consisted only of disking with the off-set disk, and all treatment plots were disked before and after treatments were applied. To reduce pH, we applied 731.6 kg/ha of water-soluble sulfur (Disper-Sul 90% elemental sulfur) in pellet form. To increase pH, we applied 2,259.6 kg/ha of powdered lime (Austin White Lime Co., $CaCO_3$). We determined additions based on pre-treatment soil analyses and added 33% to initial calculations to ensure sufficient changes in pH to below five or above nine. To decrease nitrogen, we applied 1,360.8 kg/ha of sucrose ($C_{12}H_{22}O_{11}$; Alpert 2010). To augment the mycorrhizal fungal community, we applied 10.5 kg/ha of MycoGrow™ micronized endo/ecto seed mix (Appendix D), commercially available from Fungi Perfecti LLC (Olympia, WA); we mixed the inoculants with a small amount of soil for even distribution. We planted a mixture of native seeds on 25 of the 50 treatment plots in June 2011, at a rate of 13.0 kg/ha of pure live seed (PLS), using a native seed drill (Truax Flex III). The species and quantities included in the seed mix were based on native plants observed during pre-treatment sampling, as well as native plants selected by the South Texas Natives (Kingsville, TX, Appendix E). We randomly assigned treatments to plots and established five replicates of each treatment, for a total of 50

plots. All plots were disked multiple times after treatment application to mix soils evenly. In addition, we established five plots at random in an undisturbed part of the OWB monoculture to serve as a reference. Kleberg plots were at least 110 m away from treated plots.

Vegetation Sampling

We measured vegetation density, canopy cover, and maximum height on two 1-m^2 quadrats in each plot for every month after initial treatment during summer 2011-2013. We placed quadrats at random within each plot for each sampling period, but quadrats always were at least 1 m from plot boundaries to avoid edge effects. All plants were identified to species using Everitt et al. (2011) for grasses, Everitt et al. (2002) for woody plant species, and Everitt et al. (1999) for herbaceous plant species, and cross-referenced with type specimens in the Welder Wildlife Foundation herbarium.

We measured vegetation density by species as the number of plants in each quadrat. We considered grasses as separate individuals if the crowns from stolons occurred more than 10 cm from the original crown base. Biennial or perennial species that appeared dead were considered as living due to uncertainty created by drought conditions. We estimated canopy cover (≤ 1-m tall) by species, as well as cover of bare ground and litter (vegetative material separate from living vegetation or growing structures attached to the ground). We then grouped plant species into three specific cover classes: grasses, forbs (herbaceous plant species), and woody plants. Finally, we measured the height of the tallest plant of each species in each quadrat, and averaged

the height of each species in each quadrat as a measure of height in the plant community.

We used species richness, canopy cover by cover class, and maximum plant height as measures of community richness and structure, and canopy cover of Kleberg bluestem as a measure of OWB dominance. In addition, we measured the proportion of total plant density comprised of OWB plants as another measure of OWB dominance (OWB density) to make comparisons with the microcosm study.

Soil Sampling

In May of each field season, we sampled 1 L of soil from each quadrat to determine soil chemistry. We collected soil up to a depth of 15 cm in each quadrat and combined samples from quadrats within plots. Soil samples were analyzed by Texas Plant and Soil Labs (Edinburg, TX) to determine soil pH, as well as available nitrogen (NO_3) using an extractable CO_2 method (McGeorge and Breazeale 1931; Texas Plant and Soil Labs 2012). We used these soil characteristics to assess treatment efficacy and to make comparisons with the microcosm experiment.

Precipitation

We obtained precipitation data from a nearby weather station at the headquarters of the Welder Wildlife Refuge, approximately 7.2 km from the study area. We quantified monthly precipitation from October 1956 (from the start of the water year, October 1) until September 2013 and we compared annual precipitation during

our study to the long-term annual mean to assess the severity of drought. We used the Palmer Drought Severity Index (NCDC-NOAA 2014) as a measure of drought severity for each field season (June-August) in the study.

Microcosm Experiment

In June 2013, we collected 378 L of the first 30 cm of soil from random locations in unused portions of the study area as the substrate for the microcosm experiment. We considered the same soil modification treatments, with and without native seed mix (10 total treatments), as in the field study. We randomly assigned treatments to Treepot™ containers (10 x 36 cm, 2.83 L volume, Stuewe and Sons Inc., Tangent, OR), with 10 replicates of each treatment combination (100 total pots). We adjusted application rates used in the field study to accommodate the smaller pot size. We assumed that collected soils had a seed bank that included viable OWB seeds, based on their proximity to standing OWB vegetation and because we observed seeds in collected soil.

We placed pots in an enclosure to prevent herbivory, and exposed pots to typical warm-season growing conditions in San Patricio County (High 34.5^{o} C, Low 24.5^{o} C, with a total of 14 h of light). We used the annual median precipitation recorded from the weather station on the refuge to guide the water ration, rather than the annual mean, due to the high variability recorded. We applied 170 ml of water to each container at the beginning of the study to ensure germination of seed, and added 60 – 70 ml of water to each container every three days to retain soil moisture.

We used the same methods as the field study to sample plant species richness, density, and maximum height, but only collected these measurements eight weeks after initial soil treatments, which would reflect two months post-treatment. We used plant species richness, density, and maximum height as measures of community richness and structure. We converted plant density from plants/pot to plants/m^2 to make comparisons with the field study. We measured the proportion of OWB plants in each container as a measure of dominance (OWB density) to make comparisons with the field study. After eight weeks, we collected 1 L of soil from each container to quantify soil pH and available nitrogen, based on the same methods as the field study.

Data Analysis

We examined the efficacy of soil modification treatments on vegetation and soil characteristics using linear mixed models. We included soil treatment, seeding treatment, and year as independent factors in all models, and explored evidence for two-way interactions (soil treatment * seeding, year * soil treatment, and year * seeding). We removed interaction terms when $P > 0.1$, but retained all simple effects in final models. When appropriate, we accounted for repeated measurements and considered three possible covariance structures: no within-group covariance, compound symmetric, or first-order autoregressive, selecting the most appropriate covariance structure based on lowest AIC values. When necessary, we transformed response variables to meet assumptions. All analyses were completed using the nlme package in R (Pinheiro et al. 2013; R Core Development Team 2013).

No vegetation grew in treatment plots during the first two months post-treatment due to lack of rain. As such, we did not analyze vegetation data from the field study for 2011. We did not analyze cover of litter and woody plants in the field study because most values were zero. We computed means and 95% confidence intervals for Kleberg plots, to make comparisons with treatment plots.

Results

Precipitation

Total rainfall for the water year (October 1—September 30) measured 32.3 cm for 2011, 62.5 cm for 2012, and 69.1 cm for 2013, which was 36%, 69%, and 76% of the long-term average (90.2 cm), respectively. The majority of rainfall did not occur during our sampling periods (Fig. 3.1). We observed the most precipitation between field seasons in 2013 (2011 = 1.1 cm, 2012 = 12.0 cm, 2013 = 26.0 cm).

Field Study

We identified a total of 53 plant species during the field study, including 17 species of native grasses, four invasive grasses, 30 forbs, and two woody plants (Appendix F). Kleberg bluestem and wooly croton (*Croton capitatus*) were commonly sampled in all plots (Appendix F). Woody vegetation, although present in some plots, represented less than 1% of all individual plants sampled during the study (Appendix F). We observed seven species of plants from the native seed mix (*Bouteloua curtipendula*, *B. repens*, *Chloris cucullata*, *Elymus canadensis, Panicum halli* var. *halli, Pappophorum*

bicolor, *Setaria* spp.), but these species represented only 2% of all plants observed in seeded plots (Appendix F).

Soil pH and available nitrogen differed by soil and seeding treatment, but the magnitude of some differences changed over time (Table 3.1). Adding lime increased soil pH immediately after treatment, relative to soil disturbance and Kleberg plots, but this effect did not persist in subsequent years (Fig. 3.2). In contrast, adding sulfur did not change soil pH at any time post-treatment, relative to soil disturbance and Kleberg plots (Fig. 3.2). We did not observe differences in soil pH for plots with added carbon ($F_{4,44}$ = 0.50, P = 0.62) or mycorrhizal fungi ($F_{4,44}$ = 0.31, P = 0.42) relative to plots with soil disturbance. Although we detected differences in soil pH based on seeding (Table 3.1), seeding only increased soil pH by 0.1 units (95% CI = 0.0 – 0.2).

Adding carbon did not alter available nitrogen relative to soil disturbance and Kleberg plots, but both carbon-treated and disturbance plots had more nitrogen than Kleberg plots by the second year post-treatment (Fig. 3.3). Disturbed plots had 17.0 kg/ha less available nitrogen than plots treated with lime (8.3 – 34.8) and 7.1 kg/ha less than plots treated with sulfur (3.5 – 14.7) within the first year post-treatment (Table 3.2). Available nitrogen in plots with added mycorrhizal fungi was comparable to disturbed plots (Table 3.2). Plots with added seed had 2.7 kg/ha more available plant nitrogen (1.2 – 3.4) compared to plots without seed (Table 3.2).

We did not detect differences in vegetation composition or structure based on soil treatment (Table 3.3). Seeding did affect vegetation characteristics, but the

magnitude often changed over time (Table 3.3; Figs. 3.4 and 3.5). Adding seed reduced dominance of OWBs during both years post-treatment, relative to plots without seed (Figs. 3.4 and 3.5). One year after we added seed, density of OWBs was 30% lower (95% CI = 12.2 – 77.1) and cover was 27% lower (7.3 – 94.6) than plots without seed; two years post-treatment, density of OWBs was 32% lower (18.5 – 56.6) and cover was 38% lower (17.1 – 82.1). Adding seed also resulted in an increase in species richness of plants (1.2 species/m^2, 0.6 – 2.0) and cover of native grasses (3.9%, 0.4 – 9.0; Figs. 3.4 and 3.5). In addition, plots with added seed had more bare ground (6.2%, 1.9 – 25.4) and forb cover (12.7%, 5.2 – 29.8) relative to plots without seed two years post-treatment (Fig. 3.5). Plants in plots with seed were 9.4 cm shorter (9.2 – 9.7) than plants in plots without seed one year post-treatment and 29.1 cm shorter (26.9 – 31.2) two years post-treatment (Fig. 3.4). However, all plots had lower cover and density of OWBs, more species of plants, and higher cover of native grasses and forbs, regardless of soil and seed treatment, than Kleberg plots dominated by OWBs in both years post-treatment (Figs. 3.4 and 3.5).

Microcosm Experiment

We identified a total of 11 plant species during the microcosm experiment, including six species of native grasses, two invasive grasses, one sedge, and three forbs (Appendix G). The most common plant species during the two months post-treatment included Kleberg bluestem, knotroot bristlegrass (*Setaria ramiseta* var. *formula*), and junglerice (*Echinochloa colona*; Table 3.4). We observed only three plant species from

the seed mix (*Bouteloua curtipendula*, *B. repens*, and *Panicum halli* var. *filipes*; Appendix E) growing in seeded pots (Appendix G).

We did not detect differences in soil pH based on soil or seed treatments (Table 3.5), nor between microcosm pots and OWB-dominated plots in the field study (Table 3.6). We did not detect differences in available nitrogen based on soil treatments, but pots with seed had 7.2 kg/ha less (4.4 – 11.1) available nitrogen relative to pots with seed (Tables 3.5 and 3.6). Soils in all microcosm pots had more available nitrogen than Kleberg plots in the field study (Table 3.6; Fig. 3.6).

As in the field study, we did not detect differences in vegetation composition and structure based on soil treatments in the microcosm experiment (Table 3.5). OWB density did not differ between plots with and without seed, but OWB density was lower in pots relative to Kleberg plots (Table 3.6). Seeding resulted in 0.7 more plant species/m^2 (0.5 – 0.8) relative to pots without seed (Fig. 3.7). Plants also were taller (5.0 cm, 4.5 – 5.7) in pots with added seed relative to pots without seed (Table 3.6).

Discussion

Understanding the mechanisms that promote plant invasion may provide guidance about management tools that increase restoration success (Levine et al. 2003). Disturbance can augment restoration efforts if the disturbance event increases plant diversity and decreases dominance of invasive plants (Bard et al. 2004; Brooks et al. 2004; Johnson and Fulbright 2008; Limb et al. 2010). In addition to plant invasion and

drought, we explored the effects of restoration tools (soil modification and seeding treatments) as a form of disturbance to restore native plant communities. We documented increased species richness and cover of native plants and reduced dominance of invasive plants on plots treated with simple soil disturbance and seeding. When we alleviated drought in the microcosm study, we observed similar results as in the field study, suggesting that the influence of soil disturbance and seeding on vegetation composition and structure was independent of drought effects. However, we found that changes in soil chemistry were either short-lived or absent in both studies, indicating that other factors might be influencing vegetation characteristics.

Altering soil pH in combination with soil disturbance could increase potential for restoration by providing conditions that favor some native plants (Elliott et al. 2013; Farrel et al. 2005; Lawson et al. 2004; Longhurst et al. 1999; Owen and Marrs 2000; Tibbett and Diaz 2005). Adding elemental sulfur can increase soil fertility and native plant cover (Owen and Marrs 2000; Farrel et al. 2005) and liming can reduce soil toxicity in acidic soils and increase native forbs (Dorland et al. 2005; Elliot et al. 2013; Longhurst et al. 1999). An observed change in soil pH from applying sulfur may take months (Owen and Marrs 2001; Tibbet and Diaz 2005), whereas soil pH responds to liming immediately (Elliott et al. 2013). Although we observed an immediate change in soil pH for lime-treated plots, neither sulfur nor lime altered soil pH over longer time periods in either study.

Initial soil chemistry may have influenced the efficacy of our soil treatments. In previous studies, soils treated with lime or sulfur were acidic before treatment (Dorland et al. 2005; Elliot et al. 2011; Owen and Marrs 2000; Tibbett and Diaz 2005), whereas untreated soils in our study were alkaline. Carbonate compounds commonly buffer out acidic or basic components in alkaline soils (Brady and Weil 2004). Soils in our study had large concentrations of carbonates and may have been resilient to changes in chemistry (USDA-SCS 1965). However, applying multiple applications of lime or in higher concentrations may be neither practical nor feasible for land management.

Changes in nutrient availability, such as increased soil nitrogen from agriculture or nitrogen-fixing plants, may increase the competitive ability of invasive plants (Abraham et al. 2009; Alpert 2010; Alpert and Maron 2000; Blumenthal 2009; Huenneke et al. 1990; Siemann and Rogers 2007; Sigüenza et al. 2006; Suding et al. 2004; Vitousek et al. 1996). Therefore, reducing the availability of nutrients may reduce the dominance of invasive plants (Alpert 2010; Blumenthal et al. 2003). Adding organic carbon to enriched soils promotes nitrogen uptake by microbial communities that reduce available nitrogen for plants, which in turn facilitate negative interactions between invasive plants, the microbial community, and the native plant community (Alpert 2010; Blumenthal et al. 2003; Corbin and D'Antonio 2004). OWB grasses respond quickly to nitrogen fertilizers (Berg 1993) and enriched soils may increase OWB dominance. Our study area was previously grazed (Box 1961), which can increase nitrogen deposition in localized areas (Bardgett and Wardle 2003; McNaughton et al. 1997). We assumed that

soils in our study area were rich in nitrogen, allowing OWB grasses to compete with native plants. However, we documented lower available nitrogen in untreated plots than other studies that used carbon addition (Abraham et al. 2009; Alpert 2010). Soils in the field study may have been nitrogen-limited, and given that carbon additions are effective when nitrogen is not limiting plant growth (Alpert 2010; Mangold and Sheley 2008), we would not consider this modification treatment practical in nitrogen-limited soils. Understanding the effects of carbon additions on OWBs where soils are nitrogen-rich may offer evidence to support the efficacy of this restoration tool.

Changes in soil chemistry can have negative impacts on soil communities and reintroductions of native soil biota may be necessary to restore native plant communities (Biondini et al. 1985; Doerr et al. 1984; Heneghan et al. 2008; Jansa et al. 2003; Ohsowski et al. 2012). Mycorrhizal fungi can increase soil stability through hyphae and can aid in acquiring resources such as nitrogen, phosphorous, and water in colonized plants (Koide and Dickie 2002; Ohsowski et al. 2012). However, some invasive plants are just as likely to be colonized by mycorrhizal fungi as native plants, and mycorrhizae could increase the competitive ability of the invasive plant (Callaway et al. 2003). Mycorrhizal colonization has been observed in *Bothriochloa* (Wilson and Hartnett 1998; Wilson et al. 2012) and only recently in *Dichanthium* (Jalonen et al. 2013; Pérez and Peroza 2013). Although we did not observe change in vegetation or soil characteristics for plots treated with mycorrhizae, inoculations could increase the ability of OWBs to compete with native grasses. The lack of changes we observed in vegetation

characteristics may have been a failure of mycorrhizae to establish, due in part to a lack of localized spores or differences in soil chemistry (Ji et al. 2012; Paluch et al. 2012; Vogelsang et al. 2006; Wang and Qiu 2006).

Like soil biota, native seeds are affected by changes in soil properties. In invaded soils, invasive plants may establish before native plants because seed banks may be inundated with seeds of invasive plants (Lockwood et al. 2005) or depauperate of native seeds (D'Antonio and Meyerson 2003; Middleton 2003). Augmenting the native seed bank could increase establishment of native plants in disturbed landscapes (Blumenthal et al. 2003; Carter and Blair 2012; Falk et al. 2013; Sheley and Half 2006). Furthermore, locally-adapted propagules would allow native plant communities to overcome the complex effects of plant invasion and other disturbances, such as drought (Carter and Blair 2012; Falk et al. 2013). Although we observed more species and cover of native plants in plots with added seed during the field study, most plants in plots with added seed were not species included in the native seed mix. In contrast, ~20% of all plants sampled in pots with added seed during the microcosm study were part of the seed mix. Most plants detected in seeded plots were early-succession and drought-tolerant species (Everitt et al. 2002, 2011), suggesting that plant composition may be more influenced by drought than by soil treatment and seeding.

We suggest that the seeding effect in the field study resulted because propagule pressure from OWBs was higher in plots without added seed or added seeds were consumed by seed predators. Invasive plants, such as OWBs, produce large quantities of

seed and competition from seedlings of the invasive plant may exacerbate stress on native plants competiting with invasive adults (Abraham et al. 2009; Bryson and Carter 2004; Coyne and Bradford 1985; D'Antonio et al. 2000; Lockwood et al. 2005; Sanders et al. 2007). Propagules may be deposited by wind and adjacent sites may serve as reservoirs (Archer and Pyke 1991; Wilson and Pärtel 2003). OWB monocultures were adjacent and upwind of plots without seed, but not to plots with added seed, which may explain increased cover and density of OWBs in plots without seed relative to plots with seed. However, we observed more OWBs in pots without seed in the microcosm experiment, and propagule pressure may not be responsible for the differences between seeding in the field study.

Alternatively, restoration projects that implement seeding can be complicated by granivores, such as rodents and ants (Díaz 1992; Everett et al. 1978; Fisher and Cover 2007; Maron et al. 2012, 2014; MacDougall and Wilson 2007; Pearson et al. 2011, 2012; Retana et al. 2004). We collected arthropods on plots for a concurrent study (Chapter 4), and observed granivorous ants (*Pogonomyrmex* spp.) only in plots with added seed. Although we did not sample rodents, we did observe predators such as western diamondback rattlesnakes (*Crotalus atrox*), suggesting that rodents may be present (Mitchell, personal observation). Adding seed may have provided food for these seed predators, while allowing native seeds in the seed bank to germinate.

Conclusions

Compounding effects of biotic and abiotic disturbances create unique challenges for restoration and management (Hobbs and Huenneke 1992; Hobbs et al. 2009; Paine et al. 1998). Furthermore, the frequency and severity of drought events are expected to increase (IPCC 2007) and changes in climate may create novel situations where plant communities cannot return to a pre-disturbed state (Hobbs et al. 2009; Westoby et al. 1989). Understanding the mechanisms behind how disturbances, such as drought, influence invaded plant communities may provide insight to develop more resilient techniques for restoration and management.

We incorporated an alternative approach to restoring coastal prairies impacted by an invasive grass and drought, but soil composition inhibited our efforts to modify soil characteristics. Mechanical soil disturbance and seeding with locally-adapted plant species increased diversity and cover of native plants and reduced dominance by invasive plants, relative to undisturbed monocultures of invasive plants. We documented that the effects of soil disturbance and seeding either persisted or increased over time and were similar in the presence and absence of drought. Therefore, the capacity of soil disturbance and seeding to reduce dominance of OWBs is resilient to drought, at least over the short term, but long-term monitoring of treated plots may reveal additional benefits as drought effects subside. Additionally, investigating the efficacy of soil treatments in other soil types that have been invaded by OWBs may provide insights for restoring native plant communities.

Tables

Table 3.1. Factors affecting soil characteristics in the field study, southern Texas, summers 2011-2013.

Soil Variable	Soil Treatment		Seed		Year		Soil*Year[a]	
	$F_{4,44}$	*P*	$F_{1,44}$	*P*	$F_{2,90}$	*P*	$F_{8,90}$	*P*
Soil pH	27.25	<0.001	5.07	0.029	7.06	0.001	4.86	<0.001
Available NO_3	0.91	0.469	5.66	0.022	2.08	0.131	3.06	0.004

[a] We did not detect interactions between seeding and soil treatment or seeding and year.

Table 3.2. Available nitrogen (NO_3,kg/ha) for plots in the field study (medians and back-transformed 95% CIs), southern Texas, summers 2011-2013.

		Time		
Soil Treatment	**Seeding**	**2011**	**2012**	**2013**
Disturbance	Without Seed	14.7	10.5	21.2
		(8.5-25.4)	(5.1-21.5)	(10.4-43.5)
	With Seed	19.1	13.6	27.5
		(15.3-23.8)	(6.5-28.5)	(13.4-56.4)
Carbon	Without Seed	9.1	13.2	17.1
		(6.1-13.5)	(6.3-27.6)	(10.3-28.3)
	With Seed	11.8	17.1	22.1
		(9.5-14.7)	(8.3-35.0)	(13.3-36.7)
Fungi	Without Seed	15.5	11.6	17.5
		(9.0-26.7)	(5.7-23.8)	(8.5-35.8)
	With Seed	20.1	15.1	22.6
		(16.1-25.0)	(7.4-30.9)	(11.0-46.4)
Lime	Without Seed	14.9	27.5	13.1
		(8.6-25.7)	(13.4-56.3)	(6.4-26.9)
	With Seed	19.3	35.6	17.0
		(15.5-24.1)	(17.4-72.9)	(8.3-34.9)
Sulfur	Without Seed	11.7	17.6	13.6
		(6.8-20.2)	(8.6-36.2)	(6.6-27.9)
	With Seed	15.2	22.9	17.6
		(12.2-19.0)	(11.2-46.9)	(8.6-36.1)
OWB	Reference	9.5	3.5	5.4
		(5.5-16.6)	(2.0-6.0)	(2.9-10.1)

Table 3.3. Factors affecting vegetation characteristics in the field study, southern Texas, summers 2012-2013.

Vegetation Variable	Soil Treatment		Seed		Year		Seed*Year[a]	
	$F_{4,44}$	*P*	$F_{1,44}$	*P*	$F_{1,249}$	*P*	$F_{1,248}$	*P*
Richness	1.28	0.294	20.79	<0.001	8.89	0.003		
OWB Density	0.38	0.820	49.18	<0.001	8.66	0.004	7.56	0.006
Height	0.21	0.934	67.22	<0.001	0.26	0.613	28.94	<0.001
Cover Class								
Bare Ground	1.84	0.137	15.32	<0.001	76.00	<0.001	39.03	<0.001
Native Grasses	0.60	0.663	20.21	<0.001	30.96	<0.001		
OWB Grasses	0.35	0.843	53.00	<0.001	4.20	0.042	9.47	0.002
Forbs	0.17	0.951	4.44	0.041	0.00	0.975	5.63	0.018

[a] We did not detect interactions between seeding and soil treatment or soil treatment and year.

Table 3.4. Five most common plant species observed in the microcosm study based on the percentage of individuals in pots with seed (*n* = 162 individuals), and without seed (*n* = 142 individuals), southern Texas, summer 2013.

Category	Common Name	Scientific Name	% of Individuals
Without Seed	Kleberg bluestem	*Dichanthium annulatum*	51.4
	Knotroot bristlegrass	*Setaria ramiseta* var. *formula*	15.5
	Junglerice	*Echinochloa colona*	12.7
	Texas signalgrass	*Urochloa texana*	8.5
	White-margined euphorbia	*Euphorbia albomarginata*	5.6
With Seed	Kleberg bluestem	*Dichanthium annulatum*	26.5
	Knotroot bristlegrass	*Setaria ramiseta* var. *formula*	19.1
	Western sedge	*Carex occidentalis*	14.2
	Junglerice	*Echinochloa colona*	11.1
	Slender gramma	*Bouteloua curtipendula*	8.0

Table 3.5. Factors affecting vegetation and soil characteristics in the microcosm study, southern Texas, summer 2013.

Soil/Vegetation Variable	Soil Treatment		Seed[a]	
	$F_{4,93}$	*P*	$F_{1,93}$	*P*
Soil pH	1.72	0.151	0.55	0.458
Available NO_3	1.26	0.293	5.53	0.021
Richness	0.37	0.827	9.25	0.003
OWB Density	0.27	0.898	2.50	0.117
Height	1.99	0.102	14.66	<0.001

[a] We did not detect interactions between seeding and soil.

Table 3.6. Vegetation and soil characteristics for pots in the microcosm study (medians and back-transformed 95% CIs). Kleberg plots (OWB, medians and back-transformed 95% CIs) from the field study are provided for comparison. We did not compare plant height between Kleberg plots and pots in the microcosm study because of differences in the length of the growing period.

	Soil pH	**Available NO_3 (kg/ha)**	**Richness**	**OWB Density (%)**	**Height (cm)**
Without	7.96	28.3	1.3	9.1	14.6
Seed	(7.87-8.04)	(20.9-38.2)	(1.0-1.9)	(3.2-25.3)	(11.3-17.8)
With	7.93	21.1	2.0	4.7	19.6
Seed	(7.86-8.00)	(16.5-27.1)	(1.5-2.6)	(2.0-10.8)	(17.0-22.3)
OWB	7.80	5.4	2.3	95.0	
Reference	(7.19-8.42)	(1.7-17.5)	(1.5-3.3)	(79.5-100.0)	

Figures

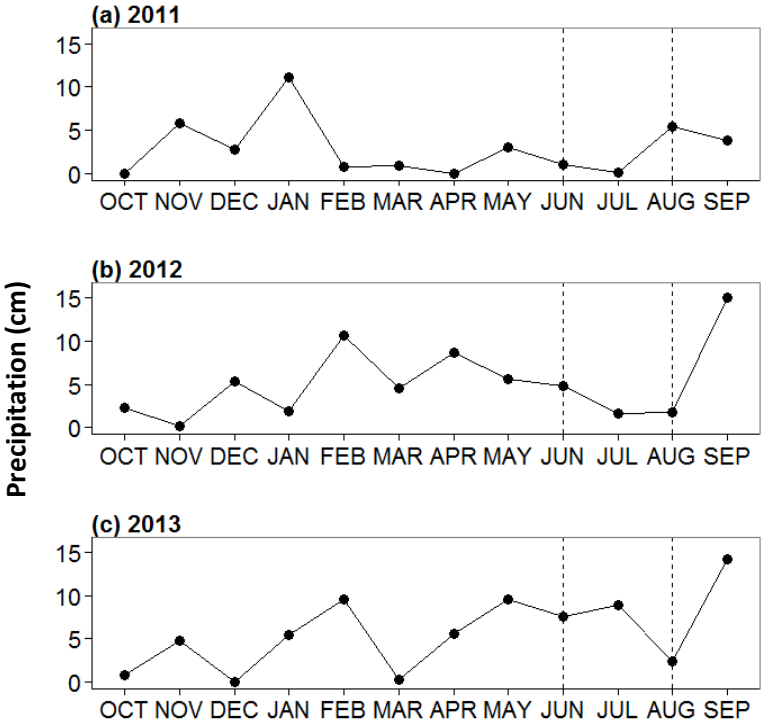

Figure 3.1. Total monthly precipitation for the Welder Wildlife Refuge, starting at the beginning of the water year (Oct 1), southern Texas, 2011-2013. The dashed lines represent precipitation observed during the months of sampling.

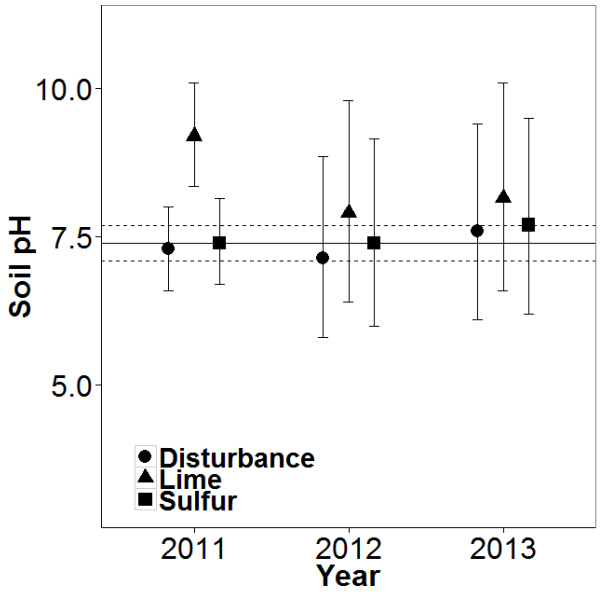

Figure 3.2. Soil pH (means and 95% CIs) for plots in the field study, southern Texas, summers 2011-2013. We include the mean (solid) and 95% CI (dashed) for the Kleberg (OWB) plots for comparison.

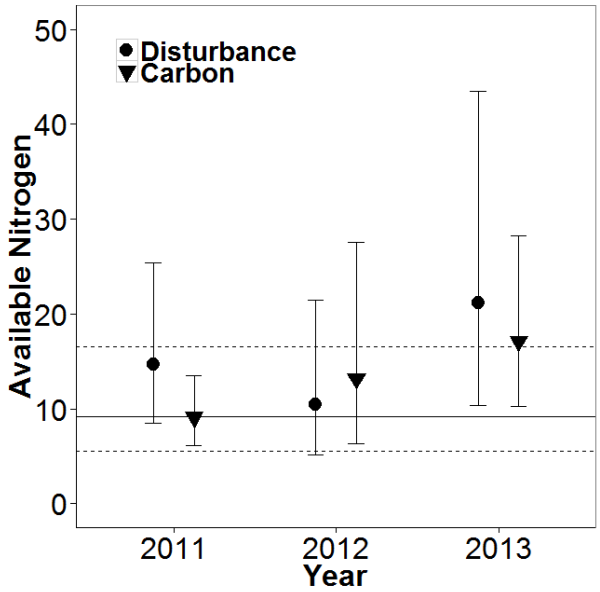

Figure 3.3. Available nitrogen (kg/ha, means and 95% CIs) for plots in the field study, southern Texas, summers 2011-2013. We include the mean (solid) and 95% CI (dashed) for the Kleberg (OWB) plots for comparison.

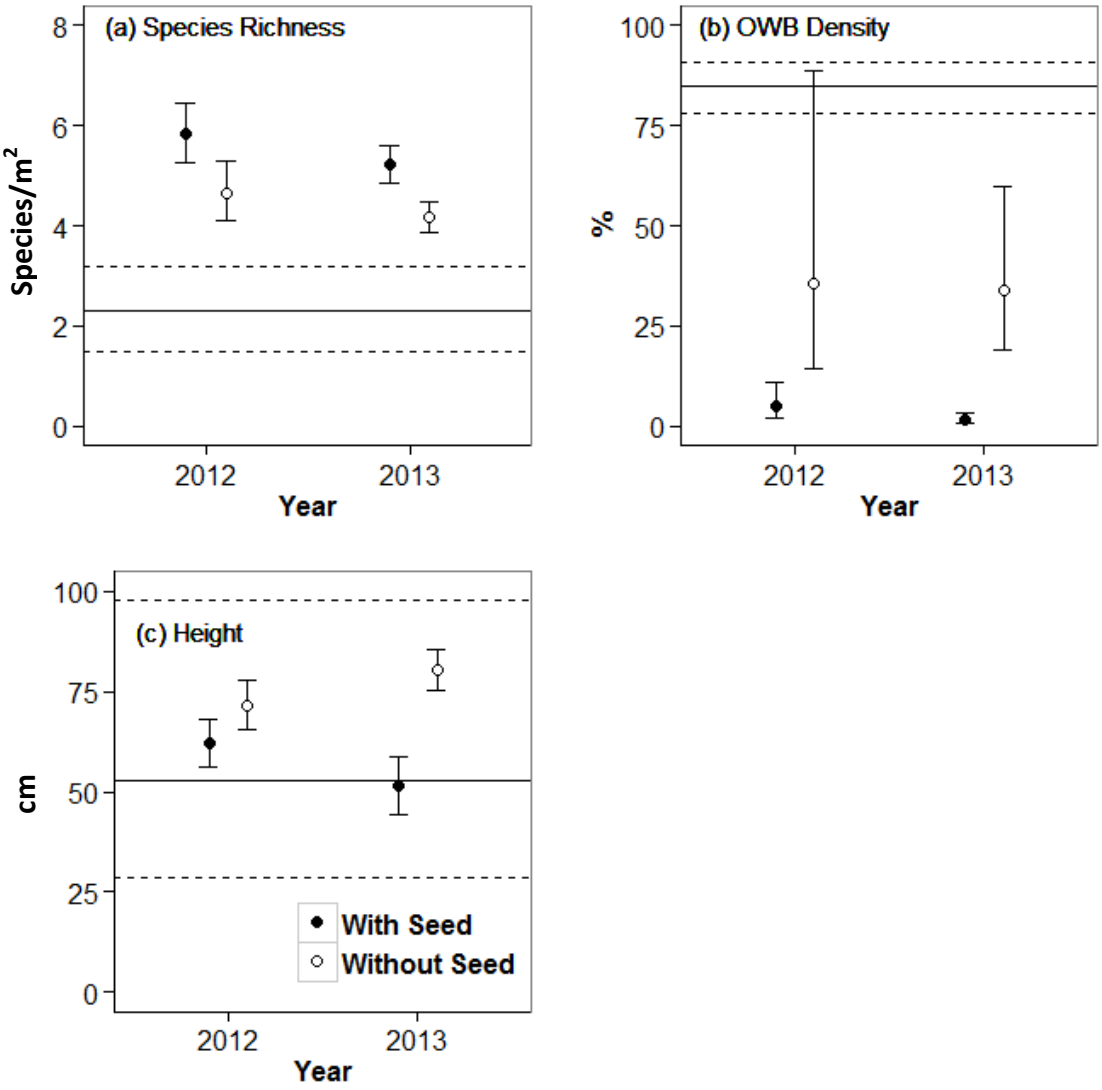

Figure 3.4. Vegetation characteristics (means and 95% CIs) for plots in the field study with and without seed added, including (a) plant richness, (b) OWB density, and (c) maximum plant height. We include the mean (solid) and 95% CI (dashed) for the Kleberg (OWB) plots for comparison.

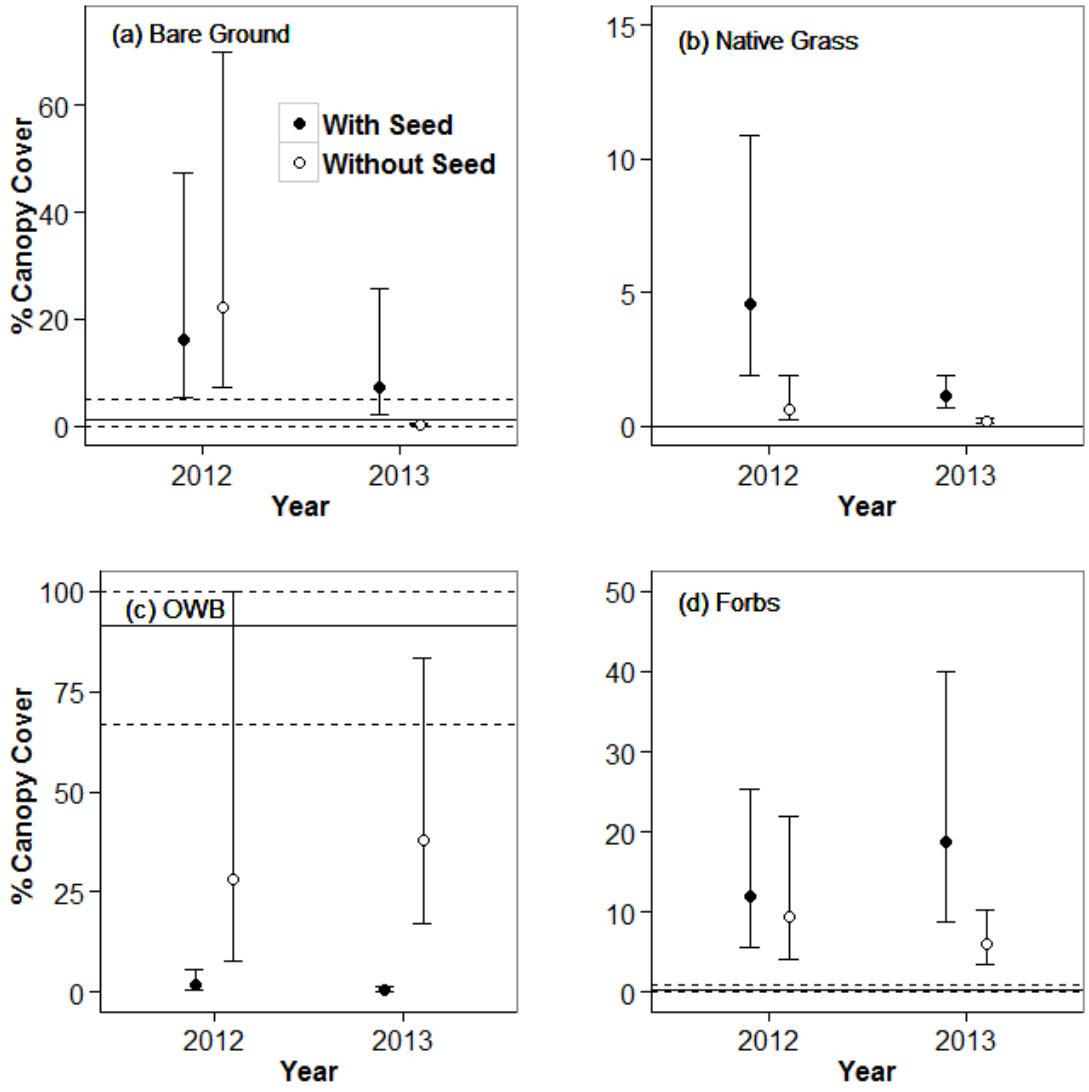

Figure 3.5. Canopy cover by cover class (means and 95% CIs) of plots in the field study with and without native seed added, southern Texas, summers 2012-2013. We include the mean (solid) and 95% CI (dashed) for Kleberg (OWB) plots for comparison.

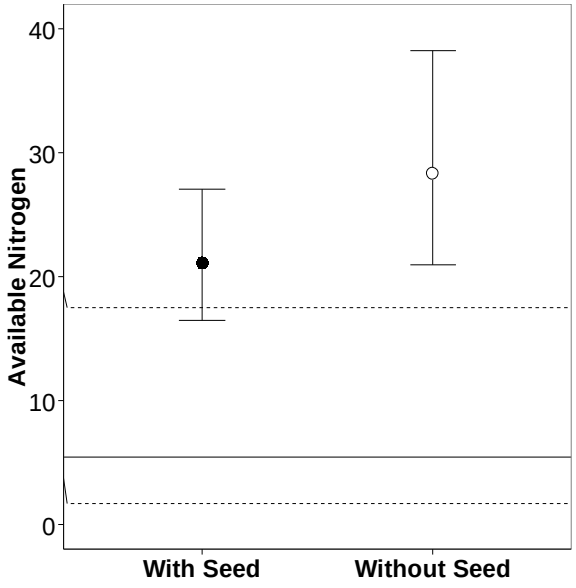

Figure 3.6. Available NO_3 (kg/ha, means and 95% CIs) for pots in the microcosm study, southern Texas, summer 2013. We include the mean (solid) and 95% CI (dashed) for the Kleberg (OWB) plots from the field study for comparison.

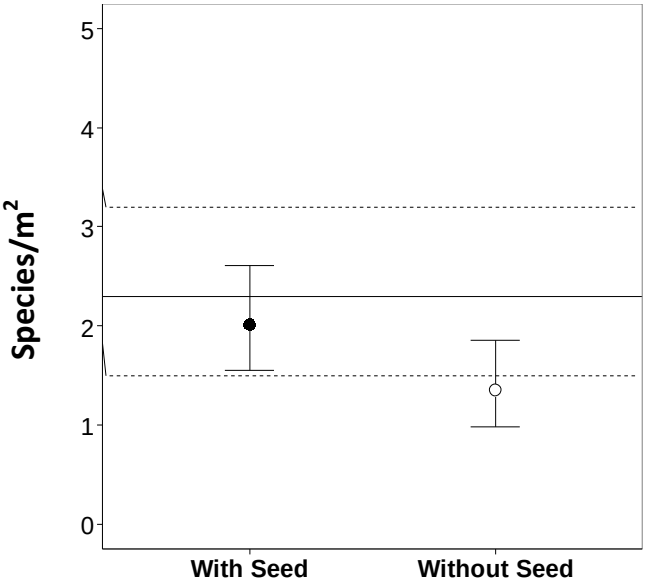

Figure 3.7. Plant species richness (means and 95% CIs) for pots in the microcosm study, southern Texas, summer 2013. We include the mean (solid) and 95% CI (dashed) for the Kleberg (OWB) plots for comparison.

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CHAPTER FOUR

SOIL MODIFICATION TO RESTORE NATIVE ARTHROPOD COMMUNITIES IMPACTED BY PLANT INVASION AND DROUGHT

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CHAPTER FOUR

SOIL MODIFICATION TO RESTORE NATIVE ARTHROPOD COMMUNITIES IMPACTED BY PLANT INVASION AND DROUGHT

Abstract

Changes in the characteristics of plant and soil communities following plant invasion can have detrimental effects on native arthropod communities. Invasive plants may alter soil conditions and the combination of multiple disturbances affecting plant, arthropod, or soil characteristics may create novel situations that alter the efficacy of restoration. Modifying properties of the soil to favor native plant and arthropod communities may provide alternatives to traditional management strategies. We conducted a field experiment to measure the efficacy of soil modification techniques at reducing dominance of invasive plants and restoring native arthropod communities. We conducted our study in grasslands dominated by Old World bluestems (OWBs, *Dichanthium annulatum*) during an extreme drought, which provided us with the opportunity to test the efficacy of soil treatments and seeding under varying weather conditions. We applied 10 treatments (simple soil disturbance, pH decrease, pH increase, carbon addition, mycorrhizal fungi, and each of the previous in combination with native seed) to 50 plots in June 2011 and compared treated plots to Kleberg plots dominated by OWBs. We sampled arthropods June-August in 2011-2013, quantified arthropod abundance and richness, presence of morphospecies, and abundance and richness of functional groups (herbivores, decomposers, predators, and ants). Treated plots had more species and cover of native plants as drought conditions subsided, which was associated with an increased diversity of herbivore, decomposer, and predator communities, compared to Kleberg plots. Although invasive arthropods were present in all plots, treated plots had fewer invasive arthropods than Kleberg communities. Seeded plots also had more litter cover and more decomposer arthropods;diverse plant litter may provide higher quality habitat for detritivores than monocultures of invasive plants. Some arthropods, such as ants and isopods, may have inhibited some treatments, such as seeding, and a thorough understanding of the arthropod community prior to treatment may help determine which restoration tools may be most effective. Soil disturbance and seeding with native propagules increased diversity of native plants and arthropods and reduced OWB dominance in the short term, but long-term monitoring after soil modification may reveal additional benefits to native communities.

Introduction

Invasive plant species are a threat to biodiversity (Chornesky and Randall 2003; Hobbs and Huenneke 1992; Wilcove 1989), as invasive plants reduce richness and composition of native plants (Gaertner et al. 2009; Levine et al. 2003; Vilà et al. 2011). Changes in the composition or diversity of plant communities may affect availability and quality of habitat for organisms at different trophic levels, especially arthropods (Gratton and Denno 2006; Litt and Steidl 2010; Litt et al., in press; van Hengstum et al. 2014). Arthropods respond quickly to changes in the environment and shifts in plant community characteristics associated with plant invasion can alter native arthropod communities (Burger et al. 2003; de Buryn 1999; Haimi 2000; Litt et al., in press; Samways 1996; Snyder and Hendrix 2008; Wilson 1987).

Native arthropods, such as herbivores and pollinators, may not recognize novel plants as habitat or food (Brown et al. 2002; Burghardt et al. 2010; Grabas and Laverty 1999; Tallamy 2004; Williams et al. 2011). Changes in litter composition or abundance associated with invasive plants have concomitant effects on the abundance and composition of detritivore arthropods (Alerding and Hunter 2013; Kappes et al. 2007; Wolkovich et al. 2009). Other changes in plant structure following plant invasion, such as cover, density, and height, alter the behavior or movement of arthropods (Crist et al. 2006; Pearson 2009; Samways et al. 1996; Schirmel et al. 2011; Standish 2004; Wolkovich et al. 2009; Wu et al. 2009). Finally, changes in the abundance and diversity of prey associated with changes in plant characteristics have concomitant effects on the

composition and abundance of predaceous arthropods (deHart and Strand 2012; Gratton and Denno 2006).

Traditional management strategies to reduce plant invasion may be inhibited by feedback loops that favor invasive plants (Chornesky and Randall 2003; Vinton and Goergen 2006). In contrast, modifying soil characteristics to favor native plant species may provide alternative restoration strategies. Altering soil pH may increase nutrient acquisition or soil fertility for native plants (Elliott et al. 2013; Farrel et al. 2005; Lawson et al. 2004; Longhurst et al. 1999; Owen and Marrs 2000; Tibbett and Diaz 2005). Reducing the availability of nutrients, such as carbon additions,,may allow native plants to better compete with invasive plants (Alpert 2010; Blumenthal et al. 2009). Invasive plants and other disturbances could reduce native mutualistic symbionts in the soil, and inoculating mutualists like mycorrhizal fungi may promote the establishment of native plants (Archer and Pyke 1991; Biondini et al. 1985; Callaway et al. 2003; Stinson et al. 2006; Wolfe et al. 2008).

Arthropods also are sensitive to changes in soil conditions (Brussaard 1997; Haimi 2000) and modifying soil properties may increase abundance and diversity of native arthropod communities. Altering soil pH can alter abundance and composition of soil-dwelling arthropods by immobilizing heavy metals or increasing the availability of nutrients (Aldering and Hunter 2013; Geissen et al. 1997; Haimi et al. 2000; Haimi and Mätäsniemi 2002; Liiri et al. 2002; McGrath and Binkley 2009). Increasing available nitrogen can alter species richness and abundance of herbivores due to changes in

palatability or abundance of native plants (Blue et al. 2011; Haddad et al. 2000). Mycorrhizal fungi may increase the growth rate or nutrient uptake of native plants, with positive effects for herbivorous and pollinator insects (de la Peña 2011; Gange et al. 2003, 2005; Gange and Smith 2005; Hemple et al. 2009).

Old World bluestems (OWBs, *Bothriochloa* and *Dichanthium* spp.) are a group of warm-season perennial grasses (Celarier 1958) that were introduced in the early twentieth century as a potential cattle forage and have become dominant in the central and southern Great Plains of the United States (Nixon 1949; USDA-NRCS 2014). Old World bluestems can tolerate disturbances such as grazing and drought and respond quickly to increased nitrogen loads (Berg 1993; Coyne and Bradford 1985; White and Dewald 1996). Old World bluestems may alter ecological processes, such as fire (Reed et al. 2005), and reduce the diversity of native plant and animal communities (Cord 2011; Gabbard and Fowler 2007; Hickman et al. 2006; Sammon and Wilkens 2005; Schmidt et al. 2008; Woodin et al. 2010).

Previous management strategies to reduce the dominance of OWBs have been unsuccessful. Prescribed burns have had variable results at reducing OWBs (Berg 1993; Ruckman et al. 2011; Simmons et al. 2007; Twidwell et al. 2012). Applying prescribed fire may not be feasible if local burn bans are (Simmons et al. 2007; Ruckman et al. 2011). Similarly to prescribed fire, application of herbicides to reduce OWB populations have been variable and OWBs generally recover within one year (Harmoney et al. 2004, 2007; Mittelhauser et al. 2011; Ruffner and Barnes 2012). Herbicides also may inhibit

the establishment of native plants and impede restoration success (Harmoney et al. 2007; Ruckman et al. 2011; Ruffner and Barnes 2012). Soil modification techniques have not been examined for OWBs, but may provide an alternative restoration tool to reduce dominance of OWBs in invaded landscapes.

Plant and arthropod communities are exposed to variable conditions in the environment, and multiple stressors may combine to produce novel effects (Darling and Cote 2008; Paine et al. 1998; Turner 2010). Drought, for example, may reduce the competitive ability of native plants against invasive plants that tolerate drought conditions (Boulant et al. 2008; Crous et al. 2012; Everard et al. 2010; Larios et al. 2013; Miller 1994; Schumacher et al. 2008). Drought also may reduce arthropod abundance or diversity by altering quality and availability of prey or host plants (Buchholz et al. 2013; Frampton et al. 2000; Kindvall 1995; Scheirs and De Bruyn 2005). Understanding how plant invasions combine with other disturbances to affect native plant and arthropod communities may improve our ability to implement restoration tools that are robust to variable conditions.

We developed a field-based experiment to test the efficacy of soil modification techniques for reducing dominance of OWBs (Chapter 3), and we also were interested in understanding how these restoration techniques would affect native arthropod communities. We predicted that if abundance and species richness of native plants increased where soils were modified, we would observe subsequent increases in abundance and species richness of arthropods, particularly for herbivores and

pollinators. We expected these changes to persist several years after treatment. In 2011, a severe drought occurred throughout the introduced distribution of OWBs (NDMC-UNL 2014), which persisted for several years in the southern portion of this range. This drought event provided us with the opportunity to explore the combined effects of plant invasion, drought severity, and the efficacy of soil and seeding treatments on arthropod communities. We predicted that the effects of soil modification techniques on native plants and OWBs in the absence of drought would be of greater magnitude than we would observe under drought conditions, and that species richness and abundance of arthropods would increase in both soil-modified and OWB-dominated areas as drought conditions subsided.

Methods

Study Area

We conducted our research at the Welder Wildlife Refuge (N 28.121155, W 97.442808), a 3,157-ha refuge located 12 km northeast of Sinton, southern Texas. The wildlife refuge represents an intermediate between the Gulf Coastal Prairie and Rio Grande Plain vegetative zones (Box 1961). Soils in the study are Victoria Clay, a typic Ustert common to the refuge several adjacent counties (USDA-SCS 1965). The soil is heavy, neutral to calcareous, with large concentrations of calcium, manganese, and sodium (USDA-SCS 1965), which binds essential nutrients like nitrogen and phosphorous from root systems (Brady and Weil 2004).

We selected a study area at the southernmost border of the refuge, which was classified historically as a mesquite-buffalograss community (Box 1961). The study area now is dominated by Kleberg bluestem (85-90% of total vegetation cover), with trace amounts ofl forbs (e.g., *Cienfuegosia drummondii*, *Ratibida columnifera*, and *Solanus elagnifolium*) and is bordered by a 2-m wide, disked, firebreak on one side and a fence on the other. We established 50, 6 x 9-m plots within the study area, with 1.5-m buffers between plots. The plot and buffer sizes were determined to permit maneuverability of disking equipment around and through the plots. Prior to treatment application, we collected soil samples in April 2011, which were analyzed for type and chemical composition (Texas Soil and Plant Lab, Edinburg, TX). We also estimated canopy cover of vegetation by species on two, 1-m^2 quadrats placed randomly within each plot in June 2011.

Treatment Application

In June 2011, we removed all standing vegetation in the treatment plots and disturbed the soil by disking with an off-set disk prior to treatment application. We applied 10 treatments: soil disturbance alone, decrease in pH, increase in pH, decrease in available N, and increase in mycorrhizal fungi as well as each in combination with a native seed mix. Soil disturbance consisted only of disking with the off-set disk, and all treatment plots were disked before and after treatments were applied. To reduce pH, we applied 731.6 kg/ha of water-soluble sulfur (Disper-Sul 90% elemental sulfur) in pellet form. To increase pH, we applied 2,259.6 kg/ha of powdered lime (Austin White

Lime Co., $CaCO_3$). We determined additions based on pre-treatment soil analyses and added 33% to initial calculations to ensure sufficient changes in pH to below 5 or above 9. To decrease nitrogen, we applied 1,360.8 kg/ha of sucrose ($C_{12}H_{22}O_{11}$; Alpert 2010). To augment the mycorrhizal fungal community, we applied 10.5 kg/ha of MycoGrow™ micronized endo/ecto seed mix (Appendix D), commercially available from Fungi Perfecti LLC (Olympia, WA); we mixed the inoculants with a small amount of soil for even distribution. We planted a mixture of native seeds on 25 of the 50 treatment plots in June 2011, at a rate of 13.0 kg/ha of pure live seed (pls), using a native seed drill (Truax Flex III). The species and quantities included in the seed mix were based on native plants observed during pre-treatment sampling, as well as native plants selected by the South Texas Natives (Kingsville, TX, Appendix E). We randomly assigned treatment combinations to plots and established 5 replicates of each for a total of 50 plots. All plots were disked multiple times after treatment application to mix soils evenly. In addition, we established 5 plots at random in an undisturbed part of the OWB monoculture to serve as a reference. Kleberg plots were at least 110 m away from treated plots.

Vegetation Sampling

We measured species richness and canopy cover on two, 1-m^2 quadrats per plot for every month in the summer 2011-2013. We placed quadrats at random within each plot for each sampling period, but quadrats always were at least 1 m from plot boundaries to avoid edge effects. All plants were identified to species using Everitt et al.

(2011) for grasses, Everitt et al. (2002) for woody plant species, and Everitt et al. (1999) for herbaceous plant species, and cross-referenced with type specimens in the Welder Wildlife Foundation herbarium.

We estimated horizontal canopy cover (≤ 1-m tall) by species, as well as cover of bare ground and litter (vegetative material separate from living vegetation or growing structures attached to the ground). We then grouped plant species into specific cover classes: grasses, forbs (herbaceous plants), and woody plants. We used species richness and canopy cover of plants as measures of community richness and vegetation composition.

Soil Sampling

In May of each field season, we sampled 2 L of soil from each plot at the beginning of each field season to determine soil chemistry. We collected soil up to a depth of 15 cm in each quadrat and combined samples from quadrats within plots. Soil samples were analyzed by Texas Plant and Soil Labs (Edinburg, TX) to determine soil pH, as well as available nitrogen (NO_3) using an extractable CO_2 method (McGeorge and Breazeale 1931; Texas Plant and Soil Labs 2012). We used these soil characteristics to assess treatment efficacy.

Arthropod Sampling

We sampled arthropods within the same 1-m^2 quadrats where we sampled vegetation in each plot. Although a variety of methods are used to sample arthropods,

each method is taxonomically biased to some degree (Greenslade 1964; Southwood 1982, Standen 2000). In an attempt to sample the arthropod community completely, we used three techniques: pitfall traps, vacuum sampling, and Berlese-Tullgren funnels. We started sampling arthropods 24 hours after we completed vegetation sampling and waited at least 24 hours between each technique to allow the arthropod community to recover.

We placed two pitfall traps (266-ml plastic cups) randomly within each quadrat, ensured that pitfall traps were flush with the soil surface and filled traps halfway with propylene glycol (Prestone Low Tox® Antifreeze/Coolant). We left the traps undisturbed for 24 hours, after which we collected the contents of all traps. We used a vacuum sampler (Model 122, Rincon-Vitoca Insectaries, Ventura, CA) to sample each quadrat for 90 seconds and transferred specimens to a plastic bag. We removed specimens attached to the net with an aspirator (BioQuip model 1135A, Rancho Dominquez, CA). To prevent or reduce predation, we placed cotton balls soaked with ethyl acetate in the plastic bag. Finally, we used Berlese-Tullgren funnels (BioQuip model 2845) and decreased the diameter of the mesh filter (0.32 x 0.32 cm) from the original model to keep soil particles from falling into the collecting cup. We collected 473 ml of soil from each quadrat and placed the sample within the upper part of the funnel. Soil and funnels were exposed to sunlight for 48 hours to facilitate extraction.

We combined samples from all techniques within each quadrat to obtain more comprehensive estimates of the arthropod community (Southwood 1982) and

combined all quadrats within each plot for each month of sampling. We froze or stored all specimens in 70% ethyl alcohol for later sorting and identification. We identified all arthropods to family based on Krantz and Walter (2009) for mites, Richardson (1905) for isopods, Stockwell (1992) for scorpions, Summers (1979) for centipedes and millipedes, and Triplehorn and Johnson (2005) for insects and spiders. When possible, we identified arthropod families to morphospecies (Oliver and Beattie 1996; hereafter referred to as species) for greater taxonomic resolution. Specimens that could not be identified beyond family (e.g., all Acari, most Araneae) were not considered as separate species for analysis if other species had been identified from the same family.

We also assigned all arthropods to a single functional group that represented the role of that group in an ecosystem (Appendix A). We classified herbivores as arthropods that consume living vegetation as a majority of their diet. We classified pollinators as arthropods that consume pollen or nectar as a majority of their diet, or pollinate plants by consuming flowering parts of the plant (Triplehorn and Johnson 2005). We classified decomposers as arthropods that either consume dead animal or plant matter as a majority of their diet, or consume microorganisms (i.e., bacteria and fungi) and concentrate available nutrients in excrement (Brussaard 1997; Clarholm 1985). We classified predators as arthropods that consume other arthropods during at least part of their life cycle and we also included parasitoids in this group. We designated ants (family Formicidae) as their own functional group, as ants perform multiple roles in ecosystems (Brussaard 1997; Folgarait 1998; Triplehorn and Johnson 2005; Wilson 1987). We did

not assign immature or larval specimens to functional groups that had different life strategies than their adult morphs (e.g., Lepidoptera), due to a lack of taxonomic resolution; these specimens comprised <1% of all individuals sampled (Appendix H).

We used species richness and abundance of all arthropods and by functional groups as coarse measures of community structure and composition. We also examined presence and abundance of species; presence indicated that the plot provided habitat and abundance provided a measure of habitat quality.

Precipitation

We obtained precipitation data from a nearby weather station at the headquarters of the Welder Wildlife Refuge, approximately 7.2 km from the study area. We quantified monthly precipitation from October 1956 (from the start of the water year, October 1) until September 2013 and we compared annual precipitation during our study to the long-term annual mean to assess the severity of drought. Lags between rain events and arthropod responses are common (Frampton et al. 2000; Tanaka and Tanaka 1982); we quantified precipitation during the 2 – 4 week period prior to the start of each sampling period to better understand changes in the arthropod community (Frampton et al. 2000; Tanaka and Tanaka 1982). We used the Palmer Drought Severity Index (NCDC-NOAA 2014) as a measure of drought severity for each year of the study.

Data Analysis

We examined the effects of soil modification treatments on soil, vegetation, and arthropod characteristics using generalized linear mixed models. We included soil treatment, seeding with native plants, and year, a proxy for drought, as independent factors in all models and explored evidence for two-way interactions (soil treatment * seeding, year * soil treatment, and year * seeding). We removed interaction terms from models when $P > 0.1$, but retained all simple effects in final models. When appropriate, we accounted for repeated measurements and considered three possible covariance structures: no within-group covariance, compound symmetric, or first-order autoregressive, selecting the most appropriate covariance structure based on lowest AIC values. When necessary, we transformed response variables to meet assumptions. We used the appropriate distribution and link function for each response variable; we used a binomial distribution and logit link to analyze differences in presence, and Poisson distribution and log link to analyze differences in abundance. We used a quasi-likelihood method to test for overdispersion in the Poisson model when necessary (Ramsey and Schafer 2002; Zuur et al. 2009). All analyses were completed using the lme4, MASS, and nlme packages in R (Bates et al. 2014; Pinheiro et al. 2013; R Core Development Team 2013; Venables and Ripley 2002). We computed means and 95% confidence intervals for OWB-dominated plots (Kleberg) to make informal comparisons with treatment plots.

No vegetation grew in treatment plots during the first two months post-treatment due to lack of rain. As such, we did not analyze vegetation data for 2011 with

the exception of cover of bare ground and litter. We did not analyze cover of litter and woody plants for any other year because most values were zero. Instead, we computed means and 95% CIs for litter cover after 2011, to make informal comparisons.

We excluded ants from total arthropod abundance, due to random trap placement. We did not analyze richness or abundance for pollinator arthropods because pollinators represented <5% of all arthropods sampled (Appendix H). We examined changes in presence for species that occurred in 10 – 90% (40 – 360) of 400 total plot samples (i.e., 50 plots * 8 sampling periods), and changes in abundance for species that occurred in at least 25% of total plot samples. Arthropods that met the criteria for presence or abundance but were not observed in 2011 were only analyzed for 2012-2013, because presence of species may have been dependent on plant cover for habitat rather than the effect of drought. Therefore, we analyzed presence of 43 taxa (including 6 herbivores, 10 decomposers, 22 predators, and 5 ants), and abundance of 22 taxa (including 4 herbivores, 3 decomposers, 12 predators, and 3 ants). We explored only simple effects of soil treatment, seeding, and year in models for presence of arthropod species due to issues with convergence. We provide results from all analyses of arthropod presence and abundance by taxa, but in the text we focus on taxa that drive effects at the level of functional group.

Results

Precipitation

Total rainfall for the water year (October 1—September 30) measured 32.3 cm for 2011, 62.5 cm for 2012, and 69.1 cm for 2013, which was 36%, 69%, and 76% of the long-term average (90.2 cm), respectively. Most precipitation did not occur during our sampling periods (Fig. 4.1). We observed the highest precipitation during the sampling period in 2013 (2011 = 1.1 cm, 2012 = 12.0 cm, 2013 = 26.0 cm). We categorized magnitude of drought (based on PDSI) during each year of the study as extreme (<-4.00), moderate (-3.99 to -3.00), and none (-1.99 to 1.99) for 2011, 2012, and 2013, respectively (NCDC-NOAA 2014).

Soils

Soil pH differed by soil and seeding treatment, but the magnitude of some differences changed over time (Table 4.1). Adding lime increased soil pH immediately after treatment, relative to soil disturbance and Kleberg plots, but this effect did not persist in subsequent years (Fig. 4.2). In contrast, adding sulfur did not change soil pH at any time post-treatment, relative to soil disturbance and Kleberg plots (Fig. 4.2). Although we detected differences in soil pH based on seeding (Table 4.1), seeding only increased soil pH by 0.1 units (95% CI = 0.0 – 0.2).

Available nitrogen did not differ among soil treatments immediately after treatment, but became more variable over time (Tables 4.1 and 4.2). Adding carbon did

not alter available nitrogen relative to soil disturbance and Kleberg plots, but both carbon-treated and disturbance plots had more nitrogen than Kleberg plots one year post-treatment (Fig. 4.3). We observed more available nitrogen in plots with added lime (17.0 kg/ha, 8.3 – 34.8) and sulfur (7.1 kg/ha, 3.5 – 14.7) during moderate drought relative to disturbed plots (Table 4.2). Plots with seed had 2.7 kg/ha more available nitrogen (1.2 – 3.4) than plots without seed (Table 4.2).

Vegetation

Although we detected differences in bare ground and litter cover for some treated plots immediately after treatment in 2011 (Table 4.3), these differences were due to disking of plots prior to applying treatments and not changes in soil chemistry. Plots with added seed had 11% less bare ground (95% CI = 10.7 – 11.7) relative to plots without seed during extreme drought (Fig. 4.4). Plots with added seed also had 21% less litter cover (9.7 – 44.1) relative to plots without seed during extreme drought, except in plots treated with carbon, where litter cover did not differ based on seeding treatment (Fig. 4.4). In contrast, plots with seed had 12% more litter cover (6.0 – 17.5) relative to plots without seed during moderate drought, except in disturbed plots, where litter cover did not differ based on seeding treatment (Fig. 4.4). All treated plots had more cover of bare ground but less litter than Kleberg plots during extreme drought, and cover of both classes decreased in treated plots over time (Fig. 4.4).

We did not detect differences in vegetation composition and structure based on soil treatment (Table 4.3). Seeding did affect vegetation characteristics, but the

magnitude of several detected differences changed over time (Table 4.3). Dominance of OWBs was lower in plots with added seed relative to plots without seed as drought severity decreased; cover of OWBs on plots with seed was 26.6% (95% CI = 7.3 – 94.6) lower during moderate drought and 37.5% (17.1 – 82.1) lower when drought subsided (Fig. 4.5a). Adding seed also resulted in an increase in species richness of plants (1.2 species/m^2, 0.6 – 2.0) and cover of native grasses (3.9%, 0.4 – 9.0; Figs. 4.5b and 4.6). In addition, plots with added seed had more forb cover (12.7%, 5.2 – 29.8) relative to plots without seed when drought subsided (Fig. 4.5c). However, all plots with and without seed, had less OWB cover, more cover of native grasses and forbs, and more species of plants than Kleberg plots dominated by OWBs in both years post-treatment (Figs. 4.5 and 4.6).

Arthropods

We captured a total of 36,588 arthropods, representing 35 orders, 209 families, and 456 species (Appendix H). Although the number of arthropod species did not differ among soil treatments (Table 4.4), plots with added seed had 0.8 more species/m^2 (95% CI = 0.4 – 1.1) relative to plots without seed and the number of species increased over time (Fig. 4.7). Abundance of arthropods in treated plots did not differ from Kleberg plots during extreme drought, but all treated plots had fewer arthropods than Kleberg plots as drought severity decreased (Fig. 4.7). Plots treated with carbon had fewer arthropods (7.4 arthropods/m^2, 5.2 – 10.0) during moderate drought relative to disturbed plots and all other soil treatments (Fig. 4.7). Plots treated with sulfur had

fewer arthropods (25.7, 22.9 – 28.9) when drought subsided relative to disturbed plots, but abundance of arthropods on other treatment plots did not differ from disturbed plots (Fig. 4.7). Plots with added seed had more arthropods during moderate drought (21.1, 20.8 – 22.5), but fewer arthropods (17.3, 14.7 – 20.2) when drought subsided, relative to plots without seed (Fig. 4.7). Both the number of arthropods and arthropod species in treated plots were comparable to Kleberg plots during extreme drought (Fig. 4.7), despite the lack of living vegetation in treated plots.

Herbivores

Richness and abundance of herbivorous arthropods differed among soil and seed treatments, but the magnitude of differences among some treatments differed over time (Table 4.4). Plots treated with both lime and seed had fewer herbivore species (0.2, 0.1 – 0.3) relative to disturbed plots, but differences were relatively small (Fig. 4.8). Plots with added seed had 1.6 more herbivore species (0.6 – 3.3), but 4.8 fewer herbivores/m^2 (3.33 – 6.86) relative to plots without seed when drought subsided (Fig. 4.8). Plots treated with carbon had fewer herbivores (2.5, 1.5 – 4.2) relative to disturbed plots, but only during moderate drought (Fig. 4.8). As drought severity decreased, richness and abundance of herbivorous arthropods in treatment plots generally increased, but abundance of herbivores was lower on all treated plots than Kleberg plots (Fig. 4.8).

Of the six herbivore taxa studied, presence did not change in response to soil treatments (Table 4.5) and did not differ between treated and Kleberg plots (Table 4.6).

Presence of one herbivorous arthropod increased (*Corythucha* spp.) and another decreased (*Balclutha rubrostriata*) with seeding (Table 4.6). Presence of two herbivorous arthropods decreased (*Xyonysius californicus* and *Mecidea minor*) and one increased (Mochlozetidae) as drought subsided (Table 4.6).

Although we observed relatively few changes in presence, abundance of herbivore taxa did change with soil treatment, seeding, or both (Table 4.7). Changes in the abundance of herbivores we observed in treated plots were driven by an invasive leafhopper (*Balclutha rubrostriata*), which represented 33% of the individuals captured in this functional group (Appendix H). Plots with added seed had fewer individuals of *B. rubrostriata* relative to plots without seed during moderate drought (2.3 leafhoppers/m^2, 0.7 – 7.0) and when drought conditions subsided (18.4, 12.5 – 27.0), but all treated plots had fewer individuals than Kleberg plots (Table 4.8). Abundance of Mochlozetid mites differed slightly among soil and seeding treatments, but treated plots always had fewer mites than Kleberg plots (Table 4.8).

Decomposers

Richness of decomposers differed by soil treatment and abundance of decomposers differed by soil and seeding treatment, but the magnitude of differences changed over time (Table 4.4; Fig. 4.9). Disturbed plots had more species of decomposers than plots treated with fungi (1.1, 0.8 – 1.3) or lime (1.2, 1.0 – 1.6) during extreme drought, but as drought subsided, species richness of decomposers became more similar among treated plots (Fig. 4.9). Plots with added seed had more

decomposers/m^2 (19.1, 15.8 – 23.1) relative to plots without seed and Kleberg plots during moderate drought (Fig. 4.9). Treated plots also had more decomposers during moderate drought, relative to extreme drought (18.0, 17.2 – 18.7) and when drought subsided (3.8, 3.5 – 4.2; Fig. 4.9).

We did not detect differences in presence of decomposer taxa with soil treatments, but presence of some taxa differed with seeding or over time (Table 4.5). Of the 10 decomposer taxa studied, presence of two taxa (*Acanthinus scitulus* and *Gryllus* spp.) decreased and two taxa (*Armalia texana* and Galumnidae) increased with seeding (Table 4.6). Presence of decomposers generally decreased and did not differ from Kleberg plots as drought subsided (Table 4.6).

Abundance of decomposer taxa differed among seeding treatments and some taxa differed by soil treatment, and these differences often changed over time (Table 4.7). Changes in the abundance of decomposers were driven by pillbugs (*Armadillidium vulgare*; Table 4.8), which represented 47% of the individuals captured in this functional group (Appendix H). Plots treated with lime had more pillbugs during extreme drought conditions (1.7, 1.1 – 2.6) relative to disturbed plots (Table 4.8). Also, plots with seed had more pillbugs than plots without seed during extreme (1.86, 1.67 – 1.91) and moderate drought conditions (11.0, 10.3 – 11.4). All treated plots had more pillbugs than Kleberg plots after 2011 (Table 4.8).

Predators

Although we did not observe differences in species richness of predators among soil treatments (Table 4.4; Fig. 4.10), abundance of predaceous arthropods differed among soil treatments and seeding over time (Table 4.4). Abundance of predators was lower in plots with lime (4.6, 3.6 – 6.1) during extreme drought and lower in plots treated with sulfur (13.7, 11.7 – 16.4) in the absence of drought, relative to disturbed plots (Fig. 4.10). Plots with seed had more predators during extreme drought (6.6, 6.0 – 7.0), did not differ during moderate drought, and had fewer predators when drought subsided (6.8, 5.5 – 8.4), relative to plots without seed (Fig. 4.10). Plots with seed generally had more predators (2.6, 1.5 – 5.5) relative to Kleberg plots during extreme drought, but all treatment plots had fewer predators (9.6, 7.1 – 12.8) than Kleberg plots when drought subsided (Fig. 4.10).

In general, presence of predator arthropods did not change with soil treatments (Table 4.5), but nearly half of all predator taxa we observed in treated plots were absent or collected only once in Kleberg plots, including spiders (Order Araneae; Table 4.6). Presence of lynx spiders (*Oxyopes* spp.) increased with seeding (Table 4.6). Of the 22 predator taxa studied, presence of 11 taxa increased (e.g., *Vonones* spp.) and three taxa decreased (e.g., *Centruroides vittatus*) as drought subsided (Table 4.6).

Abundance of predators differed by soil treatment, seeding, or both, and the difference among treatments changed over time (Table 4.7). Changes in abundance of predators were driven by multiple dominant taxa; Anystid mites, thrips (*Aeolothrips*

spp.), and harvestmen (*Vonones* spp.) represented 28%, 14%, and 10% of all predators collected in treated plots, respectively (Appendix H), whereas the same taxa represented 29%, 21%, and 3% of all predators collected in Kleberg plots, respectively (Appendix B). Plots treated with lime had more mites (2.1, 1.6 – 2.8) than other treated plots during moderate drought, and plots treated with sulfur had fewer mites (2.8, 2.0 – 3.6) when drought subsided, but all treated plots had fewer mites than Kleberg plots as drought subsided (Table 4.8). Abundance of harvestmen increased as drought subsided, and plots with seed generally had more harvestmen during moderate drought (3.8, 1.2 – 13.4) and when drought subsided (3.5, 1.2 – 11.8), relative to Kleberg plots (Table 4.8). Plots with seed also had fewer thrips compared to treated plots (8.6, 6.4 – 11.6) and Kleberg plots (13.6, 4.0 – 22.3) as drought subsided (Table 4.8).

Ants

Species richness and abundance of ants differed with soil treatments and seeding and these effects changed over time (Table 4.5). Although we detected a seeding effect on species richness of ants (Table 4.4), the difference between plots with and without seed was relatively small (0.4, 0.2 – 0.5; Fig. 4.11). Treated plots had more ant species during extreme drought (0.7, 0.5 – 0.8) relative to other years, but again, differences were small (Fig. 4.11). We observed more ants in plots treated with carbon (13.4, 9.1 – 19.6), fungi (6.3, 3.5 – 11.3), and lime (7.4, 4.1 – 13.4) relative to disturbed plots during extreme drought, and more ants in plots treated with sulfur (20.5, 16.8 – 25.0) during moderate drought, but fewer ants in all other treatment plots compared to

disturbed plots when drought subsided (Fig. 4.11). Plots with added seed had more ants relative to plots without seed during moderate drought (2.6, 2.4 – 2.8) and non-drought conditions (3.0, 2.7 – 3.3; Fig. 4.11). Plots with added seed generally had more ants relative to Kleberg plots (Fig. 4.11).

Presence of ant species generally did not change in response to soil treatment or seeding, but some did change with drought severity (Table 4.5). Of the five ant taxa considered, presence of two species (*Forelius mccooki* and *Solenopsis invicta*) decreased as drought conditions subsided (Table 4.6). The presence of one species (*Forelius pruinosus*) decreased during moderate drought, but increased during non-drought conditions (Table 4.6). Presence of all native ants was higher in treated plots relative to Kleberg plots during extreme drought, but presence of all ant species, except native fire ants (*Solenopsis geminata*), were comparable among treated plots as drought conditions subsided (Table 4.6).

Abundance of ants changed with soil treatment, seeding, or both, and the difference among treatments changed over time (Table 4.7). Although *Forelius mccooki* represented 52% of all ants collected in treated plots, we could not analyze abundance because the species was observed infrequently (Appendix H). We may have captured large densities of *F. mccooki* if traps were close to ant nests. Fire ants (*Solenopsis* spp.) responded to changes in soil and seeding treatments and represented <30% of all ants collected (Table 4.7; Appendix H). We observed more native fire ants (*S. geminata*) in plots with lime (2.6, 1.2 – 5.3) during extreme drought, more in plots with sulfur (5.7,

3.7 – 9.0) during moderate drought, and fewer in plots with carbon when drought subsided (1.5, 1.0 – 2.5), relative to disturbed plots (Table 4.8). Plots with added seed also had more ants (3.4, 2.8 – 3.7) than plots without seed when drought subsided, and all treated plots had more ants than Kleberg plots during drought (Table 4.8). Plots with seed had fewer invasive fire ants (*S. invicta*, 0.7, 0.3 – 1.2) than plots without seed during moderate drought, and plots treated with lime had more fire ants (1.9, 1.2 – 3.1) than disturbed plots as drought subsided (Table 4.8). Treated plots generally had fewer ants than Kleberg plots as drought severity decreased (Table 4.8).

Discussion

Arthropod communities may provide reliable indicators of restoration success because arthropods respond quickly to changes in plant and soil communities (Bennett 2010; Burger et al. 2003; de Buryn 1999; Haimi 2000; Mayer et al. 2007; Samways 1996; Snyder and Hendrix 2008). Because diversity of native plants often is associated with diversity of native arthropods (Bernays and Graham 1988; Niemela and Mattson 1996; Wu et al. 2009), we assumed that by restoring native plant communities, we would facilitate the restoration of native arthropod communities. Although we documented increased species richness and cover of native plants along with reduced dominance of invasive plants, responses of arthropods to soil treatments and seeding differed based on drought severity and were taxa-specific.

In general, changes in the richness and abundance of arthropods were associated with decreasing drought severity rather than changes in soil chemistry and seeding. In contrast, the composition of arthropods was more likely to change in response to soil and seeding treatments, and we observed changes in the dominance of arthropod taxa in soil and seeding treatments over time. Certain arthropod species (e.g., *Armadillidium vulgare*) may be present in both plant communities, but at low densities, and changes in the characteristics of soil and plants associated with restoration treatments may have altered the dominance and composition of arthropods in areas where invasive plants once were dominant. Soil modification treatments may promote evenness of arthropods in certain functional groups instead of increasing richness.

Herbivores

Shifts in soil properties following disturbance (e.g., plant invasion) may affect herbivore communities indirectly through changes in the availability or diversity of host plants (Bennett 2010; Haddad et al. 2000; Niemela and Mattson 1996; Wardle et al. 2004; Wu et al. 2009). For example, increased available nitrogen reduces species richness of plants, which in turn results in decreased richness of herbivorous insects (Haddad et al. 2000). Mycorrhizae can increase nutrient acquisition and growth of plants, which may affect plant palatability for herbivorous arthropods (Gange and Nice 1997; Gange et al. 2002; Gange and Smith 2005; Goverde et al. 2000). Although we did not observe changes in species richness of plants or herbivorous arthropods with carbon or mycorrhizal treatments, we observed more plant species in treatment plots relative

to Kleberg plots, and more herbivore species in plots with added seed as drought conditions subsided. However, we observed fewer herbivores in treated plots relative to Kleberg plots after 2011, and the difference in herbivore abundance among plant communities may be related to host preferences instead of changes in soil chemistry.

Specialist herbivores, such as true bugs (Hemiptera), butterflies (Lepidoptera), thrips (Thysanoptera), and some beetles (Triplehorn and Johnson 2005) may be affected negatively by changes in plant composition that result from invasive plants (Burghardt et al. 2010; Litt et al., in press; Tallamy 2004). However, generalist herbivores may utilize invasive plant species as food (Tallamy 2004; Tallamy et al. 2010), resulting in increased abundance if the invasive plant also is abundant. We observed high densities of red-streaked leafhoppers (*Balclutha rubrostriata*), an invasive leafhopper that may be associated with OWB grasses (Morgan et al. 2013; Woodin et al. 2010; Zahniser et al. 2010), in Kleberg plots as drought subsided, but lower densities in treated plots. We also observed many more plant-feeding mites (Mochlozetidae) in Kleberg plots than in treated plots. Both taxa appear to use OWBs as habitat, and habitat quality may increase for both taxa as OWBs become dominant. The dominance of these generalist herbivores may have reduced the benefits of increased species richness and cover of native plants for other herbivores in our treatment plots.

Although herbivores were not as abundant in treatment plots in relative to Kleberg plots, changes in plant composition associated with reduced dominance of OWBs in treatment plots may have allowed for increases in herbivore diversity. We

could not analyze data for all herbivores in our study, but we recorded more than three times as many species of herbivores in treatment plots than in Kleberg plots (Appendices B and H). The number of herbivore species observed in treatment plots was comparable to Kleberg plots, but the composition of herbivore communities in treated plots varied over time with changes in composition of plants. In contrast, herbivore communities in Kleberg plots were dominated by only two taxa. Decreases in diversity of herbivores with plant invasion may reduce habitat quality for grassland birds and other species that rely on this functional group as a major source of food (Wiens and Rotenberry 1979; Woodin et al. 2010).

Decomposers

Of all arthropods, detritus and fungal-feeding arthropods may be the most sensitive to soil treatments, as changes in soil characteristics could alter arthropod abundance, composition, or habitat quality (Alerding and Hunter 2013; Haddad et al. 2000; Haimi 2000; Kappes et al. 2007; Meyer et al. 2005; Wolkovich et al. 2009). Changes in soil pH, for example, can alter the abundance and composition of springtails (Alerding and Hunter 2013; Geissen et al. 1997; Haimi et al. 2000; Haimi and Mätäsniemi 2002; Liiri et al. 2002; McGrath and Binkley 2009), mites (Liiri et al. 2002; Maraun and Scheu 2000; McGrath and Binkley 2009; Wardle et al. 2004), and pillbugs (van Straalen and Verhoef 1997; Zimmer et al. 2000). Although changes in soil pH in lime-treated plots did not persist after the first year, we observed fewer species of decomposers in these plots than in disturbed or Kleberg plots, and increasing soil pH may reduce richness of

decomposers in this system. Increasing available nitrogen also can increase the number of detritivore species (Haddad et al. 2000; Haimi and Mätäsniemi 2002; McGrath and Binkley 2009). Although we did not observe more species of detritivores following an increase in available nitrogen in treated plots in 2012, we did observe increased abundance of decomposers in treated plots relative to Kleberg plots. However, changes in composition of the decomposer community associated with changes in available nitrogen may be related to changes in the composition and abundance of plant litter, rather than soil treatment alone.

Plant litter may influence communities of decomposer arthropods, as increased abundance of litter and rates of decomposition often are associated with plant invasion (Gratton and Denno 2006; Kappes et al. 2007; Levin et al. 2006; Standish et al. 2004; Wolkovich et al. 2009). We observed increased abundance of decomposer arthropods following increases in litter cover and species of plants in treatment plots during moderate drought. Although litter cover in seeded plots during moderate drought was comparable to litter cover in Kleberg plots the previous year, we observed more decomposers in seeded plots in 2012 than Kleberg plots in both years. Given that litter from multiple plant species can increase the quality of habitat for decomposers (Kappes et al. 2007; Wardle et al. 2004), litter diversity may be more important for detritivores rather than litter abundance.

Differences in abundance of decomposers between soil treatments were driven mainly by pillbugs (*Armadillidium vulgare*). *Armadillidium vulgare* is an invasive isopod

that can accelerate rates of decomposition and replace native detritivores (David and Handa 2010; Ellis et al. 2000; Frouz et al. 2008; Singer et al. 2012). Abundance of *A. vulgare* increased during moderate drought, which drove the increase in abundance of decomposers in treatment plots, particularly in plots with added seed. Although *A. vulgare* is sensitive to changes in soil pH and prefers near neutral soils (van Straalen and Verhoef 1997; Zimmer et al. 2000), we did not observe changes in abundance of *A. vulgare* in plots where we changed soil pH. Plots with seed had more litter cover relative to plots without seed during moderate drought, and treatment plots had more litter and native plant cover relative to Kleberg plots. Even when Kleberg plots had more litter than treatment plots, we did not observe differences in abundance of decomposers or of *A. vulgare* specifically among plant communities. In addition to plant litter, *A*. *vulgare* also may consume seeds when litter is scarce (Saska 2008), and dormant seeds in the seed bank may have attributed to increased abundance of *A. vulgare*. The efficacy of seeding may be affected negatively by the abundance of pillbugs.

Predators

Changes in abundance or diversity of prey associated with changes in the characteristics of plants and soils may affect predatory arthropods. For example, changes in soil pH resulting from invasion of garlic mustard (*Alliaria petiolata*) are associated with a decrease in abundance of some springtail families, which result in predators shifting their diets from detritivores to herbivores (Alerding and Hunter 2013; deHart and Strand 2012). Although we observed fewer predators in plots where soil pH

increased in 2011, we do not think this type of dietary shift was likely, given the lack of vegetation that year. In contrast, ants were more abundant in plots immediately after treatment with lime, and ants may have competed with predators and decomposers for food.

Changes in plant and soil characteristics that affect the diversity of prey also may affect parasitoids negatively (Gange et al. 2003; Guerrieri et al. 2004; Haddad et al. 2000; Simao et al. 2010). Where plants are present, increases in available plant nitrogen decreases species richness of both plants and herbivores, with concomitant effects on species richness of parasitoids (Haddad et al. 2000). Although we observed an increase in available nitrogen in treated plots after drought subsided, associated with increases in presence of many parasitoid taxa (e.g., Mymarmidae), these changes in presence may be associated with increased plant species richness with decreased drought severity rather than availability of soil nutrients. Mycorrhizal fungi may alter plant traits, such as the release of volatile chemicals, which may also increase the ability of parasitoids to detect herbivorous prey (Gange et al. 2003; Guerrieri et al. 2004). We did not, however, observe changes in the composition or abundance of parasitoids in plots treated with mycorrhizal fungi relative to other treated plots.

Although some soil treatments increased the richness and abundance of predators, all treated plots had fewer predators than Kleberg plots as drought subsided. We observed an abundance of mites in Kleberg plots, which may have contributed to increased abundance of predators that actively feed on mites, such as predatory thrips

(e.g., *Haplothrips* spp.; Triplehorn and Johnson 2005). Most generalist predators, such as spiders, consume a variety of prey (Nyffeler 1999) and did not respond to changes we observed in plant composition or structure. Lynx spiders (*Oxyopes* spp.) were more likely to occur in plots with seed relative to plots without seed, which may be a function of concurrent increases in abundance of *Pseudatomoscelis seriatus*, its preferred prey, in plots with seed (Nyffeler et al. 1992). Plots with added seed had more species of plants and herbivores as drought subsided, which may result in increased habitat quality for generalist predators like *Oxyopes* spp. In contrast, spiders were relatively rare in Kleberg plots. A lack of prey diversity may be responsible for the observed decrease in generalist predators, and the homogenization of plant and prey communities may have large effects on food web dynamics (Gratton and Denno 2006; Hansen et al. 2009).

Ants

Ants inhabit the soil and influence ecosystem processes belowground, and changes in soil characteristics may affect ant communities directly (Brussaard 1997; Cammeraat et al. 2002; de Bruyn 1999; Dostal et al. 2005; Folgarait 1998; Whitford et al. 2012). Changes in soil pH can alter abundance of arthropod prey for predatory ants (deHart and Strand 2012). Although we did not observe changes in ant abundance immediately following changes in soil pH, ants may have been competing with decomposers and predators for food where the latter functional groups were less abundant. In our study, we found that ant abundance increased immediately after adding carbon (sucrose) and mycorrhizae; these treatments may provide a food source

for ants. Seed predators, such as harvester ants (*Pogonomyrmex* spp.) may impede the efficacy of seeding as a restoration treatment (Díaz 1992; Fisher and Cover 2007; MacDougall and Wilson 2007; Retana et al. 2004). We observed harvester ants exclusively in plots with seed (Appendix H), and ants may have been consuming seeds from our seeding treatments rather than those in the native seed bank. Efficacy of soil modification and seeding treatments may be impeded where ants are abundant, but treatments may benefit ant communities in the short term.

Conclusions

We examined alternative approaches to restoring coastal prairies impacted by an invasive grass and drought and observed reduced dominance of the invasive plant and increased cover of native plants, with subsequent changes in the community of native arthropods. Mechanical soil disturbance and seeding with locally-adapted plant species increased diversity and cover of native plants, relative to undisturbed monocultures of invasive plants; the effects of soil disturbance and seeding persisted or increased over time. Although the richness and abundance of arthropods in treated plots increased over time, treated plots had fewer arthropods than monocultures of invasive plants. Monitoring the efficacy of soil treatments over longer time periods or during different seasons may reveal positive effects for native arthropod communities or functional groups, such as pollinators. In addition to soil composition and drought, some arthropods, such as seed predators, inhibited our efforts to modify soil and plant characteristics. Sampling arthropod communities prior to treatments can help

determine which restoration tools would be most effective. Nonnative arthropods were dominant in both treated and Kleberg plots, and the diversity and abundance of native arthropod communities may have already been altered by these nonnative arthropods prior to plant invasion. Additional tools may be needed to reduce the effects of arthropod invasion even after restoration techniques have increased native plant diversity. Shifts in the composition of the arthropod community as a result of plant invasion and drought can alter food availability for native grassland fauna (Burghardt et al. 2008; Hickman et al 2006; Litt and Steidl 2010; Woodin et al. 2010; Wiens and Rotenberry 1979). Therefore, management strategies that aim to restore habitat for organisms at other trophic levels will likely require an understanding of how arthropod communities respond to changes in plant and soil characteristics following multiple disturbances.

Tables

Table 4.1. Factors affecting soil characteristics, southern Texas, summers 2011-2013.

Soil Variable	Soil Treatment		Seed		Year		Soil*Year[a]	
	$F_{4,44}$	*P*	$F_{1,44}$	*P*	$F_{2,90}$	*P*	$F_{8,90}$	*P*
Soil pH	27.25	<0.001	5.07	0.029	7.06	0.001	4.86	<0.001
Available NO_3	0.91	0.469	5.66	0.022	2.08	0.131	3.06	0.004

[a] We did not detect interactions between seeding and soil treatment or seeding and year.

Table 4.2. Available NO_3 (kg/ha, median and back-transformed 95% CIs), southern Texas, summers 2011-2013.

		Year		
Soil Treatment	**Seeding**	**2011**	**2012**	**2013**
Disturbance	Without Seed	14.7	10.5	21.2
		(8.5-25.4)	(5.1-21.5)	(10.4-43.5)
	With Seed	19.1	13.6	27.5
		(15.3-23.8)	(6.5-28.5)	(13.4-56.4)
Carbon	Without Seed	9.1	13.2	17.1
		(6.1-13.5)	(6.3-27.6)	(10.3-28.3)
	With Seed	11.8	17.1	22.1
		(9.5-14.7)	(8.3-35.0)	(13.3-36.7)
Fungi	Without Seed	15.5	11.6	17.5
		(9.0-26.7)	(5.7-23.8)	(8.5-35.8)
	With Seed	20.1	15.1	22.6
		(16.1-25.0)	(7.4-30.9)	(11.0-46.4)
Lime	Without Seed	14.9	27.5	13.1
		(8.6-25.7)	(13.4-56.3)	(6.4-26.9)
	With Seed	19.3	35.6	17.0
		(15.5-24.1)	(17.4-72.9)	(8.3-34.9)
Sulfur	Without Seed	11.7	17.6	13.6
		(6.8-20.2)	(8.6-36.2)	(6.6-27.9)
	With Seed	15.2	22.9	17.6
		(12.2-19.0)	(11.2-46.9)	(8.6-36.1)
OWB	Reference	9.5	3.5	5.4
		(5.5-16.6)	(2.0-6.0)	(2.9-10.1)

Table 4.3. Factors affecting vegetation characteristics, southern Texas, summers 2012-2013.

Vegetation Variable	Soil Treatment		Seed		Year		Soil*Seed[†]		Seed*Year[a]	
	$F_{4,44}$	*P*	$F_{1,44}$	*P*	$F_{1,249}$	*P*	$F_{4,40}$	*P*	$F_{1,248}$	*P*
Bare Ground[†]	0.39	0.818	5.43	0.024						
Litter[†]	0.71	0.587	36.72	<0.001			2.37	0.069		
Richness	1.28	0.294	20.79	<0.001	8.89	0.003				
OWB Density	0.38	0.820	49.18	<0.001	8.66	0.004			7.56	0.006
Height	0.21	0.934	67.22	<0.001	0.26	0.613			28.94	<0.001
Bare Ground	1.84	0.137	15.32	<0.001	76.00	<0.001			39.03	<0.001
Native Grasses	0.60	0.663	20.21	<0.001	30.96	<0.001				
OWB Grasses	0.35	0.843	53.00	<0.001	4.20	0.042			9.47	0.002
Forbs	0.17	0.951	4.44	0.041	0.00	0.975			5.63	0.018

[†]2011 only

Table 4.4. Factors affecting arthropod characteristics, southern Texas, summers 2011-2013.

Arthropod Variable	Functional Group	Soil Treatment		Seeding		Year		Soil*Seed		Soil*Year		Seed*Year	
		$F_{4,44}$	*P*	$F_{1,44}$	*P*	$F_{2,348}$	*P*	$F_{4,40}$	*P*	$F_{8,338}$	*P*	$F_{2,338}$	*P*
Richness	Total	1.05	0.391	6.26	0.016	94.44	<0.001						
	Herbivores	1.11	0.365	3.06	0.088	131.21	<0.001	2.97	0.031			2.56	0.07
	Decomposers	0.99	0.422	0.32	0.577	55.42	<0.001			2.027	0.043		
	Predators	0.85	0.502	0.42	0.523	92.33	<0.001						
	Ants	1.71	0.165	5.59	0.023	5.90	0.003						
Abundance	Total	0.48	0.750	0.52	0.475	690.91	<0.001			20.47	<0.001	234.73	<0.0(
	Herbivores	0.08	0.988	0.52	0.475	375.36	<0.001			21.28	<0.001	64.62	<0.0(
	Decomposers	0.21	0.932	27.12	<0.001	807.86	<0.001					100.73	<0.0(
	Predators	1.00	0.418	0.33	0.569	109.29	<0.001			15.38	<0.001	52.77	<0.0(
	Ants	2.31	0.073	3.29	0.077	122.14	<0.001			386.11	<0.001	48.23	<0.0(

Table 4.5. Factors affecting presence of arthropod species (*n* = 400 samples, 50 plots*8 sampling periods), southern Texas, summers 2011-2013.

Functional Group	Order	Family	Species	Soil Treatment		Seeding		Year	
				$F_{4,44}$	*P*	$F_{1,44}$	*P*	$F_{2,348}$	*P*
Herbivores	Hemiptera	Cicadellidae	*Balclutha rubrostriata**	0.08	0.988	3.19	0.081	0.36	0.700
		Lygaeidae	*Xyonysius californicus**	0.37	0.829	0.03	0.863	15.42	<0.001
		Miridae	*Pseudatomoscelis seriatus**	0.73	0.576	0.62	0.435	0.72	0.487
		Pentatomidae	*Mecidea minor**	0.45	0.772	0.22	0.641	13.36	<0.001
		Tingidae	*Corythucha* spp.*	0.20	0.937	10.08	0.003	0.03	0.970
	Sarcoptiformes	Mochlozetidae		0.01	1.00	0.59	0.447	49.22	<0.001
Decomposers	Blattodea	Blattellidae	*Blattella vaga*	0.29	0.883	0.00	1.000	5.74	0.004
	Coleoptera	Anthicidae	*Acanthinus scitulus*	0.70	0.596	7.03	0.011	7.44	0.001
		Latridiidae	*Melanophthalma* spp.*	0.65	0.630	1.61	0.211	0.26	0.771
		Tenebrionidae	*Armalia texana*	0.54	0.707	26.63	<0.001	3.43	0.033
	Diptera	Chloropidae	*Liohippelates* spp.*	0.73	0.576	1.51	0.223	8.57	<0.001
		Phoridae	*Megaselia* spp.	0.57	0.686	1.49	0.229	11.15	<0.001
		Scatopsidae	Unknown sp*	0.89	0.478	0.57	0.454	11.20	<0.001
	Orthoptera	Gryllidae	*Gryllus* spp.	0.78	0.544	4.93	0.032	0.59	0.555
	Psocoptera	Liposcelidae	*Liposcelis* spp.	0.22	0.926	0.25	0.620	40.78	<0.001
	Sarcoptiformes	Galumnidae		0.47	0.757	14.29	<0.001	11.32	<0.001
Predators	Araneae	Anyphaenidae	*Hibana* spp.	0.96	0.439	1.06	0.309	0.81	0.446
		Araneidae	*Araneus* spp.*	1.68	0.172	0.02	0.888	0.60	0.549
		Linyphiidae		1.05	0.393	2.14	0.151	0.10	0.905
		Oxyopidae	*Oxyopes* spp.	0.97	0.434	19.39	<0.001	7.30	0.001
		Salticidae	*Phiddipus* spp.*	0.98	0.428	0.09	0.766	0.36	0.700
		Thomisidae	*Misumena* spp.	1.16	0.341	5.33	0.026	0.11	0.896
	Coleoptera	Carabidae	*Dromochorus welderensis**	1.06	0.388	1.54	0.221	0.02	0.980
	Hymenoptera	Bethylidae	*Pristocera hyaline*	1.00	0.418	0.08	0.779	11.61	<0.001
		Encyrtidae		0.50	0.736	1.16	0.287	6.50	0.002
		Mymarmidae		1.83	0.140	0.77	0.385	5.59	0.004
		Scelionidae	*Telenomus* spp.*	0.26	0.902	0.33	0.569	0.08	0.923
			Trissolcus spp.	1.43	0.240	0.11	0.742	6.49	0.002

(Table 4.5 continued)

Functional Group	Order	Family	Species	Soil Treatment		Seeding		Year	
				$F_{4,44}$	*P*	$F_{1,44}$	*P*	$F_{2,348}$	*P*
Predators	Hymenoptera	Trichogrammatidae		1.35	0.267	6.76	0.012	6.27	0.002
	Opiliones	Cosmetidae	*Vonones* spp.	0.31	0.870	1.79	0.188	34.82	<0.001
	Scorpiones	Buthidae	*Centruroides vittatus*	1.07	0.383	1.50	0.227	8.50	<0.001
	Scutigeromorpha	Scutigeridae	*Scutiger coleopteran*	0.26	0.902	0.96	0.333	2.17	0.116
	Thysanoptera	Aeolothripidae	*Aeolothrips* spp.*	0.02	1.00	0.15	0.700	80.50	<0.001
		Phlaeothripidae	*Haplothrips* spp.	0.97	0.434	0.12	0.731	17.79	<0.001
		Thripidae	*Scolothrips* spp.	0.32	0.863	3.71	0.061	20.28	<0.001
	Trombidiformes	Bdellidae		1.00	0.418	0.08	0.779	11.61	<0.001
		Erythraeidae		1.00	0.418	0.25	0.620	3.03	0.050
		Smarididae*		1.53	0.210	1.55	0.220	10.24	<0.001
Ants	Hymenoptera	Formicidae	*Forelius mccooki*	0.03	1.000	0.19	0.665	61.42	<0.001
			Forelius pruinosus	0.26	0.902	0.42	0.520	17.21	<0.001
			Nylanderia terricola	0.92	0.461	0.01	0.921	2.88	0.057
			Solenopsis geminata	0.28	0.889	0.52	0.475	0.98	0.376
			Solenopsis invicta	1.07	0.383	0.93	0.318	22.39	<0.001

*2012-2013 data only

Table 4.6. Presence of arthropod species (median values and back-transformed 95% CIs), southern Texas, summers 2011-2013. We provide means and 95% CIs for presence of arthropod species and multiple estimates when we detect differences among years.

Functional Group	Order	Family	Species	Year	Seeding		OWB
					Without	With	
Herbivores	Hemiptera	Cicadellidae	*Balclutha rubrostriata**		0.43 (0.29-0.59)	0.33 (0.24-0.44)	0.57 (0.30-0.84)
		Lygaeidae	*Xyonysius californicus**	2012	0.25 (0.12-0.45)	0.24 (0.13-0.39)	0.13 (0.00-0.31)
				2013	0.01 (0.00-0.04)	0.01 (0.00-0.04)	0.00
		Miridae	*Pseudatomoscelis seriatus**		0.35 (0.21-0.51)	0.39 (0.28-0.52)	0.13 (0.00-0.31)
		Pentatomidae	*Mecidea minor**	2012	0.22 (0.11-0.39)	0.25 (0.14-0.40)	0.33 (0.09-0.58)
				2013	0.06 (0.03-0.12)	0.07 (0.03-0.14)	0.33 (0.09-0.58)
		Tingidae	*Corythucha* spp.*		0.05 (0.02-0.14)	0.17 (0.08-0.33)	0.00
	Sarcoptiformes	Mochlozetidae		2011	0.09 (0.06-0.20)	0.05 (0.03-0.09)	0.70 (0.40-1.00)
				2012	0.69 (0.49-0.83)	0.55 (0.36-0.74)	1.00
				2013	0.90 (0.79-0.96)	0.84 (0.68-0.93)	1.00
Decomposers	Blattodea	Blattellidae	*Blattella vaga*	2011	0.28 (0.16-0.44)	0.16 (0.11-0.24)	0.97 (0.76-1.00)
				2012	0.52 (0.38-0.67)	0.36 (0.23-0.50)	0.70 (0.20-0.95)
				2013	0.15 (0.08-0.26)	0.08 (0.04-0.15)	0.26 (0.03-0.80)

(Table 4.6 continued)

Functional Group	Order	Family	Species	Year	Seeding		OWB
					Without	**With**	
Decomposers	Coleoptera	Anthicidae	*Acanthinus scitulus*	2011	0.24 (0.11-0.45)	0.10 (0.06-0.16)	0.45 (0.23-0.70)
				2012	0.08 (0.04-0.15)	0.03 (0.01-0.06)	0.16 (0.05-0.40)
				2013	0.08 (0.04-0.17)	0.03 (0.01-0.6)	0.14 (0.04-0.36)
		Latridiidae	*Melanophthalma* spp.*		0.26 (0.15-0.41)	0.20 (0.13-0.30)	0.33 (0.09-0.58)
		Tenebrionidae	*Armalia texana*	2011	0.10 (0.05-0.22)	0.42 (0.27-0.59)	0.00
				2012	0.10 (0.05-0.17)	0.40 (0.26-0.57)	0.00
				2013	0.05 (0.02-0.09)	0.23 (0.13-0.39)	0.00
	Diptera	Chloropidae	*Liohippelates* spp.*	2012	0.20 (0.09-0.38)	0.13 (0.07-0.24)	0.43 (0.16-0.70)
				2013	0.08 (0.04-0.14)	0.05 (0.02-0.10)	0.50 (0.23-0.77)
		Phoridae	*Megaselia* spp.	2011	0.40 (0.25-0.58)	0.31 (0.20-0.44)	0.31 (0.14-0.57)
				2012	0.19 (0.11-0.31)	0.14 (0.08-0.23)	0.28 (0.10-0.57)
				2013	0.11 (0.06-0.21)	0.08 (0.04-0.15)	0.31 (0.12-0.61)
		Scatopsidae	Unknown spp. *	2012	0.19 (0.10-0.33)	0.23 (0.14-0.35)	0.20 (0.00-0.41)
				2013	0.39 (0.22-0.58)	0.45 (0.31-0.60)	0.20 (0.00-0.41)
	Orthoptera	Gryllidae	*Gryllus* spp.		0.41 (0.27-0.56)	0.30 (0.23-0.39)	0.73 (0.49-0.89)

(Table 4.6 continued)

Functional Group	Order	Family	Species	Year	Seeding		OWB
					Without	With	
Decomposers	Psocoptera	Liposcelidae	*Liposcelis* spp.	2011	0.56 (0.06-0.72)	0.64 (0.51-0.76)	0.35 (0.16-0.60)
				2012	0.12 (0.07-0.20)	0.16 (0.09-0.26)	0.20 (0.17-0.59)
				2013	0.05 (0.02-0.11)	0.07 (0.03-0.15)	0.30 (0.11-0.61)
	Sarcoptiformes	Galumnidae		2011	0.02 (0.01-0.06)	0.05 (0.03-0.08)	0.45 (0.22-0.70)
				2012	0.18 (0.07-0.39)	39 (0.18-0.65)	0.61 (0.30-0.85)
				2013	0.19 (0.07-0.40)	0.40 (0.18-0.66)	0.36 (0.13-0.67)
Predators	Araneae	Araneidae	*Araneus* spp.*		0.17 (0.08-0.31)	0.16 (0.09-0.26)	0.00
		Anyphaenidae	*Hibana* spp.		0.15 (0.07-0.30)	0.12 (0.07-0.19)	0.07 (0.00-0.20)
		Linyphiidae	Unknown spp.		0.12 (0.06-0.24)	0.19 (0.13-0.27)	0.00
		Lycosidae	*Rabidosa rabida**		0.14 (0.06-0.30)	0.14 (0.08-0.25)	0.00
		Oxyopidae	*Oxyopes* spp.	2011	0.02 (0.01-0.06)	0.10 (0.05-0.17)	0.00
				2012	0.04 (0.02-0.09)	0.16 (0.07-0.31)	0.00
				2013	0.09 (0.04-0.18)	0.31 (0.16-0.51)	0.07 (0.00-0.20)
		Salticidae	*Phiddipus* spp.*		0.19 (0.09-0.33)	0.17 (0.10-0.27)	0.00
		Thomisidae	*Misumena* spp.*		0.04 (0.01-0.13)	0.10 (0.05-0.17)	0.00
	Coleoptera	Carabidae	*Dromochorus welderensis**		0.27 (0.16-0.42)	0.34 (0.23-0.47)	0.00

(Table 4.6 continued)

Functional Group	Order	Family	Species	Year	Seeding		OWB
					Without	**With**	
Predators	Hymenoptera	Bethylidae	*Pristocera hyalina*	2011	0.16 (0.07-0.34)	0.18 (0.10-0.29)	0.00
				2012	0.06 (0.3-0.11)	0.06 (0.03-0.12)	0.00
				2013	0.03 (0.01-0.06)	0.03 (0.01-0.07)	0.13 (0.00-0.31)
		Encyrtidae	Unknown spp.	2011	0.03 (0.01-0.10)	0.04 (0.02-0.08)	0.10 (0.00-0.30)
				2012	0.08 (0.02-0.25)	0.11 (0.04-0.33)	0.20 (0.00-0.41)
				2013	0.18 (0.06-0.42)	0.24 (0.08-0.52)	0.27 (0.04-0.50)
		Mymarmidae	Unknown spp.	2011	0.07 (0.03-0.19)	0.10 (0.05-0.16)	0.00
				2012	0.15 (0.06-0.34)	0.20 (0.08-0.41)	0.07 (0.00-0.20)
				2013	0.27 (0.12-0.50)	0.33 (0.16-0.57)	0.00
		Scelionidae	*Telenomus* spp.*		0.26 (0.14-0.41)	0.23 (0.14-0.34)	0.20 (0.00-0.41)
			Trissolcus spp.	2011	0.03 (0.01-0.09)	0.03 (0.01-0.05)	0.00
				2012	0.18 (0.07-0.40)	0.16 (0.06-0.37)	0.33 (0.09-0.58)
				2013	0.16 (0.06-0.37)	0.15 (0.05-0.35)	0.33 (0.09-0.58)
		Trichogrammatidae		2011	0.01 (0.00-0.06)	0.00 (0.00-0.01)	0.00
				2012	0.18 (0.03-0.63)	0.09 (0.01-0.43)	0.53 (0.27-0.79)
				2013	0.13 (0.02-0.52)	0.06 (0.01-0.32)	0.60 (0.34-0.86)

(Table 4.6 continued)

Functional Group	Order	Family	Species	Year	Seeding		OWB
					Without	With	
Predators	Opiliones	Cosmetidae	*Vonones* spp.	2011	0.10 (0.05-0.19)	0.15 (0.10-0.22)	0.16 (0.05-0.40)
				2012	0.67 (0.51-0.81)	0.77 (0.63-0.87)	0.03 (0.01-0.14)
				2013	0.43 (0.28-0.60)	0.55 (0.39-0.71)	0.20 (0.00-0.41)
	Scorpiones	Buthidae	*Centruroides vittatus*	2011	0.50 (0.35-0.64)	0.56 (0.46-0.66)	0.50 (0.17-0.83)
				2012	0.34 (0.24-0.47)	0.41 (0.29-0.54)	0.53 (0.27-0.79)
				2013	0.24 (0.16-0.35)	0.30 (0.20-0.42)	0.20 (0.00-0.41)
	Scutigeromorpha	Scutigeridae	*Scutiger coleoptera*	2011	0.28 (0.17-0.43)	0.24 (0.17-0.32)	0.20 (0.00-0.41)
	Thysanoptera	Aeolothripidae	*Aeolothrips* spp.*	2012	0.31 (0.18-0.47	0.33 (0.22-0.46)	0.20 (0.00-0.41)
				2013	0.85 (0.77-0.91)	0.86 (0.78-0.92)	0.87 (0.69-1.00)
		Phlaeothripidae	*Haplothrips* spp.	2011	0.00 (0.00-0.04)	0.01 (0.00-0.01)	0.15 (0.04-0.44)
				2012	0.35 (0.06-0.82)	0.38 (0.07-0.84)	0.68 (0.28-0.92)
				2013	0.57 (0.14-0.92)	0.61 (0.15-0.93)	0.82 (0.45-0.96)
		Thripidae	*Scolothrips* spp.	2011	0.01 (0.00-0.09)	0.01 (0.00-0.01)	0.07 (0.01-0.38)
				2012	0.53 (0.13-0.90)	0.37 (0.07-0.81)	0.74 (0.25-0.96)
				2013	0.22 (0.03-0.69)	0.12 (0.02-0.52)	0.14 (0.02-0.63)

(Table 4.6 continued)

Functional Group	Order	Family	Species	Year	Seeding		OWB
					Without	**With**	
Predators	Trombidiformes	Bdellidae		2011	0.17 (0.08-0.35)	0.18 (0.10-0.30)	0.00
				2012	0.10 (0.05-0.19)	0.10 (0.05-0.19)	0.07 (0.01-0.38)
				2013	0.04 (0.01-0.09)	0.04 (0.01-0.10)	0.07 (0.01-0.38)
		Erythraeidae		2011	0.38 (0.25-0.53)	0.40 (0.31-0.50)	0.05 (0.01-0.28)
				2012	0.46 (0.34-0.59)	0.49 (0.36-0.61)	0.65 (0.18-0.94)
				2013	0.32 (0.22-0.44)	0.34 (0.24-0.47)	0.25 (0.04-0.75)
		Smarididae*		2012	0.08 (0.03-0.18)	0.12 (0.07-0.21)	0.13 (0.00-0.31)
				2013	0.22 (0.12-0.37)	0.31 (0.18-0.48)	0.07 (0.01-0.38)
Ants	Hymenoptera	Formicidae	*Forelius mccooki*	2011	0.86 (0.61-0.96)	0.92 (0.81-0.97)	0.27 (0.04-0.50)
				2012	0.07 (0.03-0.15)	0.12 (0.06-0.25)	0.00
				2013	0.09 (0.04-0.19)	0.16 (0.08-0.31)	0.07 (0.01-0.38)
			Forelius pruinosus	2011	0.49 (0.32-0.66)	0.54 (0.41-0.66)	0.04 (0.00-0.26)
				2012	0.25 (0.17-0.37)	0.29 (0.20-0.42)	0.02 (0.00-0.17)
				2013	0.59 (0.46-0.71)	0.64 (0.51-0.75)	0.76 (0.24-0.97)

(Table 4.6 continued)

Functional Group	Order	Family	Species	Year	Seeding		OWB
					Without	**With**	
Ants	Hymenoptera	Formicidae	*Nylanderia terricola*	2011	0.29	0.28	0.00
					(0.17-0.45)	(0.19-0.39)	
				2012	0.23	0.22	0.20
					(0.14-0.35)	(0.14-0.34)	(0.00-0.41)
				2013	0.22	0.22	0.40
					(0.13-0.35)	(0.13-0.35)	(0.14-0.66)
			Solenopsis geminata		0.65	0.69	0.22
					(0.51-0.78)	(0.59-0.77)	(0.07-0.50)
			Solenopsis invicta	2011	0.72	0.65	0.86
					(0.56-0.83)	(0.55-0.75)	(0.61-0.96)
				2012	0.29	0.23	0.62
					(0.19-0.42)	(0.15-0.35)	(0.65-0.94)
				2013	0.31	0.25	0.75
					(0.20-0.44)	(0.16-0.37)	(0.41-0.93)

*2012-2013 Only

Table 4.7. Factors affecting abundance of arthropod species (*n* = 400 samples, 50 plots*8 sampling periods), southern Texas, summers 2011-2013.

Order/ Functional Group	Species/ Taxa	Soil Treatment		Seeding		Year		Soil*Seed		Soil*Year		Seed*Year	
		$F_{4,44}$	*P*	$F_{1,44}$	*P*	$F_{2,348}$	*P*	$F_{4,40}$	*P*	$F_{8,338}$	*P*	$F_{2,338}$	*P*
Herbivores													
Hemiptera	*Balclutha rubrostriata**	1.03	0.402	52.60	<0.001	739.42	<0.001			10.26	<0.001	44.80	<0.001
	*Xyonysius californicus**	0.78	0.544	4.15	0.048	33.70	<0.001						
	*Pseudatomoscelis seriatus**	1.44	0.237	3.85	0.056	111.64	<0.001			6.22	<0.001	41.88	<0.001
	Corythucha spp.*	0.84	0.507	9.12	0.004	13.46	<0.001					10.19	<0.001
Sarcoptiformes	Mochlozetidae	0.49	0.743	3.85	0.056	119.55	<0.001	2.96	0.031	4.62	<0.001	6.33	0.002
Decomposers													
Coleoptera	*Melanophthalma* spp.*	2.71	0.042	25.29	<0.001	17.71	<0.001			2.62	0.009	5.85	0.003
Oniscidea	*Armadillidium vulgare*	0.74	0.570	15.07	<0.001	649.83	<0.001			5.02	<0.001	87.34	<0.001
Orthoptera	*Gryllus* spp.	0.75	0.563	10.18	0.003	2.11	0.123						
Predators													
Araneae	*Araneus* spp.*	1.90	0.127	0.07	0.793	6.00	0.003			3.71	<0.001		
	*Rabida rabidosa**	0.18	0.948	0.16	0.691	6.45	0.002						
	Phiddipus spp.*	1.81	0.144	0.70	0.407	0.93	0.396					6.43	0.002
	Misumena spp.*	1.62	0.186	2.00	0.164	5.30	0.005			3.64	<0.001	8.29	<0.001
Coleoptera	*Dromochorus welderensis**	2.02	0.108	4.07	0.050	0.61	0.544					2.34	0.098
Hymenoptera	*Telenomus* spp.*	1.43	0.240	0.00	1.00	1.28	0.279			2.85	0.005	6.11	0.002
Opiliones	*Vonones* spp.	0.96	0.439	0.57	0.454	111.82	<0.001			2.50	0.012	21.50	<0.001
Scorpiones	*Centruroides vittatus*	1.43	0.240	2.84	0.099	27.95	<0.001			2.15	0.033		

(Table 4.7 continued)

Order/ Functional Group	Species/ Taxa	Soil Treatment		Seeding		Year		Soil*Seed		Soil*Year		Seed*Year	
		$F_{4,44}$	*P*	$F_{1,44}$	*P*	$F_{2,348}$	*P*	$F_{4,40}$	*P*	$F_{8,338}$	*P*	$F_{2,338}$	*P*
Predators													
Scutigeromorpha	*Scutiger coleoptera*	0.72	0.583	5.92	0.019	2.89	0.057			2.53	0.011		
Trombidiformes	Anystidae	0.83	0.513	22.02	<0.001	89.82	<0.001			12.88	<0.001		
	Erythraeidae	3.30	0.019	0.82	0.370	1.73	0.179	3.58	0.014	6.44	<0.001	4.13	0.017
Thysanoptera	*Aeolothrips* spp.*	0.48	0.750	25.17	<0.001	524.81	<0.001					21.93	<0.001
Ants													
Hymenoptera	*Forelius pruinosus*	0.93	0.455	0.16	0.691	34.99	<0.001			6.50	<0.001	11.81	<0.001
	Solenopsis geminata	1.68	0.172	0.07	0.793	2.73	0.067			31.77	<0.001	74.77	<0.001
	Solenopsis invicta	0.29	0.883	0.49	0.488	81.41	<0.001			13.87	<0.001	9.57	<0.001

*2012-2013 Only

Table 4.8. Abundance of arthropod species (individuals/m^2, median values and back-transformed 95% CIs), southern Texas summers 2011-2013. We provide means and 95% CIs for abundance of arthropod species and multiple estimates when we detect a difference among treatments or years.

Order/ Functional Group	Species/Taxon	Seeding	Year	Soil and Seed Treatments					
				Disturb	**Carbon**	**Fungi**	**Lime**	**Sulfur**	**OWB**
Herbivores									
Hemiptera	*Balclutha rubrostriata**	Without	2012	3.1 (1.1-8.5)	1.5 (0.7-3.4)	2.7 (1.0-7.4)	2.9 (1.1-8.2)	3.1 (1.1-8.7)	12.9 (2.9-22.8)
		With		0.8 (0.4-1.5)	0.4 (0.2-0.7)	0.7 (0.4-1.3)	0.8 (0.4-1.3)	0.8 (0.4-1.5)	
		Without	2013	20.5 (14.1-29.7)	20.6 (14.8-28.6)	16.0 (10.6-24.1)	8.1 (5.4-12.1)	20.2 (13.6-29.9)	109.1 (21.9-196.4
		With		2.1 (1.6-2.7)	2.1 (1.6-2.8)	1.6 (1.2-2.1)	0.8 (0.6-1.1)	2.1 (1.6-2.7)	
	*Xyonysius californicus**	Without	2012	0.4 (0.1-1.2)					0.0
		With		1.1 (0.4-2.6)					
		Without	2013	0.0					0.3 (0.0-0.7)
		With		0.0					
	*Pseudatomoscelis seriatus**	Without	2012	1.2 (0.4-3.8)	0.3 (0.1-0.7)	1.1 (0.4-3.4)	0.9 (0.3-2.7)	1.0 (0.3-3.2)	0.0
		With		0.7 (0.4-1.4)	0.2 (0.1-0.3)	0.7 (0.3-1.3)	0.5 (0.3-1.0)	0.6 (0.3-3.2)	
		Without	2013	2.0 (1.0-4.1)	1.0 (0.6-1.9)	1.3 (0.6-2.7)	1.6 (0.8-3.3)	0.4 (0.2-1.0)	0.2 (0.0-0.5)
		With		4.5 (3.0-6.7)	2.3 (1.5-3.4)	2.9 (1.9-4.3)	3.6 (2.4-5.4)	1.0 (0.7-1.5)	

(Table 4.8 continued)

Order/ Functional Group	Species/Taxon	Seeding	Year	Soil and Seed Treatments					
				Disturb	**Carbon**	**Fungi**	**Lime**	**Sulfur**	**OWB**
Herbivores									
Hemiptera	*Corythucha* spp.*	Without	2012	0.1 (0.0-0.4)					0.0
		With		0.2 (0.1-0.6)					
		Without	2013	0.0 (0.0-0.1)					0.0
		With		0.6 (0.1-5.9)					
Sarcoptiformes	Mochlozetidae[a]	Without	2011	0.2 (0.0-1.1)	0.0 (0.0-0.2)	0.0	0.0 (0.0-0.1)	0.0 (0.0-0.1)	1.9 (1.1-3.3)
		With		0.0 (0.0-0.1)	0.2 (0.0-1.2)	0.0	0.1 (0.0-0.4)	0.0 (0.0-0.2)	
		Without	2012	0.1 (0.0-0.5)	0.0 (0.0-0.1)	0.0	0.1 (0.0-0.7)	0.1 (0.0-1.1)	41.1 (27.5-61.2)
		With		0.0 (0.0-0.1)	0.0 (0.0-0.2)	0.0	0.1 (0.0-0.4)	0.1 (0.0-0.4)	
		Without	2013	0.1 (0.0-0.4)	0.0 (0.0-0.1)	0.0	0.0 (0.0-0.3)	0.0 (0.0-0.4)	84.3 (56.7-125.3)
		With		0.0 (0.0-0.1)	0.0 (0.0-0.2)	0.0	0.1 (0.0-0.3)	0.0 (0.0-0.2)	
Decomposers									
Coleoptera	*Melanophthalma spp.**	Without	2012	0.3 (0.1-0.7)	0.1 (0.1-0.3)	0.3 (0.1-0.8)	0.7 (0.3-1.5)	0.3 (0.1-0.7)	0.7 (0.0-1.3)
		With		1.1 (0.7-1.9)	0.5 (0.3-0.9)	1.3 (0.8-2.2)	2.6 (1.6-4.5)	1.2 (0.7-2.0)	
		Without	2013	0.2 (0.1-0.7)	0.3 (0.1-0.8)	0.4 (0.2-1.0)	0.4 (0.4-1.5)	0.2 (0.1-0.5)	0.7 (0.3-1.0)
		With		0.4 (0.2-0.8)	0.6 (0.3-1.2)	0.7 (0.4-1.4)	0.8 (0.4-1.5)	0.3 (0.2-0.6)	

(Table 4.8 continued)

Order/ Functional Group	Species/Taxon	Seeding	Year	Soil and Seed Treatments					
				Disturb	**Carbon**	**Fungi**	**Lime**	**Sulfur**	**OWB**
Decomposers									
Oniscidea	*Armadillidium vulgare*	Without	2011	2.3 (1.4-3.8)	3.2 (2.2-4.7)	2.9 (1.8-4.6)	4.0 (2.5-6.4)	3.8 (2.3-6.1)	4.2 (2.9-6.2)
		With		4.2 (3.1-5.7)	5.7 (4.2-7.8)	5.2 (3.8-7.0)	7.2 (5.3-9.8)	6.8 (5.0-9.3)	
		Without	2012	12.8 (9.3-17.6)	12.6 (9.9-16.1)	11.7 (8.6-16.0)	13.7 (10.2-18.4)	16.8 (13.5-20.8)	3.2 (2.2-4.9)
		With		23.8 (19.6-29.0)	23.5 (19.3-28.7)	21.9 (18.0-26.7)	25.6 (21.0-31.2)	31.3 (25.7-38.1)	
		Without	2013	7.9 (5.6-11.2)	10.1 (7.8-13.1)	6.9 (4.9-9.7)	10.0 (7.3-13.7)	8.0 (5.8-11.1)	0.5 (0.2-1.1)
		With		6.0 (4.9-9.7)	7.7 (6.2-9.6)	5.3 (4.3-6.6)	7.7 (6.2-9.5)	6.1 (4.9-7.6)	
Orthoptera	*Gryllus* spp.	Without	2011	0.6 (0.4-0.9)					1.2 (0.6-2.4)
		With		0.4 (0.3-0.5)					
		Without	2012	0.8 (0.6-1.1)					0.6 (0.3-1.5)
		With		0.5 (0.3-0.7)					
		Without	2013	0.8 (0.6-1.1)					0.7 (0.3-1.6)
		With		0.5 (0.4-0.7)					

(Table 4.8 continued)

Order/ Functional Group	Species/Taxon	Seeding	Year	Soil and Seed Treatments					
				Disturb	**Carbon**	**Fungi**	**Lime**	**Sulfur**	**OWB**
Predators									
Araneae	*Araneus* spp.*	Without	2012	0.1 (0.0-0.5)	0.1 (0.1-0.4)	0.0 (0.0-0.3)	0.3 (0.1-0.9)	0.3 (0.1-1.0)	0.0
		With		0.2 (0.1-0.3)	0.2 (0.1-0.3)	0.0 (0.0-0.1)	0.3 (0.2-0.5)	0.3 (0.2-0.5)	
		Without	2013	0.5 (0.1-2.1)	0.6 (0.2-1.7)	0.2 (0.0-2.5)	0.1 (0.0-0.5)	0.3 (0.1-1.0)	0.0
		With		0.6 (0.1-2.3)	0.7 (0.2-1.8)	0.2 (0.0-2.6)	0.1 (0.0-0.5)	0.3 (0.1-1.1)	
	*Rabida rabidosa**	Without	2012	0.2 (0.1-0.6)					0.0
		With		0.2 (0.1-0.4)					
		Without	2013	0.1 (0.1-0.2)					0.0
		With		0.1 (0.1-0.2)					
	Phiddipus spp.*	Without	2012	0.6 (0.3-1.0)					0.0
		With		0.3 (0.2-0.6)					
		Without	2013	0.3 (0.1-0.5)					0.0
		With		0.4 (0.2-1.1)					
	Misumena spp.*	Without		0.0 (0.0-0.1)					0.0
		With		0.1 (0.1-0.2)					

(Table 4.8 continued)

Order/ Functional Group	Species/Taxon	Seeding	Year	Soil and Seed Treatments					
				Disturb	**Carbon**	**Fungi**	**Lime**	**Sulfur**	**OWB**
Predators									
Coleoptera	*Dromochorus welderensis**	Without	2012	0.4 (0.2-0.6)					0.0
		With		0.4 (0.2-0.7)					
		Without	2013	0.3 (0.2-0.5)					0.0
		With		0.6 (0.3-1.2)					
Hymenoptera	*Telenomus* spp.*	Without	2012	0.1 (0.0-0.5)	0.1 (0.0-0.3)	0.1 (0.0-0.4)	0.1 (0.0-0.4)	0.4 (0.1-1.2)	0.3 (0.0-0.6)
		With		0.3 (0.1-0.6)	0.2 (0.1-0.4)	0.2 (0.1-0.4)	0.2 (0.1-0.5)	0.7 (0.3-1.6)	
		Without	2013	0.3 (0.1-1.2)	0.6 (0.2-1.9)	0.2 (0.0-1.0)	0.4 (0.1-1.6)	0.3 (0.1-1.0)	0.5 (0.0-1.3)
		With		0.2 (0.1-0.5)	0.4 (0.2-1.0)	0.1 (0.1-0.3)	0.3 (0.1-0.6)	0.2 (0.1-0.4)	
Opiliones	*Vonones* spp.	Without	2011	0.1 (0.0-0.7)	0.0 (0.0-0.2)	0.0 (0.0-0.3)	0.0 (0.0-0.3)	0.1 (0.0-0.6)	1.0 (0.1-1.9)
		With		0.4 (0.1-1.3)	0.1 (0.0-0.4)	0.1 (0.0-0.4)	0.1 (0.0-0.4)	0.3 (0.1-1.1)	
		Without	2012	5.0 (1.0-25.0)	4.3 (0.8-22.2)	4.3 (0.6-31.2)	3.6 (0.5-25.8)	3.6 (0.7-18.9)	1.2 (0.4-2.0)
		With		5.0 (1.6-15.4)	4.3 (1.4-13.3)	4.3 (1.4-13.4)	3.6 (1.1-11.1)	3.6 (1.1-11.1)	
		Without	2013	1.6 (0.3-8.4)	0.9 (0.2-5.0)	1.1 (0.2-8.2)	1.4 (0.2-10.3)	0.6 (0.1-3.3)	0.7 (0.1-1.3)
		With		4.2 (1.3-13.1)	2.4 (0.8-7.6)	2.8 (0.9-8.9)	3.5 (1.1-11.2)	1.5 (0.5-4.8)	

(Table 4.8 continued)

Order/ Functional Group	Species/Taxon	Seeding	Year	Soil and Seed Treatments					
				Disturb	**Carbon**	**Fungi**	**Lime**	**Sulfur**	**OWB**
Predators									
Scorpiones	*Centruroides vittatus*	Without	2011	1.2 (0.7-2.1)	1.4 (0.9-2.1)	1.5 (0.9-2.5)	0.7 (0.4-1.3)	1.7 (1.0-2.9)	0.7 (0.2-1.2)
		With		1.5 (1.2-1.9)	1.8 (1.4-2.3)	1.8 (1.4-2.3)	0.9 (0.7-1.1)	2.2 (1.7-2.8)	
		Without	2012	1.1 (0.5-2.3)	0.4 (0.2-0.8)	0.6 (0.3-1.4)	0.8 (0.3-1.8)	0.6 (0.3-1.3)	0.9 (0.3-1.4)
		With		1.3 (0.6-2.9)	0.5 (0.3-0.6)	0.8 (0.3-1.8)	0.9 (0.4-2.3)	0.7 (0.3-1.7)	
		Without	2013	0.7 (0.3-1.6)	0.4 (0.2-0.8)	0.5 (0.2-1.1)	0.5 (0.2-1.2)	0.5 (0.2-1.2)	0.2 (0.0-1.4)
		With		0.9 (0.4-2.0)	0.5 (0.3-0.9)	0.6 (0.2-1.4)	0.6 (0.2-1.4)	0.6 (0.3-1.5)	
Scutigeromorpha	*Scutiger coleoptera*	Without	2011	0.3 (0.1-0.9)	0.2 (0.1-0.5)	0.5 (0.2-1.4)	0.3 (0.1-1.0)	0.3 (0.1-0.9)	0.0
		With		0.4 (0.3-0.6)	0.3 (0.2-0.4)	0.7 (0.5-1.0)	0.5 (0.3-0.7)	0.5 (0.3-0.6)	
		Without	2012	0.4 (0.1-1.6)	0.8 (0.3-2.0)	0.2 (0.1-0.9)	0.4 (0.1-1.5)	0.5 (0.1-1.7)	0.0
		With		0.7 (0.2-2.4)	1.2 (0.5-3.1)	0.4 (0.1-1.3)	0.6 (0.2-2.2)	0.8 (0.2-2.7)	
		Without	2013	0.4 (0.1-1.7)	0.2 (0.1-0.5)	0.2 (0.1-1.0)	0.5 (0.1-1.9)	0.3 (0.2-1.5)	0.2 (0.0-0.4)
		With		0.6 (0.2-2.7)	0.3 (0.1-0.8)	0.4 (0.1-1.6)	0.7 (0.2-2.9)	0.5 (0.1-2.3)	

(Table 4.8 continued)

Order/ Functional Group	Species/Taxon	Seeding	Year	Soil and Seed Treatments					
				Disturb	**Carbon**	**Fungi**	**Lime**	**Sulfur**	**OWB**
Predators									
Thysanoptera	*Aeolothrips* spp.*	Without	2012	0.4 (0.2-0.7)					0.4 (0.0-0.9)
		With		0.4 (0.2-0.7)					
		Without	2013	12.3 (8.6-17.6)					17.3 (6.2-28.3)
		With		3.7 (2.2-6.0)					
Trombidiformes	Anystidae	Without	2011	8.4 (6.3-11.4)	10.8 (8.6-13.5)	6.0 (4.4-8.1)	6.1 (4.5-8.3)	7.9 (5.9-10.7)	5.9 (4.4-8.1)
		With		9.2 (7.8-10.9)	11.8 (9.9-13.9)	6.5 (5.5-7.7)	6.7 (5.6-7.9)	8.7 (7.3-10.2)	
		Without	2012	5.0 (3.8-6.5)	4.6 (3.8-5.5)	5.2 (3.9-6.9)	7.1 (5.4-9.3)	5.0 (3.8-6.5)	9.3 (7.0-12.5)
		With		5.4 (4.1-7.1)	5.0 (4.1-6.0)	5.7 (4.3-7.5)	7.7 (5.8-10.1)	5.4 (4.1-7.1)	
		Without	2013	5.2 (3.9-6.8)	3.8 (3.0-6.3)	5.0 (3.8-6.8)	4.7 (3.5-6.4)	2.5 (1.9-5.0)	9.2 (7.0-12.4)
		With		5.6 (4.3-7.4)	4.1 (3.4-5.0)	5.5 (4.1-7.4)	5.2 (3.9-6.9)	2.8 (2.0-3.8)	

(Table 4.8 continued)

Order/ Functional Group	Species/Taxon	Seeding	Year	Soil and Seed Treatments					
				Disturb	**Carbon**	**Fungi**	**Lime**	**Sulfur**	**OWB**
Predators									
	Erythraeidae	Without	2011	0.9 (0.4-2.1)	0.7 (0.3-1.3)	0.7 (0.3-1.7)	0.6 (0.2-1.4)	0.9 (0.4-2.2)	0.0
		With		0.8 (0.3-2.2)	1.2 (0.5-2.6)	0.6 (0.2-1.5)	0.7 (0.3-1.7)	3.1 (1.3-7.3)	
		Without	2012	0.9 (0.4-2.0)	1.0 (0.5-1.8)	1.1 (0.5-2.6)	2.1 (0.9-4.7)	0.6 (0.3-1.1)	2.9 (1.2-4.6)
		With		0.8 (0.5-1.3)	1.7 (1.0-2.7)	0.9 (0.6-1.5)	2.3 (1.4-3.9)	1.8 (1.1-2.9)	
		Without	2013	1.7 (0.6-4.6)	0.2 (0.1-0.6)	1.1 (0.4-3.3)	2.2 (0.8-6.1)	0.6 (0.3-1.3)	0.3 (0.0-0.6)
		With		0.9 (0.5-1.5)	0.2 (0.1-0.4)	0.5 (0.3-0.9)	1.4 (0.8-2.4)	1.1 (0.6-1.9)	
Ants Hymenoptera	*Forelius pruinosus*	Without	2011	1.5 (0.7-3.1)	2.4 (1.3-4.2)	1.2 (0.6-2.7)	0.9 (0.4-2.3)	0.9 (0.4-2.4)	0.0
		With		2.4 (1.4-3.9)	3.8 (2.3-6.2)	2.0 (1.2-3.3)	1.4 (0.9-2.4)	1.5 (0.9-2.4)	
		Without	2012	0.2 (0.1-0.7)	0.1 (0.1-0.3)	0.2 (0.1-0.7)	0.9 (0.5-1.6)	1.1 (0.6-1.8)	0.1 (0.0-0.3)
		With		0.4 (0.2-0.7)	0.2 (0.1-0.3)	0.4 (0.2-0.6)	0.9 (0.5-1.6)	1.1 (0.6-1.8)	
		Without	2013	1.3 (0.7-2.3)	1.6 (1.1-2.4)	1.0 (0.7-1.4)	1.9 (1.0-3.3)	1.3 (0.7-2.4)	3.1 (1.5-4.6)
		With		0.9 (0.6-1.3)	1.1 (0.7-1.6)	1.0 (0.7-1.4)	1.2 (0.8-1.8)	0.9 (0.6-1.3)	

(Table 4.8 continued)

Order/ Functional Group	Species/Taxon	Seeding	Year	Soil and Seed Treatments					
				Disturb	**Carbon**	**Fungi**	**Lime**	**Sulfur**	**OWB**
Ants Hymenoptera	*Solenopsis geminata*	Without	2011	4.6 (2.2-9.9)	4.0 (2.2-7.2)	3.0 (1.4-6.5)	7.2 (3.4-15.2)	1.8 (0.8-3.9)	0.4 (0.1-1.7)
		With		2.6 (1.6-4.2)	2.3 (1.4-3.7)	1.7 (1.1-2.8)	4.1 (2.5-6.6)	1.0 (0.6-1.6)	
		Without	2012	2.9 (1.9-4.3)	2.8 (2.1-3.8)	3.0 (2.0-4.6)	2.4 (1.6-3.5)	8.6 (5.6-13.2)	1.1 (0.4-3.3)
		With		2.5 (1.9-3.2)	2.5 (1.9-3.2)	2.6 (2.0-3.4)	2.1 (1.6-2.7)	7.5 (5.8-9.7)	
		Without	2013	2.0 (1.3-3.3)	0.5 (0.3-0.8)	1.8 (1.1-3.0)	2.4 (1.6-3.8)	1.8 (1.1-3.1)	1.7 (0.6-5.0)
		With		5.4 (4.1-7.0)	1.4 (1.1-1.8)	4.9 (3.7-6.4)	6.5 (5.0-8.5)	4.9 (3.7-6.4)	
	Solenopsis invicta	Without	2011	3.6 (1.6-7.8)	4.3 (2.4-7.9)	3.0 (1.4-6.5)	3.2 (1.5-6.9)	3.8 (1.7-8.2)	3.7 (2.1-6.4)
		With		3.1 (1.9-5.0)	3.7 (2.2-6.1)	2.6 (1.6-4.2)	2.7 (1.6-4.4)	3.2 (2.0-5.3)	
		Without	2012	1.5 (0.9-2.5)	1.2 (0.8-1.7)	2.2 (1.3-3.5)	2.9 (1.8-4.5)	1.1 (0.6-1.9)	1.0 (0.6-1.7)
		With		0.8 (0.6-1.3)	0.7 (0.5-0.9)	1.2 (0.9-1.6)	1.6 (1.1-2.2)	0.6 (0.4-0.8)	
		Without	2013	0.8 (0.4-1.6)	0.5 (0.3-0.8)	1.6 (0.9-2.9)	2.7 (1.6-4.7)	0.3 (0.2-0.7)	4.5 (3.1-6.4)
		With		1.0 (0.7-1.4)	0.6 (0.4-0.8)	1.9 (1.4-2.7)	3.2 (2.3-4.4)	0.4 (0.3-0.5)	

[a]We could not analyze abundance of Mochlozetid mites in plots treated with mycorrhizal fungi due to issues with convergence.

Figures

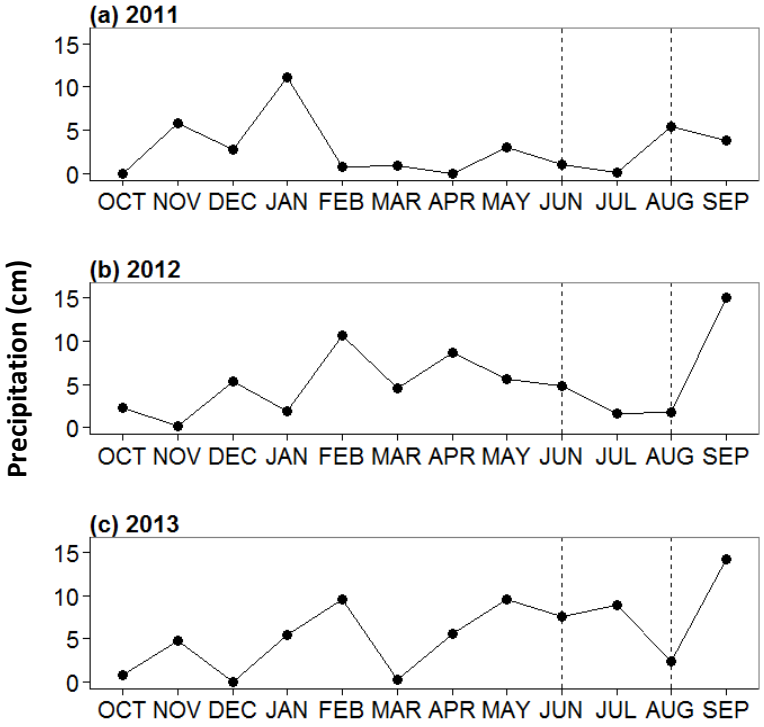

Figure 4.1. Total monthly precipitation for the Welder Wildlife Refuge, starting at the beginning of the water year (Oct 1), southern Texas, 2011-2013. The dashed lines represent precipitation observed during the months of sampling.

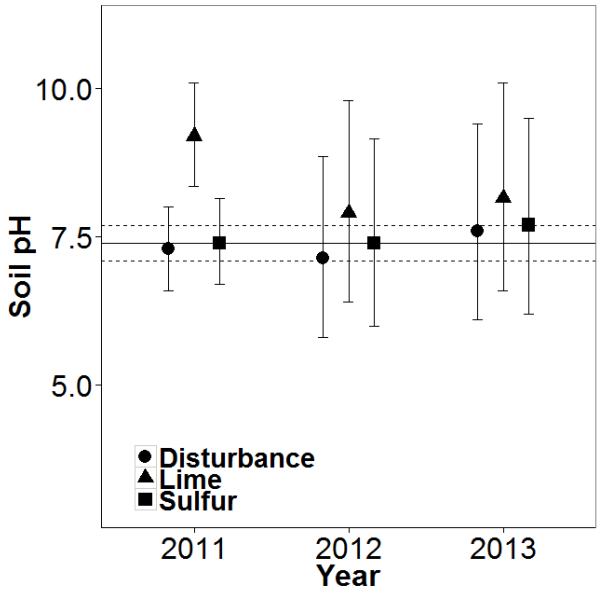

Figure 4.2. Soil pH (means and 95% CIs) for plots treated with lime, sulfur, or soil disturbance alone, southern Texas, summers 2011-2013. We include the mean (solid) and 95% CI (dashed) for the Kleberg (OWB) plots for comparison.

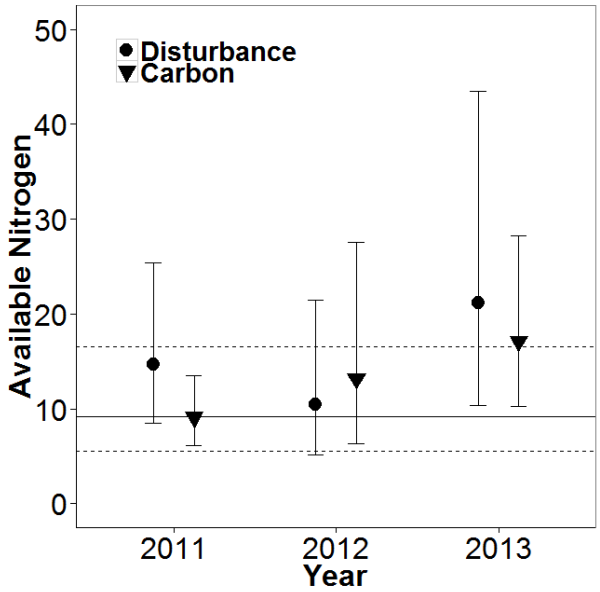

Figure 4.3. Available NO_3 (kg/ha, means and 95% CIs) for plots treated with carbon or soil disturbance alone, southern Texas, summers 2011-2013. We include the mean (solid) and 95% CI (dashed) for the Kleberg (OWB) plots for comparison.

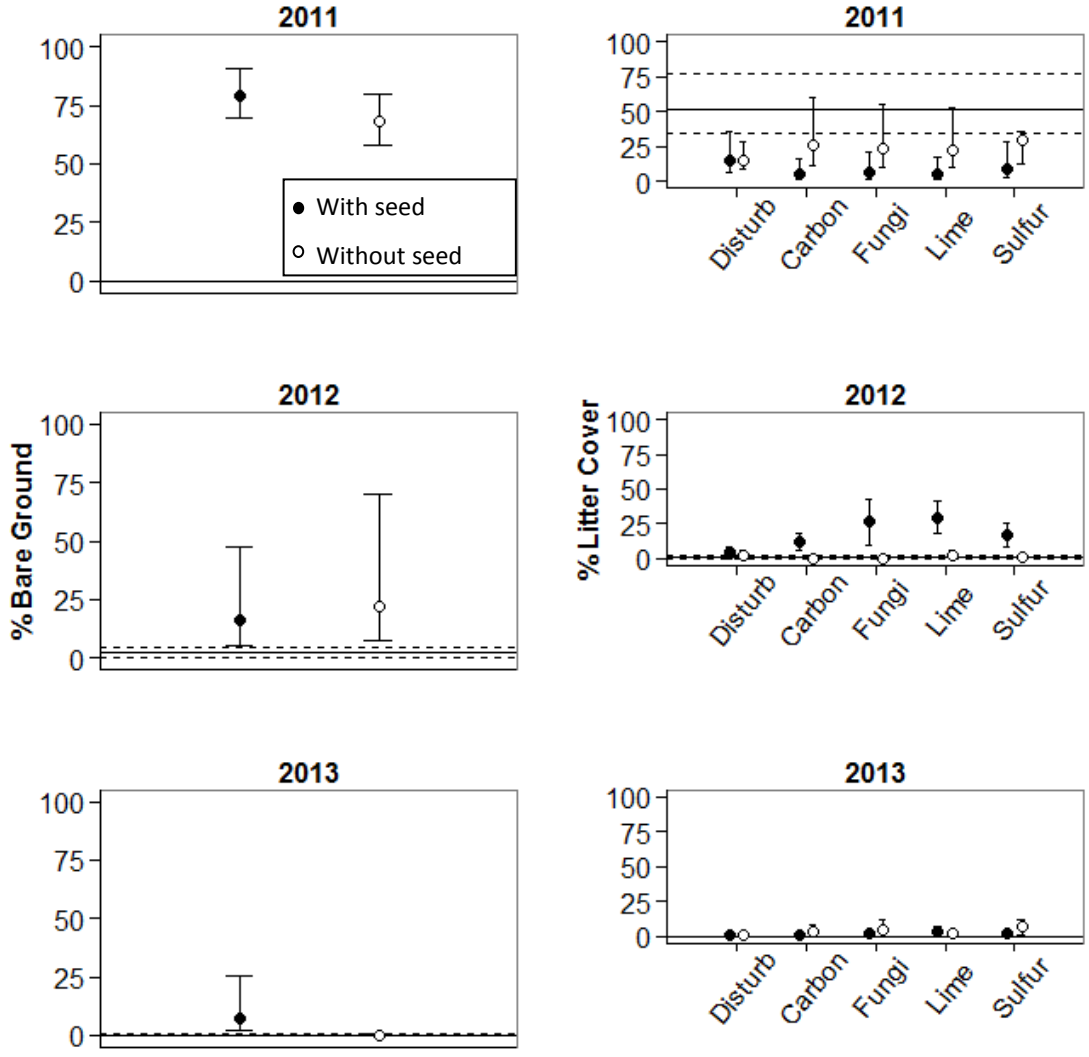

Figure 4.4 Bare ground and litter cover (means and 95% CIs) for plots with and without added seed, southern Texas, summers 2011-2013. We include the mean (solid) and 95% CI (dashed) for the Kleberg (OWB) plots for comparison.

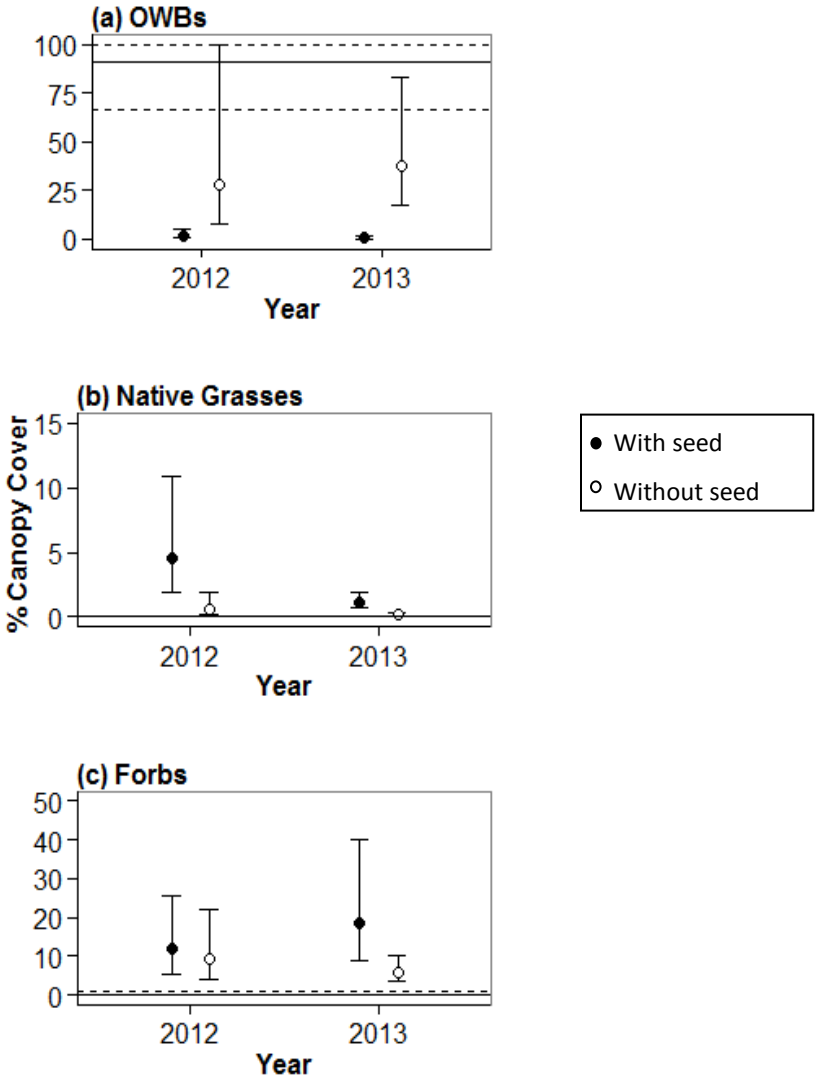

Figure 4.5. Canopy cover by cover class (means and 95% CIs) for plots with and without native seed added, southern Texas, summers 2012-2013. We include the mean (solid) and 95% CI (dashed) for Kleberg (OWB) plots for comparison.

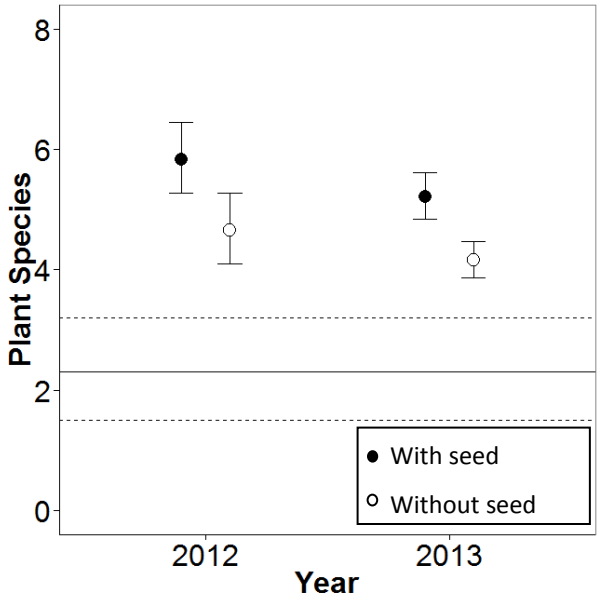

Figure 4.6. Plant species richness (species/m^2, means and 95% CIs) for plots with and without added seed, southern Texas, 2011-2013. We include the mean (solid) and 95% CI (dashed) for the Kleberg (OWB) plots for comparison.

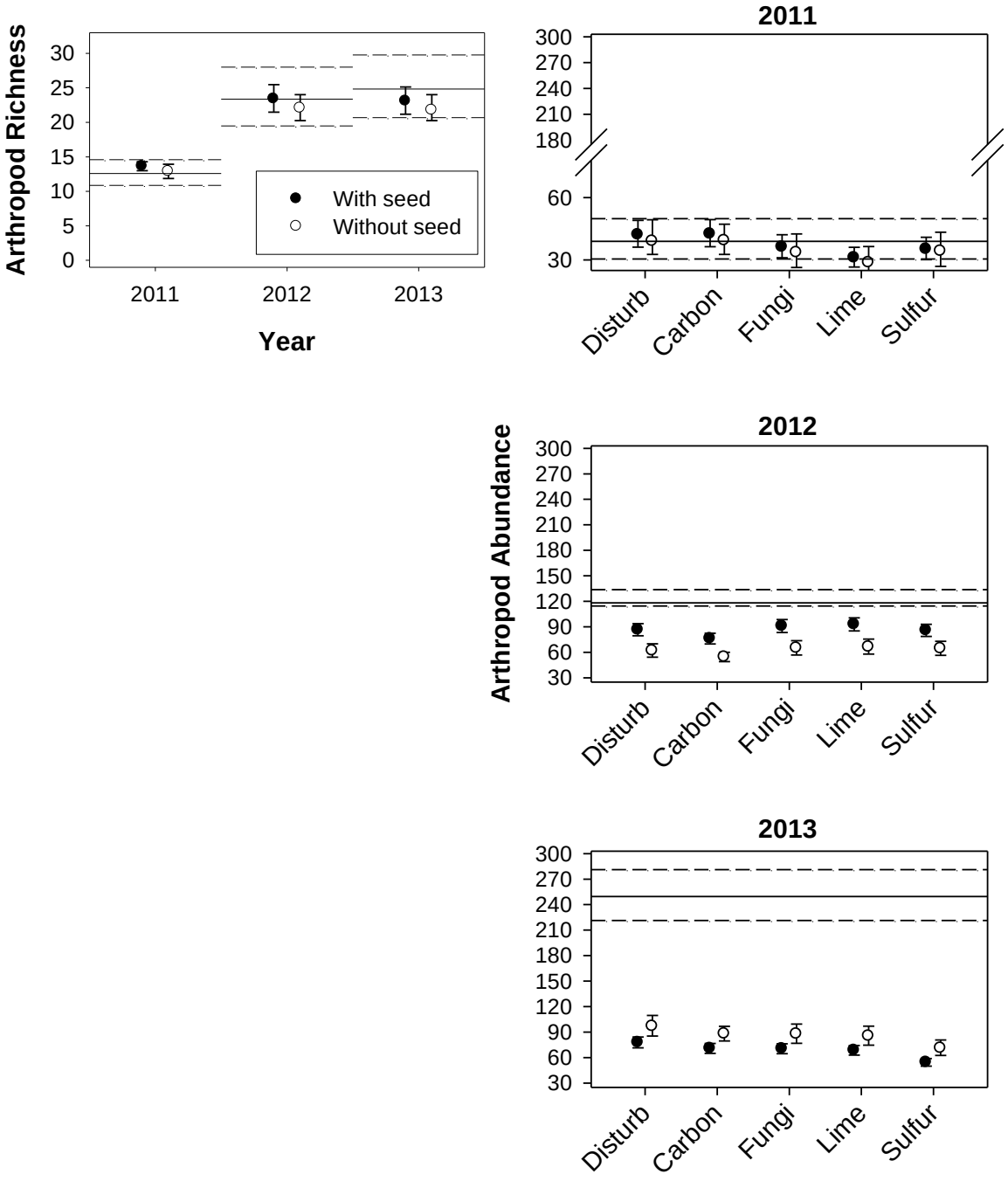

Figure 4.7. Total species richness (species/m^2) and abundance (arthropods/m^2) of arthropods (means and 95% CIs) in plots with and without added seed, southern Texas, summers 2011-2013. We include the mean (solid) and 95% CI (dashed) for the Kleberg (OWB) plots for comparison.

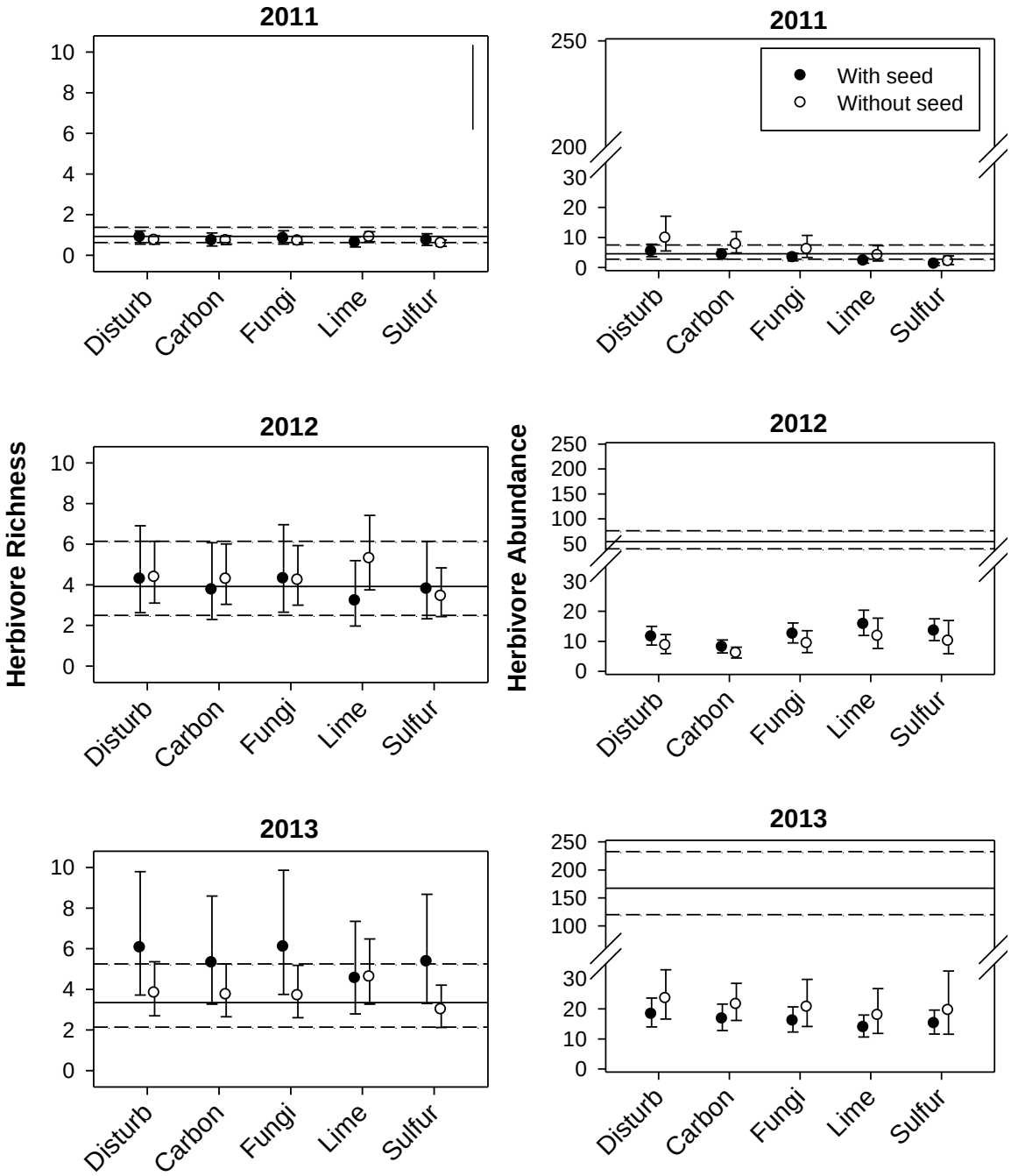

Figure 4.8. Richness (species/m^2) and abundance (arthropods/m^2) of herbivorous arthropods (means and 95% CIs) in plots with and without added seed, southern Texas, summers 2011-2013. We include the mean (solid) and 95% CI (dashed) for the Kleberg (OWB) plots for comparison.

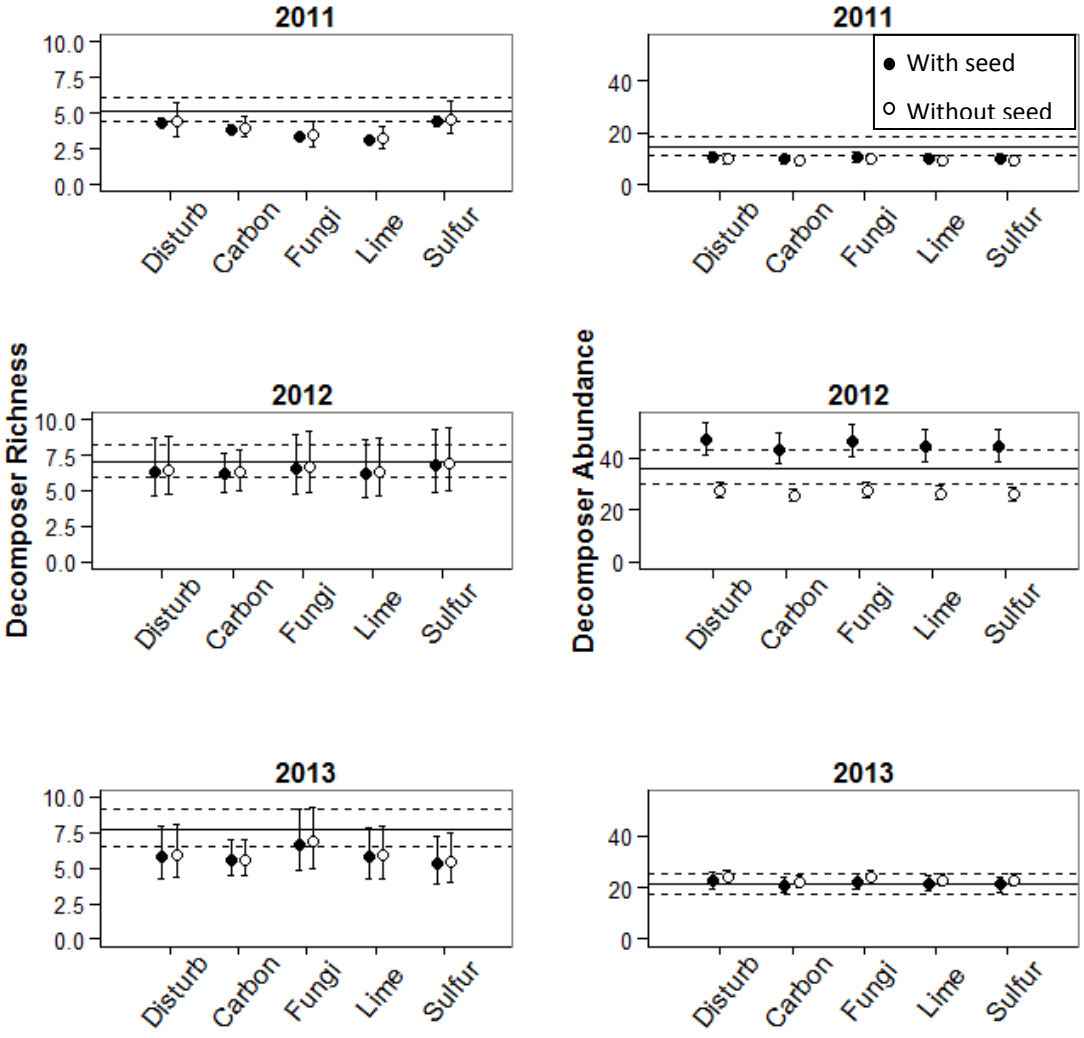

Figure 4.9. Richness (species/m^2) and abundance (arthropods/m^2) of decomposer arthropods (means and 95% CIs) in plots with and without added seed, southern Texas, summers 2011-2013. We include the mean (solid) and 95% CI (dashed) for the Kleberg (OWB) plots for comparison.

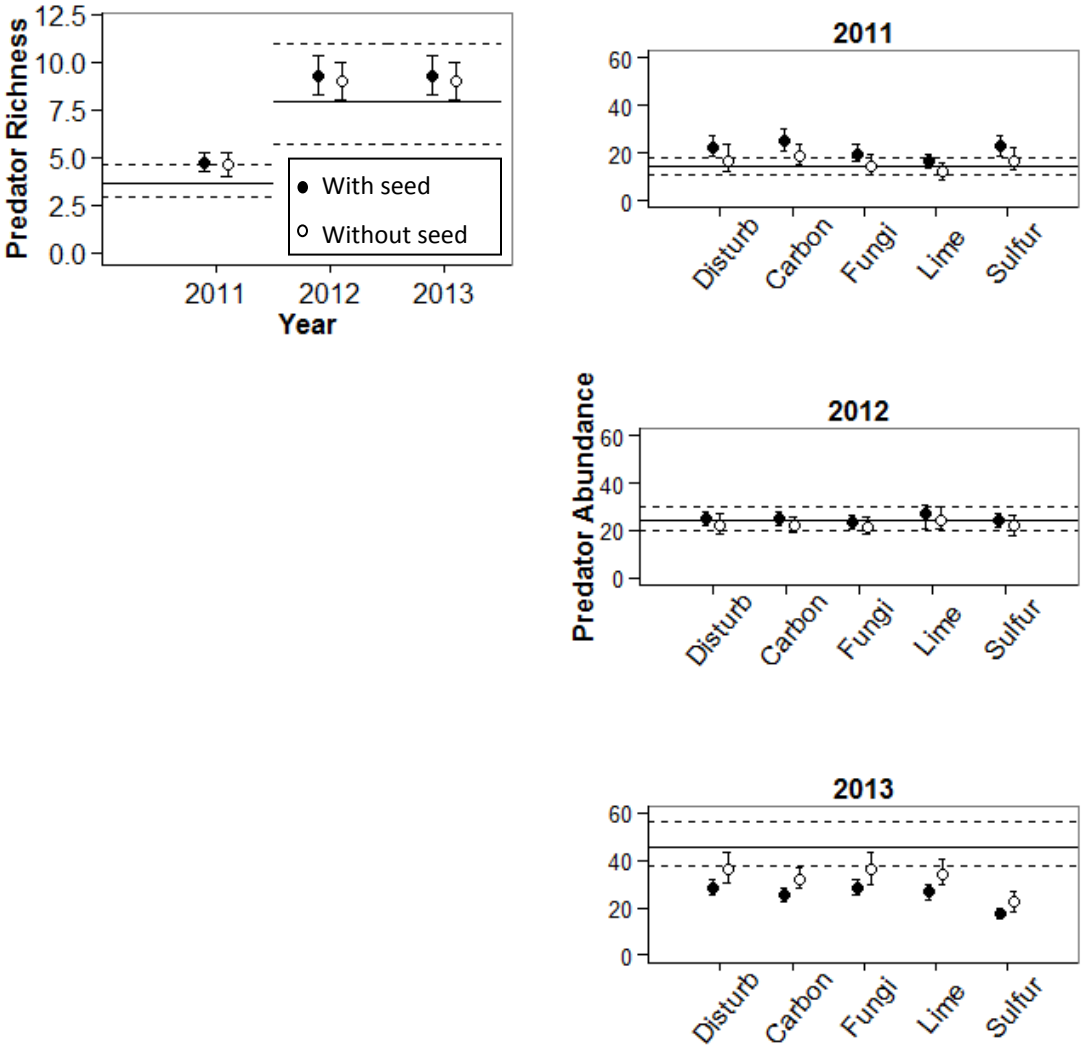

Figure 4.10. Richness (species/m^2) and abundance (arthropods/m^2) of predator arthropods (means and 95% CIs) in plots with and without added seed, southern Texas, summers 2011-2013. We include the mean (solid) and 95% CI (dashed) for the Kleberg (OWB) plots for comparison.

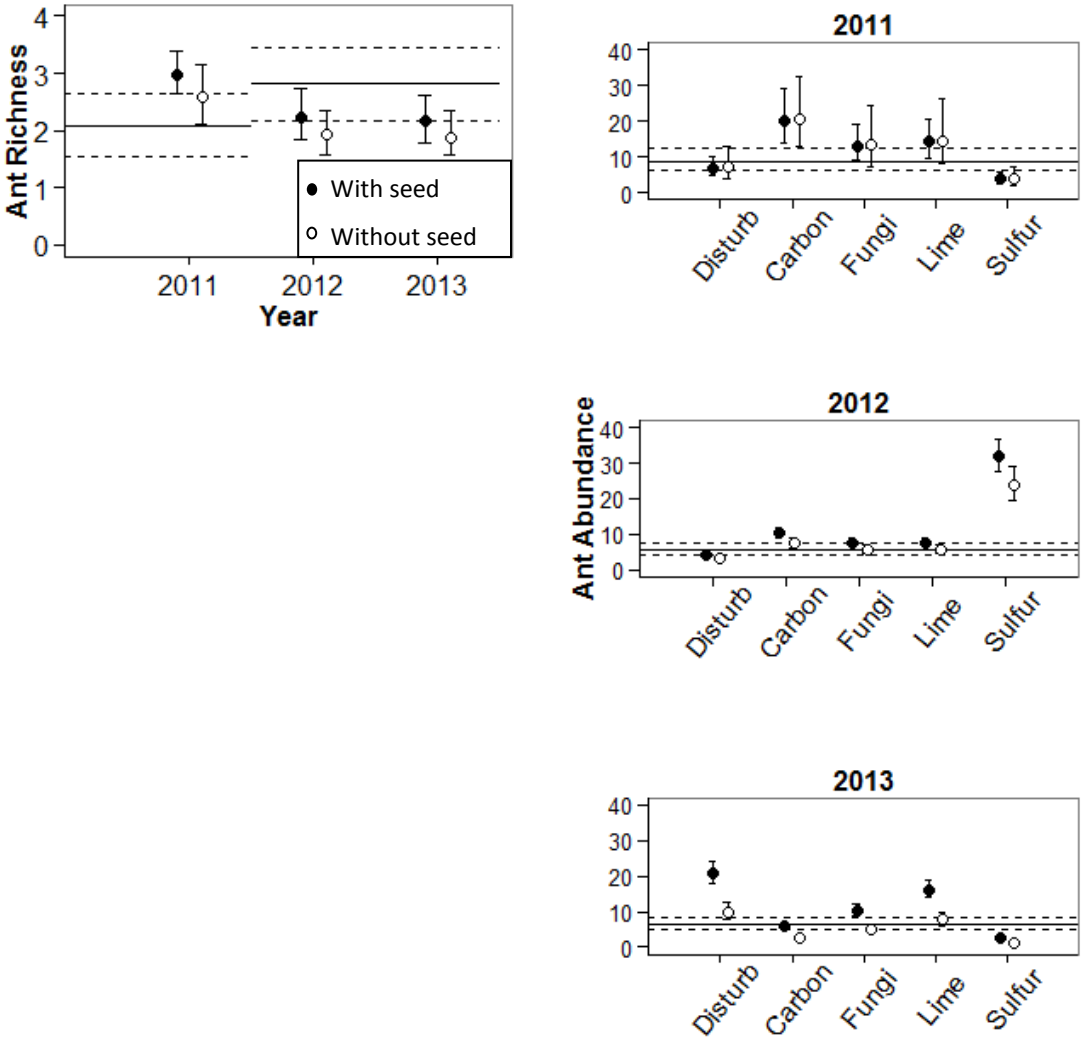

Figure 4.11. Richness (species/m^2) and abundance (arthropods/m^2) of ants (means and 95% CIs) in plots with and without added seed, southern Texas, summers 2011-2013. We include the mean (solid) and 95% CI (dashed) for the Kleberg (OWB) plots for comparison.

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CHAPTER FIVE

CONCLUSION

Plant invasion and drought can interact to produce novel effects on plant and arthropod communities. We observed fewer species of plants, as well as fewer species and lower abundance of arthropods in monocultures of OWBs during extreme drought, relative to native plant communities, but the direction and magnitude of these differences in plant community characteristics changed as drought subsided (Chapter 2). In contrast, richness and abundance of arthropods were comparable between treated plots and OWB plots during extreme drought, despite a lack of vegetation in treated plots; treated plots had more species and cover of native plants than OWBs, which resulted in different arthropod communities (Chapters 3 and 4).

Generally, native herbivorous arthropods cannot recognize or use invasive plants as food and decrease abundance where novel plants are dominant (Brown et al. 2002; Burghardt et al. 2010; Grabas and Laverty 1999). We observed a shift in the arthropod community from one driven by detritivores in native grasslands to one dominated by herbivores in areas dominated by OWBs (Chapter 2), contrary to studies reviewed in Gratton and Denno (2006), van Hengstum et al. (2014), and Litt et al. (in press). The increase in herbivore abundance in our study was driven by an invasive leafhopper (*Balclutha rubrostriata*) that may share its native range with OWBs, which may explain the association of the leafhopper with the invasive plant (Morgan et al. 2013; Zahniser et al. 2010). This pattern is consistent with the “invasional meltdown” hypothesis

(Simberloff and Von Holle 1999), where an invasive species facilitates the introduction of another. Although invasional meltdowns between invasive plants and arthropods are common (Aizen et al. 2008; Barthell et al. 2001; Helms and Vinson 2002; Holway et al. 2002; Ness and Bronstein 2004; O'Dowd et al. 2003), the effects of these associations on native plant and arthropod communities are case-dependent, and further research will be needed to determine the ecological and economic impact of the synergy between OWBs and *B. rubrostriata*. Because *B. rubrostriata* is a disease vector in sugarcane (Hanboonsong et al. 2006) and OWBs grow well in disturbed areas (Coyne and Bradford 1985), effective management tools will be necessary to reduce the densities of OWBs where susceptible crops are grown.

Increased litter generally results in an increase in soil moisture, available nutrients, and decomposition by microbial communities, all of which may benefit decomposer arthropods (Gratton and Denno 2006; Kappes et al. 2007; Wolkovich 2010). Generally, increases in litter associated with invasive plants result in increased abundance of detritivores (reviewed in Litt et al., in press). Although litter was abundant in native plant and OWB communities (Chapter 2), detritivores were more abundant in native plant communities. Plots with added seed also had more detritivores than monocultures of OWBs during moderate drought (Chapter 4), and this difference was greater than the number of detritivores observed in monocultures of OWBs during extreme drought when litter was abundant. Reed et al. (2005) documented decreased decomposition rates and high carbon:nitrogen ratios in OWBs, which may explain why

OWBs provided poor-quality habitat for detritivores, as echoed in Cord (2011). However, the differences we observed between detritivore communities were driven by an invasive pillbug (*Armadillidium vulgare*); native detritivores may have been negatively affected by the invasive arthropod rather than affected by the invasive plant (Ellis et al. 2000; Frouz et al. 2008). Experiments that provide arthropods with different compositions of plant litter may elucidate the impacts of plant invasion on native detritivore communities (Wardle et al. 2004; Wolkovich 2010).

Although we observed changes in the diversity and structure of native plants following simple disturbance and seeding, we were unable to alter chemical properties of the soil through our field experiment, and our soil modification techniques were inhibited by the inherent chemistry of the soil (Chapter 3), by the abundance of certain arthropod groups (Chapter 4), or other factors. Altering soil pH with lime in heavy clay soils may not be economically feasible for landowners, and applying carbon to soils where ants are abundant may not be effective. However, the combined effects of traditional and alternative restoration strategies to reduce OWBs in grasslands have yet to be explored, and native plant communities could benefit from changes in soil conditions (Heneghan et al. 2008). Prescribed burns, for example, can increase the competitive ability of OWBs as a result of increased soil nitrogen (Berg 1993), and subsequent additions of carbon to reduce the availability of nitrogen to OWBs may promote the establishment of native plants and arthropods.

It is difficult to generalize patterns observed from plant invasion in field studies where multiple disturbances are present (Didham et al. 2005; Levine et al. 2003). Field experiments that simulate conditions in the presence and absence of other stressors may elucidate the mechanisms behind plant invasion (Levine et al. 2003). We were able to control for the effects of drought through our microcosm experiment, and by comparing our results with the field experiment, we were able to determine that simple soil disturbance in combination with seeding can reduce dominance of OWBs (Chapter 3). However, other stressors, like invasive arthropods, may limit our understanding of how soil modification techniques may affect restoration of arthropod communities (Chapter 4). Future studies can build on our findings by incorporating restoration tools that mitigate the effects of invasive arthropods in restored areas, or compare our findings to experimental invasions of plants or arthropods to help develop a mechanistic understanding of invasion on native communities (Levine et al. 2003).

The importance of arthropods as indicators of restoration success in grasslands should not be underestimated, as arthropods are responsible for many ecological functions, including pollination, decomposition, and herbivory (Archer and Pyke 1991; Burger et al. 2003; Folgarait 1998; Potts et al. 2010; Wilson 1987). Changes in the composition of arthropods in grassland ecosystems associated with plant invasion can alter habitat quality for many fauna (Hickman et al. 2006; Wiens and Rotenberry 1979; Wilson 1987; Woodin et al. 2010). As the frequency and intensity of anthropogenic disturbances increase the likelihood of plant invasions (Bradley et al. 2009; IPCC 2007;

Hobbs et al. 2009; Tylianakis 2010), successful conservation and restoration efforts will likely require restoring habitat for native arthropod communities.

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APPENDICES

APPENDIX A

REFERENCES USED TO ASSIGN ARTHROPOD FUNCTIONAL GROUPS, SUMMERS 2011-2013, WELDER WILDLIFE FOUNDATION, SAN PATRICIO CO., TEXAS

Appendix A. References used to assign arthropod functional groups, summers 2011-2013, Welder Wildlife Foundation, San Patricio Co., Texas. Unless specified here, all arthropods were assigned to a functional group based on Triplehorn and Johnson (2005).

Order	Family	Subfamily	Species	Reference
Coleoptera	Anthicidae	Anthicinae	*Acanthinus scitulus*	Telnov 2011
		Notoxinae	*Notoxus monodon*	Telnov 2011
	Anthribidae		*Omiscus spp.*	Valentine 1998
			Trigonorhinus spp.	Valentine 1998
	Bothrideridae		*Bothrideres geminatus*	Stephan 1989
	Buprestidae	Agrilinae	*Agrilus muticus*	Nelson et al. 2008
			Agrilus ornatulus	Nelson et al. 2008
			Taphrocerus spp.	Nelson et al. 2008
		Buprestinae	*Spectralia robusta*	Nelson et al. 2008
			Spectralia spp.	Nelson et al. 2008
		Polycestinae	*Acmaeodera bowditchi*	Nelson et al. 2008
	Cleridae	Tilinae	*Clerida balteata*	Arnett et al. 2002
			Cymatodera balteata	Arnett et al. 2002
	Coccinellidae	Scyminae	*Brachiacantha quadrillum*	Arnett et al. 2002
	Dermestidae		*Cryptorhopalum* spp.	Beal 1995
	Hybosoridae	Hybosorinae	*Hybosorus illigeri*	Arnett et al. 2002
	Latridiidae	Corticariinae	*Melanophthalma* spp.	Arnett et al. 2002
		Latridiinae	*Cartodere* spp.	Arnett et al. 2002
			Unknown spp.	Arnett et al. 2002
	Phalacridae		*Olibrus* spp.	Gimmel 2013
	Scarabaeidae	Cetoniinae	*Euphoria sepulcralis*	Orozco 2012

(Appendix A continued)

Order	Family	Subfamily	Species	Reference
Coleoptera	Scarabaeidae	Dynastinae	*Dyscinetus morator*	Staines 1998
	Staphylinidae	Aleocharinae	*Phanerota fasciata*	Arnett et al. 2000
			Unknown spp.	Arnett et al. 2000
		Paederinae	*Astenus* spp.	Arnett et al. 2000
		Pselaphinae	*Cylindrarctus crinifer*	Arnett et al. 2000
		Scydmaeninae	*Euconnus* spp.	Arnett et al. 2000
		Steininae	Unknown spp.	Arnett et al. 2000
	Tenebrionidae	Diaperinae	*Platydema excavatum*	Triplehorn 1965
			Poecilocrypticus formicophilus	Triplehorn 1965
			Unknown spp.	Triplehorn 1965
		Lagriinae	*Paratenetus punctatus*	Triplehorn 1965
Diplura	Campypoedidae		Unknown spp.	Arnett 2000
	Japygidae		Unknown spp.	Arnett 2000
Diptera	Phoridae		*Apocephalus* spp.	Waller and Moser 1990
			Pseudacteon spp.	Plowes et al. 2009
Hemiptera	Alydidae	Alydinae	*Alydus eurinus*	Arnett 2000
			Alydus spp.	Arnett 2000
		Micrelytrinae	*Protenor* spp	Arnett 2000
	Geocoridae		*Geocoris* spp.	Cassis and Gross 2002
	Miridae	Byocorinae	*Sixeonotus albicornis*	Schuh 2013
		Deraeocorinae	*Hyaliodes* spp.	Schuh 2013
		Mirinae	*Megaloceroea* spp.	Schuh 2013
			Neurocolpus spp.	Schuh 2013
			Oncerometopus spp.	Schuh 2013
			Polymerus basalis	Schuh 2013
			Ploymerus spp.	Schuh 2013
			Trigonotylus spp.	Schuh 2013

			Unknown spp.	Schuh 2013

(Appendix A continued)

Order	Family	Subfamily	Species	Reference
Hemiptera	Miridae	Orthotylinae	*Lopidea major*	Schuh 2013
		Phylinae	*Plagiognathus albatus*	Schuh 2013
			Plagiognathus spp.	Schuh 2013
			Pseudatomoscelis seriatus	Schuh 2013
Mesostigmata	Parantennullidae			Krantz and Walter 2009
	Parasitidae			Krantz and Walter 2009
Oniscidea	Armadillidiidae		*Armadillidium vulgare*	Paris 1964
	Porcellionidae		*Acareoplastes* spp.	Richardson 1905
Opilioacarida	Opiloacaridae		Unknown spp.	Krantz and Walter 2009
Sarcoptiformes	Acaridae			Krantz and Walter 2009
	Cymbaeremaeidae			Krantz and Walter 2009
	Euphthiracaridae			Krantz and Walter 2009
	Galumnidae			Krantz and Walter 2009
	Lohmanniidae			Krantz and Walter 2009
	Mochlozetidae			Krantz and Walter 2009
	Nothridae			Krantz and Walter 2009
Trombidiformes	Anystidae			Krantz and Walter 2009
	Bdellidae			Krantz and Walter 2009
	Calypstomatidae			Krantz and Walter 2009
	Erythraeidae			Krantz and Walter 2009
	Parantennullidae			Krantz and Walter 2009

	Smarididae			Krantz and Walter 2009
	Stigmaeidae			Krantz and Walter 2009

(Appendix A continued)

Order	Family	Subfamily	Species	Reference
Trombidiformes	Tetryanchidae			Krantz and Walter 2009

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APPENDIX B

ARTHROPOD SPECIES OBSERVED DURING ALL SAMPLING SEASONS FOR KLEBERG BLUESTEM AND NATIVE PLANT COMMUNITIES, SUMMERS 2011-2013, WELDER WILDLIFE FOUNDATION, SAN PATRICIO CO., TEXAS

Appendix B: Arthropod species observed during all sampling seasons for Kleberg bluestem and native plant communities, summers 2011-2013, Welder Wildlife Foundation, San Patricio Co., Texas. We computed the % of plot samples (*n* = 40 samples for each community), and the % of total individuals (*n* = 6,975 individuals for Kleberg, *n* = 7,206 individuals for native) where we observed each species. Functional groups: A = Ants, D = Decomposers, H = Herbivores, P = Predators, Po = Pollinators.

				% of samples		% of individuals		Fun. Group
Order	**Family**	**Subfamily**	**Species**	*Kleberg*	*Native*	*Kleberg*	*Native*	
Araneae	Agelenidae		*Tegenaria* spp.	2.50	2.50	0.01	0.03	P
	Amphinectidae		*Metaltella* spp.	0.00	2.50	0.00	0.01	P
	Anyphaenidae		*Hibana futilis*	7.50	27.50	0.04	0.21	P
	Araneidae		*Araneus* spp.	2.50	5.00	0.01	0.04	P
			Neoscona spp.	0.00	7.50	0.00	0.12	P
			Unknown spp.	7.50	0.00	0.04	0.01	P
	Clubionidae		*Clubiona* spp.	10.00	17.50	0.09	0.11	P
	Corinnidae		Unknown spp.	2.50	7.50	0.01	0.04	P
	Gnaphosidae		Unknown spp.	5.00	0.00	0.03	0.00	P
	Linyphiidae		Unknown spp.	22.50	22.50	0.29	0.19	P
	Lycosidae		*Hogna* spp.	0.00	5.00	0.00	0.03	P
			Pardosa spp.	2.50	7.50	0.01	0.04	P
			Rabidosa rabida	2.50	7.50	0.01	0.07	P
			Schizocosa spp.	5.00	5.00	0.06	0.04	P
			Unknown spp.	12.50	12.50	0.09	0.08	P
	Nemesiidae		*Calisoga* spp.	0.00	2.50	0.00	0.01	P
	Oxyopidae		*Oxyopes* spp.	2.50	15.00	0.16	0.12	P
	Philodromidae		*Ebo* spp.	0.00	12.50	0.00	0.22	P
	Pholcidae		*Pholces* spp.	2.50	0.00	0.01	0.00	P
	Salticidae	Dendryphantinae	*Messua* spp.	2.50	17.50	0.03	0.14	P
			Metaphidippus spp.	7.50	2.50	0.04	0.03	P
			Phanias spp.	0.00	0.00	0.00	0.00	P

(Appendix B continued)

				% of samples		% of individuals		Fun. Group
Order	**Family**	**Subfamily**	**Species**	*Kleberg*	*Native*	*Kleberg*	*Native*	
Araneae	Salticidae	Dendryphantinae	*Phidippus* spp.	0.00	7.50	0.00	0.04	P
			Zygoballus spp.	2.50	10.00	0.01	0.08	P
			Unknown spp.	2.50	30.00	0.01	0.22	P
		Euophyrinae	Unknown spp.	0.00	2.50	0.00	0.01	P
		Pelleninae	*Habronattus texanus*	0.00	5.00	0.00	0.04	P
		Synagelinae	*Synageles noxiosus*	10.0	5.00	0.06	0.04	P
			Synageles spp.	2.50	15.00	0.01	0.14	P
		Thiodininae	Unknown spp.	0.00	2.50	0.00	0.01	P
	Theridiidae		*Latrodectus* spp.	0.00	2.50	0.00	0.01	P
	Thomisidae		*Mecaphesa* spp.	2.50	0.00	0.03	0.00	P
			Misumena vatia	5.00	10.00	0.03	0.06	P
			Xysticus spp.	30.00	0.00	0.19	0.00	P
			Unknown spp.	15.00	0.00	0.09	0.00	P
Blattodea	Blattellidae	Blattellinae	*Blattella vaga*	62.50	10.00	3.03	0.15	D
			Parcoblatta spp.	7.50	0.00	0.06	0.00	D
Coleoptera	Anobiidae		Unknown spp.	5.00	2.50	0.03	0.01	D
	Anthicidae	Anthicinae	*Acanthinus scitulus*	22.50	37.50	0.20	1.12	D
	Anthribidae		*Ormiscus* spp.	0.00	2.50	0.00	0.01	D
			Trigonorhinus spp.	2.50	0.00	0.01	0.00	Po
	Bostrichidae	Lcytinae	*Trogoxylon aequale*	0.00	2.50	0.00	0.01	D
	Buprestidae	Buprestinae	*Spectrlia robusta*	2.50	0.00	0.01	0.00	H
	Brachypteridae		*Bracypterus* spp.	2.50	0.00	0.01	0.00	D

(Appendix B continued)

				% of samples		% of individuals		Fun. Group
Order	**Family**	**Subfamily**	**Species**	*Kleberg*	*Native*	*Kleberg*	*Native*	
Coleoptera	Carabidae	Cicindelinae	*Dromochorus welderensis*	2.50	0.00	0.01	0.00	P
		Harpalinae	*Amblygnathus subtinctus*	0.00	2.50	0.00	0.01	P
			Harpalus spp.	0.00	2.50	0.00	0.01	P
			Lebia rufopleura	0.00	2.50	0.00	0.01	P
			Loxandrus spp.	7.50	22.50	0.04	0.33	P
			Notiobia spp.	7.50	5.00	0.08	0.03	P
			Systomus americanus	2.50	0.00	0.01	0.00	P
			Unknown spp.	0.00	2.50	0.00	0.01	P
		Scaritinae	*Scarites subterraneus*	5.00	5.00	0.03	0.03	P
	Carabidae		Larvae	2.50	10.00	0.01	0.06	P
	Cerambycidae	Lamiinae	*Spalacopsis texana*	2.50	10.00	0.01	0.07	H
	Chrysomelidae	Bruchinae	*Acanthoscelides* spp.	0.00	2.50	0.00	0.01	H
			Mimosestes spp.	0.00	2.50	0.00	0.03	H
			Stator spp.	0.00	2.50	0.00	0.01	H
		Cryptocephalinae	*Pachybrachis brevicornis*	0.00	2.50	0.00	0.03	H
		Galerucinae	*Chaetocnema* spp.	2.50	2.50	0.01	0.01	H
			Disonchya fumata	0.00	2.50	0.00	0.01	H
			Epitrix fasciata	0.00	2.50	0.00	0.01	H
			Monoxia spp.	0.00	5.00	0.00	0.04	H
			Syphrea nana	2.50	2.50	0.01	0.01	H
	Chrysomelidae		Larvae	2.50	2.50	0.01	0.01	H

(Appendix B continued)

				% of samples		% of individuals		Fun. Group
Order	**Family**	**Subfamily**	**Species**	*Kleberg*	*Native*	*Kleberg*	*Native*	
Coleoptera	Cleridae	Tillinae	*Cymatodera balteata*	0.00	2.50	0.00	0.01	H
	Curculionidae	Cryptorhynchinae	*Tyloderma pseudofoveolatum*	0.00	2.50	0.00	0.01	H
			Tyloderma sphaerocarpae	0.00	2.50	0.00	0.01	H
			Tyloderma spp.	0.00	2.50	0.00	0.01	H
		Curculioninae	*Anthonomus elogatus*	0.00	2.50	0.00	0.01	H
			Smicronyx lineolatus	0.00	7.50	0.00	0.06	H
			Smicronyx spp.	0.00	10.00	0.00	0.06	H
		Dryopthorinae	*Sitophilus* spp.	0.00	2.50	0.00	0.01	H
		Scolytinae	Unknown spp.	2.50	12.50	0.01	0.07	H
	Dermestidae		*Cryptorhopalum* spp.	2.50	2.50	0.01	0.05	Po
			Unknown spp.	7.50	0.00	0.08	0.00	D
			Larvae	35.00	7.50	0.27	0.05	D
	Elateridae	Agrypninae	*Aeolus* spp.	20.00	15.00	0.11	0.11	H
		Elaterinae	*Glyphonyx bimarginatus*	0.00	2.50	0.00	0.01	H
			Melanotus spp.	0.00	10.00	0.00	0.12	H
			Unknown spp.	2.50	0.00	0.01	0.00	H
	Elateridae		Larvae	2.50	10.00	0.03	0.42	H
	Endomychidae	Merophysiinae	*Holoparamecus* spp.	0.00	2.50	0.00	0.01	D
	Erotylidae	Erotylinae	*Dacne* spp.	0.00	2.50	0.00	0.01	D
	Hybosoridae	Hybosorinae	*Hybosorus illigeri*	0.00	2.50	0.00	0.01	D

(Appendix B continued)

				% of samples		% of individuals		Fun. Group
Order	**Family**	**Subfamily**	**Species**	*Kleberg*	*Native*	*Kleberg*	*Native*	
Coleoptera	Latridiidae	Corticariinae	*Melanophthalma* spp.	30.00	55.0	0.29	0.69	D
		Latridiinae	*Cartodere* spp.	0.00	7.50	0.00	0.04	D
	Latridiidae		Larvae	7.50	2.50	0.06	0.01	D
	Mordellidae		*Mordella atrata*	2.50	5.00	0.01	0.04	Po
	Mycetophagidae		*Berginus nigricolor*	0.00	2.50	0.00	0.01	D
	Nitidulidae		*Epuraea* spp.	0.00	5.00	0.00	0.03	D
			Nitidula spp.	12.50	0.00	0.07	0.00	D
	Phalacridae		*Olibrus* spp.	2.50	10.00	0.01	0.06	Po
	Salpingidae		*Dacoderus steineri*	0.00	10.00	0.00	0.06	P
			Larvae	0.00	2.50	0.00	0.01	D
	Scarabaeidae	Scarabaeinae	*Canthon cyanellus*	0.00	5.00	0.00	0.03	D
			Canthon vigilans	15.00	10.00	0.11	0.06	D
	Scraptiidae	Anaspidinae	*Diclidia* spp.	2.50	15.00	0.01	0.12	H
	Scraptiidae		Larvae	0.00	10.00	0.00	0.07	H
	Silvanidae	Silvaninae	*Cathartosilvanus imbellis*	0.00	10.00	0.00	0.06	H
	Staphylinidae	Aleocharinae	*Phanerota fasciata*	15.00	12.50	0.40	0.44	D
			Unknown spp.	10.00	22.50	0.06	0.57	D
		Paederinae	*Astenus* spp.	0.00	2.50	0.00	0.01	P
		Pselaphinae	*Cylindrarctus crinifer*	2.50	0.00	0.01	0.00	D
	Tenebrionidae	Diaperinae	*Poecilocrypticus formicophilus*	2.50	25.00	0.01	0.18	D
		Pimeliinae	*Armalia texanus*	0.00	37.50	0.00	0.54	D
		Stenochiinae	*Xylopinus* spp.	0.00	2.50	0.00	0.01	D

(Appendix B continued)

				% of samples		% of individuals		Fun. Group
Order	**Family**	**Subfamily**	**Species**	*Kleberg*	*Native*	*Kleberg*	*Native*	
Coleoptera	Tenebrionidae	Tenebrionidae	*Blapstinus fortis*	0.00	10.00	0.00	0.06	D
			Eleodes acutus	0.00	2.50	0.00	0.01	D
	Tenebrionidae		Larvae	5.00	10.00	0.03	0.11	D
	Trogidae		*Trox sonorae*	0.00	2.50	0.00	0.01	D
Coleoptera			Larvae	0.00	2.50	0.00	0.01	
Collembola	Entomobryidae		*Entomobrya* spp.	77.5	95.00	1.92	5.27	D
	Hypogastruridae		Unknown spp.	7.50	7.50	1.41	0.07	D
	Isotomidae		*Isotoma* spp.	12.50	37.50	0.38	0.89	D
	Sminthuridae		Unknown spp.	47.50	42.50	1.29	0.74	D
	Tomoceridae		*Tomocerus minor*	2.50	5.00	0.03	0.03	D
Dermaptera	Anisolabididae		*Anisolabis maritime*	2.50	30.0	0.01	0.25	P
			Euborellia annulipes	0.00	2.50	0.00	0.03	D
Diplura	Japygidae		Unknown spp.	10.00	7.50	0.07	0.06	P
Diptera	Anthomycidae		Unknown spp.	5.00	0.00	0.06	0.00	D
	Chloropidae	Chloropinae	Unknown spp.	12.50	17.50	0.11	0.11	H
		Oscinellinae	*Liohippelates* spp.	32.50	15.00	0.69	0.12	D
	Culicidae		*Anopheles* spp.	0.00	2.50	0.00	0.01	P
	Dolichopodidae		*Medetera* spp.	0.00	2.50	0.00	0.01	P
	Drosophilidae		*Hippelates* spp.	0.00	2.50	0.00	0.03	D
	Fannidae		Unknown spp.	12.50	2.50	0.16	0.01	D
	Phoridae		*Apocephalus* spp.	0.00	2.50	0.00	0.01	P
			Megaselia spp.	30.00	30.00	0.19	0.31	D
			Pseudacteon spp.	0.00	2.50	0.00	0.01	P
	Sarcophagidae		Unknown spp.	2.50	0.00	0.03	0.00	D
	Scatopsidae		Unknown spp.	15.00	25.00	0.22	0.19	D
	Sciaridae		*Sciara* spp.	10.00	7.50	0.14	0.04	D

(Appendix B continued)

				% of samples		% of individuals		Fun. Group
Order	**Family**	**Subfamily**	**Species**	*Kleberg*	*Native*	*Kleberg*	*Native*	
Diptera	Sciaridae		*Zygoneura* spp.	0.00	2.50	0.00	0.03	D
			Unknown spp.	0.00	0.00	0.07	0.00	D
	Sphaeoceridae		Unknown spp.	0.00	2.50	0.00	0.01	D
	Tachinidae	Tachininae	Unknown spp.	2.50	5.00	0.01	0.03	P
Geophilomorpha			Unknown spp.	2.50	0.00	0.01	0.00	P
Hemiptera	Aleyrodidae		*Bemisia* spp.	0.00	7.50	0.00	0.09	H
	Alydidae		*Alydus* spp.	2.50	7.50	0.01	0.04	H
	Anthocoridae		*Orius* spp.	10.00	7.50	0.07	0.07	P
	Aphidae		Unknown spp.	7.50	5.00	0.04	0.03	H
	Berytidae		*Pronotacantha* spp.	0.00	5.00	0.00	0.04	H
	Cicadellidae	Cicadellinae	*Chlorotettix* spp.	0.00	5.00	0.00	0.11	H
			Draeculacephala Minerva	0.00	5.00	0.00	0.08	H
			Draeculacephala savannhae	2.50	2.50	0.01	0.03	H
			Draeculacephala spp.	2.50	5.00	0.01	0.03	H
			Neokola dolobrata	5.00	0.00	0.03	0.00	H
			Pagaronia spp.	2.50	2.50	0.01	0.01	H
			Immature	2.50	12.50	0.01	0.11	H
		Deltocephalinae	*Athysanus* spp.	0.00	2.50	0.00	0.11	H
			Balclutha rubrostriata	47.50	0.00	26.24	0.00	H
			Immature	2.50	0.00	0.01	0.00	H
		Idiocerinae	*Idiocerus* spp.	0.00	7.50	0.00	0.04	H
		Ledrinae	*Xerophloea minor*	0.00	2.50	0.00	0.03	H
		Typhlocybinae	*Typhlocyba* spp.	2.50	2.50	0.01	0.01	H

(Appendix B continued)

				% of samples		% of individuals		Fun. Group
Order	**Family**	**Subfamily**	**Species**	*Kleberg*	*Native*	*Kleberg*	*Native*	
Hemiptera	Cicadellidae		Immature	10.00	20.00	0.07	0.19	H
	Cixiidae		*Oliarus* spp.	0.00	10.00	0.00	0.10	H
	Cydnidae		*Amnestus* spp.	0.00	2.50	0.00	0.01	H
			Pangeas spp.	0.00	2.50	0.00	0.04	H
			Unknown spp.	2.50	5.00	0.03	0.04	H
	Delphacidae		Unknown spp.	0.00	2.50	0.00	0.03	H
	Dictyopharidae	Dictyopharinae	*Rhynchomitra recurva*	2.50	0.00	0.01	0.00	H
	Issidae	Caliscelinae	*Fitchiella* spp.	2.50	2.50	0.01	0.01	H
	Largidae		Immature	2.50	0.00	0.01	0.00	H
	Lygaeidae		*Xyonysius* spp.	5.00	5.00	0.06	0.03	H
	Miridae	Mirinae	*Neurocolpus* spp.	2.50	2.50	0.01	0.01	H
		Phylinae	*Plagiognathus* spp.	0.00	2.50	0.00	0.01	H
			Polymerus spp.	2.50	0.00	0.01	0.00	H
			Pseudatomoscelis seriatus	5.00	10.00	0.04	0.11	H
	Miridae		Immature	0.00	5.00	0.00	0.04	H
	Nabidae		*Nabis* spp.	2.50	0.00	0.01	0.00	P
			Pagasa spp.	2.50	2.50	0.01	0.01	P
			Immature	0.00	2.50	0.00	0.01	P
	Pentatomidae	Pentatominae	*Mecidea minor*	25.00	0.00	0.32	0.00	H
			Trichopepla semivittata	2.50	2.50	0.01	0.01	H
			Immature	0.00	10.00	0.00	0.18	H
	Pseudococcidae		Immature	0.00	5.00	0.00	0.03	H
	Psyllidae		*Heteropsylla* spp.	0.00	2.50	0.00	0.01	H
			Unknown spp.	5.00	10.00	0.03	0.10	H

(Appendix B continued)

				% of samples		% of individuals		Fun. Group
Order	**Family**	**Subfamily**	**Species**	*Kleberg*	*Native*	*Kleberg*	*Native*	
Hemiptera	Reduviidae	Harpactorinae	*Apiomerus spissipes*	0.00	2.50	0.00	0.01	P
			Repita taurus	7.50	7.50	0.04	0.07	P
			Zelus spp.	15.00	2.50	0.11	0.01	P
		Peiratinae	*Melanolestes* spp.	2.50	0.00	0.01	0.00	P
		Stenopodainae	*Oncocephalus* spp.	2.50	5.00	0.01	0.02	P
	Reduviidae		Immature	2.50	2.50	0.01	0.01	P
	Rhopalidae		Liorhyssus spp.	2.50	12.50	0.03	0.17	H
			Immature	2.50	10.00	0.03	0.09	H
	Thyreocoridae		*Corimelaena* spp.	2.50	0.00	0.01	0.00	H
	Tingidae		*Corythucha* spp.	0.00	5.00	0.00	0.87	H
			Immature	0.00	2.50	0.00	0.01	H
Hymenoptera	Apidae	Xylocopinae	*Ceratina* spp.	0.00	5.00	0.00	0.06	Po
	Bethylidae	Epyrinae	*Laelius pedatus*	0.00	2.50	0.00	0.01	P
		Pristocerinae	*Pristocera hyaline*	5.00	20.0	0.03	0.11	P
	Braconidae		Unknown spp.	0.00	25.00	0.00	0.21	P
	Chalcididae		*Brachymeria* spp.	0.00	2.50	0.00	0.01	P
			Conura igneoides	0.00	2.50	0.00	0.01	P
	Encyrtidae		*Bothriothorax* spp.	2.50	0.00	0.01	0.00	P
			Cheiloneurus spp.	0.00	2.50	0.00	0.01	P
			Encyrtus spp.	7.50	2.50	0.06	0.01	P
			Unknown spp.	20.00	15.00	0.20	0.09	P
	Eulophidae		Unknown spp.	0.00	5.00	0.00	0.03	P
	Eupelmidae		*Arachnophaga* spp.	0.00	2.50	0.00	0.01	P
			Eupelmus spp.	0.00	2.50	0.00	0.01	P

(Appendix B continued)

				% of samples		% of individuals		Fun. Group
Order	**Family**	**Subfamily**	**Species**	*Kleberg*	*Native*	*Kleberg*	*Native*	
Hymenoptera	Eupelmidae		Unknown spp.	12.50	7.50	0.09	0.04	P
	Eurytomidae		Unknown spp.	0.00	2.50	0.00	0.01	P
	Evaniidae		*Evaniella* spp.	0.00	2.50	0.00	0.01	P
	Formicidae	Dolichoderinae	*Forelius mccooki*	12.50	25.00	0.09	0.92	A
			Forelius pruinosus	32.50	37.50	0.69	0.87	A
			Tapinoma sessile	40.00	15.00	0.80	0.18	A
		Ectioninae	*Neviamyrmex nigrescens*	0.00	2.50	0.00	0.22	A
		Formicinae	*Camponotus* spp.	2.50	2.50	0.01	0.03	A
			Nylanderia terricola	22.50	42.50	0.16	1.21	A
			Nylanderia spp.	0.00	5.00	0.00	0.11	A
		Myrmicinae	*Crematogaster* spp.	0.00	2.50	0.00	0.01	A
			Monomorium minimum	17.50	10.00	0.14	0.07	A
			Solenopsis geminata[†]	50.00	77.50	0.97	5.44	A
			Solenopsis invicta	72.50	70.00	2.37	4.90	A
			Stenamma spp.	12.50	15.00	0.14	0.19	A
			Tetramorium bicarinatum	5.00	2.50	0.11	0.03	A
			Tetramorium spp.	2.50	10.00	0.04	0.11	A
		Pseudomyrmecinae	*Pseudomyrmex gracilis*	0.00	2.50	0.00	0.01	A
			Pseudomyrmex pallidus	0.00	10.00	0.00	0.08	A

(Appendix B continued)

				% of samples		% of individuals		Fun. Group
Order	**Family**	**Subfamily**	**Species**	*Kleberg*	*Native*	*Kleberg*	*Native*	
Hymenoptera	Formicidae	Pseudomyrmicinae	*Pseudomyrmex* spp.	0.00	2.50	0.00	0.01	A
	Ichneumonidae	Anomaloninae	Unknown spp.	0.00	2.50	0.00	0.01	P
		Cryptinae	*Gelis* spp.	17.50	10.00	0.14	0.05	P
		Ophioninae	Unknown spp.	0.00	2.50	0.00	0.01	P
	Mutillidae		*Dasymutilla* spp.	0.00	2.50	0.00	0.01	P
	Mymaridae		Unknown spp.	2.50	17.50	0.01	0.10	P
	Platygastridae		*Platygaster* spp.	12.50	5.00	0.16	0.03	P
	Pompilidae	Pepsinae	*Auplopus* spp.	0.00	2.50	0.00	0.01	P
	Pteromalidae		Unknown spp.	12.50	10.00	0.11	0.08	P
	Scelionindae	Scelioninae	*Baeus* spp.	2.50	5.00	0.01	0.03	P
			Idris spp.	2.50	0.00	0.01	0.00	P
			Macrotelia spp.	15.00	7.50	0.11	0.04	P
			Scelio spp.	15.00	10.00	0.12	0.07	P
		Teleasinae	*Trimorus* spp.	5.00	15.00	0.03	0.12	P
		Telenominae	*Eumicrosoma* spp.	25.00	52.50	0.29	0.71	P
			Telenomus spp.	15.00	7.50	0.17	0.07	P
			Trissolcus spp.	25.00	0.00	0.27	0.00	P
	Tenethredidae		Unknown spp.	0.00	5.00	0.00	0.04	H
	Torymidae		Unknown spp.	12.50	0.00	0.09	0.00	P
	Trichogrammatidae		Unknown spp.	32.50	27.50	0.59	0.15	P
	Vespidae	Polistinae	*Polistes exclamans*	2.50	0.00	0.01	0.00	P
Hymenoptera			Larvae	0.00	2.50	0.00	0.01	
Lepidoptera	Acrolophidae		*Amydria* spp.	2.50	7.50	0.01	0.04	H
	Geometridae		Unknown spp.	0.00	5.00	0.00	0.00	H
	Hesperidiidae	Pyrgininae	*Pyrgus communis*	2.50	0.00	0.01	0.0	Po
	Hesperidiidae		Larvae	2.50	0.00	0.01	0.0	H

(Appendix B continued)

				% of samples		% of individuals		Fun. Group
Order	**Family**	**Subfamily**	**Species**	*Kleberg*	*Native*	*Kleberg*	*Native*	
Lepidoptera	Tineidae		Unknown spp.	5.00	7.50	0.06	0.04	D
Mesostigmata	Parantennullidae			0.00	2.50	0.00	0.04	P
	Parasitidae			0.00	5.00	0.00	0.03	P
Microcoryphia	Machilidae		Unknown spp.	2.50	20.00	0.01	0.15	D
Neuroptera	Chrysopidae	Chrysopinae	*Chrysopa* spp.	12.50	0.00	0.15	0.00	P
Oniscidea	Armadillidiidae		*Armadillidium vulgare*	65.00	100.00	1.58	47.46	D
Opilioacariformes	Opiloacaridae			0.00	2.50	0.00	0.01	D
Opiliones	Cosmetidae		*Vonones* spp.	17.50	42.50	0.56	0.61	P
	Sclerosomatidae		*Leiobunum* spp.	2.50	0.00	0.01	0.00	P
			Unknown spp.	0.00	12.50	0.00	0.14	P
Orthroptera	Acrididae	Melanoplinae	*Melanoplus* spp.	0.00	12.50	0.00	0.11	H
			Unknown spp.	2.50	2.50	0.01	0.01	H
		Oedipodinae	*Arphia* spp.	0.00	2.50	0.00	0.01	H
			Chortophaga viridifasciata	0.00	7.50	0.00	0.07	H
	Gryllidae	Gryllinae	*Gryllus* spp.	52.50	47.50	0.50	0.79	D
	Mogoplistidae		*Cycloptilum* spp.	5.00	5.00	0.03	0.03	D
	Tetrigidae		Unknown spp.	0.00	2.50	0.00	0.01	D
	Tettigoniidae	Conocephalinae	*Conocephalus* spp.	10.00	22.50	0.11	0.10	H
			Immature	0.00	5.00	0.00	0.06	H
Phasmida	Heteronemiidae		*Parabacillus* spp.	12.50	5.00	0.07	0.03	H
Protura	Eosemtomidae		*Eosemtomon* spp.	0.00	2.50	0.00	0.01	D
Pseudoscorpionida	Syarinidae		*Syarinus* spp.	0.00	15.00	0.00	0.10	P
Psocoptera	Lachisellidae		*Lachisella* spp.	15.00	7.50	0.22	0.07	D
	Liposcelididae		*Liposcelis* spp.	32.50	27.50	0.30	0.21	D
	Pachytroctidae		*Tapinella* spp.	15.00	7.50	0.09	0.04	D

(Appendix B continued)

				% of samples		% of individuals		Fun. Group
Order	**Family**	**Subfamily**	**Species**	*Kleberg*	*Native*	*Kleberg*	*Native*	
Psocoptera	Pseudocaeciliidae		*Pseudocaecilius citricola*	0.00	5.00	0.00	0.07	H
Sarcoptiformes	Acaridae			0.00	5.00	0.00	0.03	D
	Cymbaeremaeidae			2.50	0.00	0.01	0.00	D
	Euphthiracaridae			0.00	40.00	0.00	5.08	H
	Galumnidae			47.50	20.00	1.19	0.19	D
	Mochlozetidae			92.50	70.00	32.09	1.53	H
	Nothridae			17.50	5.00	0.26	0.03	D
Scorpionida	Buthidae		*Centruroides vittatus*	40.00	55.00	0.33	0.43	P
Scutigeromorpha	Scutigeridae		*Dendrothereua homa*	5.00	10.00	0.04	0.14	P
			Scutiger coleoptera	7.50	7.50	0.04	0.06	P
Spirobolida	Spirobolidae		*Narceus* spp.	0.00	2.50	0.00	0.01	D
Thysanoptera	Aeolothripidae		*Aeolothrips* spp.	40.00	35.00	0.14	0.78	P
	Phlaeothripidae		*Haplothrips* spp.	60.00	35.00	1.9	0.57	P
	Thripidae		*Scolothrips* spp.	35.00	22.50	0.83	0.18	P
Trombidiformes	Anystidae			97.50	92.50	5.23	2.64	P
	Bdellidae			5.00	12.50	0.03	0.12	P
	Calypstomatidae			5.00	0.00	0.03	0.00	D
	Erythraeidae			35.00	40.00	0.70	0.44	P
	Smaridae			10.00	7.50	0.07	0.04	P
	Tetraynchidae			5.00	5.00	0.03	0.03	H

APPENDIX C

PLANT SPECIES OBSERVED DURING ALL SAMPLING SEASONS FOR KLEBERG BLUESTEM AND NATIVE PLANT COMMUNITIES, SUMMERS 2011-2013, WELDER WILDLIFE FOUNDATION, SAN PATRICIO CO., TEXAS

C1. Plant species observed during all sampling seasons for the Kleberg bluestem community, summers 2011-2013, Welder Wildlife Foundation, San Patricio Co., Texas. We computed the % of plot samples (*n* = 40 samples), and % of total individuals (*n* = 4,493 individuals for Kleberg) for each species.

Community	Common Name	Scientific Name	% of samples	% of individuals
Kleberg	Kleberg bluestem	*Dichanthium annulatum*	100.00	94.39
	Bundleflower	*Desmanthus depressus*	21.25	2.25
	Tickseed	*Coreopsis tinctoria*	11.25	1.69
	Honey mesquite	*Prosopis glandulosa*	8.75	0.42
	Wild petunia	*Ruellia nudiflora*	5.00	0.08
	Hooker eryngo	*Eryngium hookeri*	3.75	0.27
	Texas huisache	*Acacia smallii*	3.75	0.13
	Little barley	*Hordeum pusillum*	2.50	0.18
	Mexican hat	*Ratibida columnifera*	2.50	0.11
	Wooly croton	*Croton capitatus*	2.50	0.11
	Ground cherry	*Physalis cinerascens*	2.50	0.08
	Devil-weed	*Leucosyris spinosa*	2.50	0.02
	Seacoast sumpweed	*Iva annua*	2.50	0.02
	Redseed plantain	*Plantago rhodosperma*	1.25	0.08
	Woodsorrel	*Oxalis drummondii*	1.25	0.08
	Western ragweed	*Ambrosia cumanensis*	1.25	0.02
	White tridens	*Tridens albescens*	1.25	0.02

(Appendix C continued)

C2. Plant species observed during all sampling seasons for the native plant community, summers 2011-2013, Welder Wildlife Foundation, San Patricio Co., Texas. We computed the % of plot samples (*n* = 40 samples), and % of total individuals (*n* = 3,284 individuals for native) for each species.

Community	Common Name	Scientific Name	% of samples	% of individuals
Native	Seacoast sumpweed	*Iva annua*	61.25	33.43
	Hall's panicum	*Panicum halli var. filipes*	38.75	12.67
	Sulphur mallow	*Cienfuegosia drummondii*	37.50	9.07
	Brown seed paspalum	*Paspalum plicatulum*	37.50	4.75
	Tickseed	*Coreopsis tinctoria*	25.00	11.51
	Pink smartweed	*Polygonum pensylvanicum*	18.75	1.86
	Knotroot bristlegrass	*Setaria ramiseta var. formula*	15.00	3.93
	Western sedge	*Carex occidentalis*	15.00	1.95
	Wild petunia	*Ruellia nudiflora*	12.50	4.38
	Woodsorrel	*Oxalis drummondii*	12.50	1.61
	Western ragweed	*Ambrosia cumanensis*	6.25	11.33
	Wild mercury	*Argythamnia humilis*	6.25	0.55
	Hooker eryngo	*Eryngium hookeri*	5.00	0.52
	Texas huisache	*Acacia smallii*	5.00	0.21
	Prairie tea	*Croton monathogynous*	5.00	0.18
	Bundleflower	*Desmanthus depressus*	3.75	0.52
	Little barley	*Hordeum pusillum*	3.75	0.27
	Texas broomweed	*Gutierrezia texana*	3.75	0.18
	Texas bristlegrass	*Setaria texana*	2.50	0.27
	Blackbrush acacia	*Acacia rigidula*	2.50	0.24
	Wright false mallow	*Malvastrum aurantiacum*	2.50	0.12
	Honey mesquite	*Prosopsis glandulosa*	2.50	0.06
	Texas signalgrass	*Urochloa texana*	1.25	0.03
	Widow's tears	*Commelina erecta*	1.25	0.03

APPENDIX D

SPECIES COMPOSITION OF ENDO/ECTO MYCORRHIZAL SEED MIX FOR SOIL MODIFICATION TREATMENTS, SUMMERS 2011-2013, WELDER WILDLIFE FOUNDATION, SAN PATRICIO CO., TEXAS

Appendix D: Species composition of endo/ecto mycorrhizal seed mix for soil modification treatments, summers 2011-2013, Welder Wildlife Foundation, San Patricio Co., Texas.

Species Name	% of seed mix	Propagules/g
Glomus aggregatum	0.12	55
Glomus etunicatum	0.12	55
Glomus intraradices	0.12	55
Glomus mosseae	0.12	55
Pisolithus tinctorius	49.75	22,000
Rhizoogon amylopogon	6.22	2,750
Rhizoogon fulvigleba	6.22	2,750
Rhizoogon loteolus	6.22	2,750
Rhizoogon villosullus	6.22	2,750
Scleroderma cepa	12.44	5,500
Scleroderma citrinum	12.44	5,500

APPENDIX E

SPECIES LIST AND COMPOSITION OF THE NATIVE SEED MIX, SUMMERS 2011-2013, WELDER WILDLIFE FOUNDATION, SAN PATRICIO CO., TEXAS

Appendix E: Species list and composition of the native seed mix, summers 2011-2013, Welder Wildlife Foundation, San Patricio Co., Texas.

Common Name	Species Name	Variety	% of seed mix	PLS (kg/ha)
Slender grama	*Bouteloua repens*	Dilley	34.81	4.48
Tallow weed blend	*Plantago* spp	Divot	13.05	1.68
Texas grama	*Bouteloua rigidiseta*	Atascosa	11.31	1.46
Buffalograss	*Buchloe dactyloides*	Texoka	6.61	0.85
Little bluestem	*Schizachyrium scoparium*	Common	6.53	0.84
Sideoats grama	*Bouteloua curtipendula*	Haskell	3.48	0.45
Pink pappusgrass	*Pappophorum bicolor*	Maverick	3.31	0.43
Whiplash pappusgrass	*Pappophorum vaginatum*	Webb	3.05	0.39
Bristlegrass	*Setaria* spp.	Catarina	2.44	0.31
Hairy grama	*Bouteloua hirsute*	Chaparral	1.83	0.24
Multiflowered false rhodesgrass	*Chlrois pluriflora*	Common	1.74	0.22
Arizona cottontop	*Digitaria californica*	La Salle	1.74	0.22
Hall's panicum	*Panicum halli* var *halli*	Oso	1.74	0.22
Canada wildrye	*Elymus Canadensis*	Lavaca	1.65	0.21
Hooded windmillgrass	*Chloris cucullata*	Mariah	1.13	0.15
Green sprangletop	*Leptochloa dubia*	Van Horn	0.87	0.11
Big sacaton	*Sporobolus wrightii*	Falfurrias	0.87	0.11
Shortspike windmillgrass	*Chloris subdolichostachya*	Welder	0.78	0.10
Purple prairie clover	*Dalea nana*	Cuero	0.78	0.10
Sand dropseed	*Sporobolus cryptandrus*	N/A	0.78	0.10
Awnless bush sunflower	*Simsia calva*	Plateau	0.44	0.06
Partridge pea	*Chamaecrista fasciculata*	Lark	0.35	0.04
Engelmann daisy	*Engelmannia pinnatfidia*	Eldorado	0.35	0.04
Illinois bundleflower	*Desmanthus virgatus*	Sabine	0.17	0.02
False rhodesgrass	*Trichloris crinita*	Kinney	0.17	0.02

APPENDIX F

PLANT SPECIES OBSERVED DURING ALL SAMPLING SEASONS FOR THE FIELD STUDY, SUMMERS 2011-2013, WELDER WILDLIFE FOUNDATION, SAN PATRICIO CO., TEXAS

Appendix F: Plant species observed during all sampling seasons for the field study, summers 2011-2013, Welder Wildlife Foundation, San Patricio Co., Texas. We computed the % of plot samples (*n* = 300 samples), and % of total individuals (*n* = 20,587 individuals) where we observed each species.

Category	Common Name	Scientific Name	% of samples	% of individuals
Native Grasses	Bristlegrass	*Setaria* spp.	0.33	0.06
	Canada wildrye	*Elymus Canadensis*	0.33	>0.01
	Common sandbur	*Cenchrus spinifex*	6.00	0.82
	Foxtail bristlegrass	*Setaria italic*	0.67	0.06
	Fringed windmillgrass	*Chloris ciliate*	1.33	0.28
	Hall's panicum	*Panicum halli* var. *halli*	6.00	0.59
	Hooded windmillgrass	*Chloris parviflora*	2.00	0.55
	Knotroot bristlegrass	*Setaria ramiseta* var. *formula*	8.00	0.67
	Little barley	*Hordeum pusillum*	30.67	3.58
	Pink pappusgrass	*Pappophorum bicolor*	1.00	0.22
	Plains bristlegrass	*Setaria lecuopila*	0.67	0.02
	Sand dropseed	*Sporobolus cryptandrus*	0.33	0.02
	Sideoats grama	*Bouteloua curtipendula*	0.33	0.02
	Slender grama	*Bouteloua repens*	2.67	0.05
	Slim tridens	*Tridens muticus* var. *muticus*	0.33	>0.01
	Texas wintergrass	*Nassella leucotricha*	44.33	6.51
	Texas signalgrass	*Urochloa texana*	23.67	3.03
Nonnative Grasses	Bermudagrass	*Cynodon dactylon*	7.00	0.67
	Junglerice	*Echinochloa colona*	6.33	0.88
	Kleberg bluestem	*Dichanthium annulatum*	70.67	30.96
	Timothy canarygrass	*Phalaris angusta*	1.00	0.05
Forbs	Bladderpod	*Lesquerella* spp.	1.00	0.06
	Bundleflower	*Desmanthus depressus*	18.67	1.50
	Common sunflower	*Helianthus annuus*	1.33	0.48
	Cowpen daisy	*Verbesina encelioides*	0.67	0.02
	Devilweed	*Leucosyris spinosa*	14.00	1.16
	Evening primrose	*Oenothera speciosa*	0.33	0.02
	Fleabane daisy	*Erigeron philadelphicus*	7.00	0.52
	Ground cherry	*Physalis cinerascens*	0.67	0.05
	Hooker eryngo	*Eryngium hookeri*	9.00	0.42
	Knotweed leaf flower	*Phyllanthus polygonoides*	0.33	0.02
	Lemon beebalm	*Monarda citriodora*	4.33	0.24
	Mexican hat	*Ratibida columnifera*	8.00	0.43
	Morning glory	*Ipomoea trichocarpa*	0.67	0.06
	Redseed plantain	*Plantago rhodosperma*	17.00	2.28
	Scarlet pimpernel	*Anagallis arvensis*	1.67	0.09

(Appendix F continued)

Category	Common Name	Scientific Name	% of samples	% of individuals
Forbs	Seacoast sumpweed	*Iva annua*	7.67	0.48
	Silverleaf nightshade	*Solanum elaeagnifolium*	41.67	3.67
	Slim lobe celery	*Cyclospermum leptophyllum*	1.33	0.14
	Snake cotton	*Froelichia drummondii*	1.00	0.05
	Sneezeweed	*Helenium* spp.	1.33	0.05
	Sulphur mallow	*Cienfugosia drummondii*	3.33	0.26
	Texas broomweed	*Gutierrezia texana*	24.67	4.08
	Texas vervain	*Verbena officinale*	14.67	0.97
	Tickseed	*Coreopsis tinctoria*	17.67	3.63
	Western ragweed	*Ambrosia cumanensis*	0.67	0.10
	White-margined euphorbia	*Euphorbia albomarginata*	0.67	0.02
	Wild mercury	*Argythamnia humilis*	1.33	0.05
	Wooly croton	*Croton capitatus*	91.00	27.82
	Woodsorrel	*Oxalis drummondi*	10.00	0.51
	Wright's false mallow	*Malvastrum aurantiacum*	5.67	0.24
Woody Plants	Honey mesquite	*Prosopis glandulosa*	4.33	0.24
	Texas huisache	*Acacia smallii*	6.67	0.40

APPENDIX G

PLANT SPECIES OBSERVED DURING THE MICROCOSM STUDY, SUMMER 2013, WELDER WILDLIFE FOUNDATION, SAN PATRICIO CO., TEXAS

Appendix G: Plant species observed during the microcosm study, summers 2011-2013, Welder Wildlife Foundation, San Patricio Co., Texas. We computed the % of plot samples (*n* = 100 samples), and % of total individuals (*n* = 312 individuals) where we observed each species.

Category	Common Name	Scientific Name	% of samples	% of individuals
Native Grasses	Hall's panicum	*Panicum halli* var. *filipes*	3.0	0.96
	Knotroot bristlegrass	*Setaria ramiseta* var. *formula*	28.0	18.59
	Sideoats gramma	*Bouteloua repens*	4.0	2.24
	Slender gramma	*Bouteloua curtipendula*	16.0	8.33
	Texas signalgrass	*Urochloa texana*	15.0	6.08
Nonnative Grasses	Junglerice	*Echinochloa colona*	30.0	11.54
	Kleberg bluestem	*Dichanthium annulatum*	43.0	36.54
Sedges	Western sedge	*Carex occidentalis*	12.0	9.62
Forbs	Bundleflower	*Lesquerella spp.*	1.0	0.32
	White-margined euphorbia	*Euphorbia albomarginata*	16.0	5.44
	Woodsorrel	*Oxalis drummondii*	1.0	0.32

APPENDIX H

ARTHROPOD SPECIES, RELATIVE PRESENCE, AND RELATIVE ABUNDANCE OF ALL SAMPLING SEASONS FOR SOIL MODIFICATION PLOTS, SUMMERS 2011-2013, WELDER WILDLIFE FOUNDATION, SAN PATRICIO CO., TEXAS.

Appendix H: Arthropod species, relative presence (*n* = 400 samples), and relative abundance (*n* = 36,588 individuals) o
sampling seasons for soil modification plots, summers 2011-2013, Welder Wildlife Foundation, San Patricio Co., Texas.
Functional groups: A = Ants, D = Decomposers, H = Herbivores, P = Predators, Po = Pollinators.

Order	Family	Subfamily	Species	% of samples	% of individuals	Functional Group
Araneae	Agelenidae		Unknown spp.	4.00	0.05	P
	Anyphaenidae		*Hibana futilis*	12.75	0.17	P
	Araneidae		*Araneus* spp.	12.75	0.25	P
			Lariniodes spp.	1.00	<0.01	P
			Unknown spp.	4.50	0.09	P
	Clubionidae		*Clubiona* spp.	8.75	0.11	P
	Corinnidae		*Castianeira thalia*	0.25	<0.01	P
	Ctenidae		Unknown spp.	0.25	<0.01	P
	Dysderidae		*Dysdera crocata*	0.25	<0.01	P
	Gnaphosidae		Unknown spp.	6.00	0.08	P
	Linyphiidae		Unknown spp.	21.75	0.34	P
	Lycosidae		*Hogna* spp.	1.25	0.01	P
			Pardosa spp.	5.75	0.08	P
			Rabida rabidosa	11.50	0.42	P
			Schizocosa spp.	0.75	<0.01	P
			Immature	8.50	0.09	P
	Oxyopidae		*Oxyopes* spp.	16.25	0.25	P
			Peucetia longipalpis	3.00	0.05	P
			Peucetia viridians	1.75	0.03	P
			Unknown spp.	0.75	0.01	P
	Philodromidae		*Ebo* spp.	7.00	0.15	P
			Tibellus spp.	1.50	0.03	P
			Unknown spp.	4.75	0.07	P
	Pholcidae		Pholces spp.	2.00	0.02	P
	Salticidae	Dendryphantinae	*Messua* spp.	3.00	0.04	P
			Metaphiddus spp.	2.50	0.03	P
			Phanias spp.	0.50	<0.01	P

(Appendix H continued)

Order	Family	Subfamily	Species	% of samples	% of individuals	Functional Group
Araneae	Salticidae	Dendryphantinae	*Phiddipus* spp.	13.25	0.20	P
			Zygoballus spp.	5.75	0.08	P
			Immature	3.75	0.05	P
		Marpissinae	*Maevia* spp.	2.25	0.02	P
		Pelleninae	*Habronattus* spp.	1.50	0.02	P
		Salticinae	*Salticus* spp.	0.25	<0.01	P
		Synagelinae	*Synageles noxiosus*	2.25	0.03	P
			Synageles spp.	4.00	0.03	P
	Salticidae		Immature	5.25	0.07	P
	Sparassidae		*Heteropoda* spp.	0.25	<0.01	P
	Tetragnathidae		*Tetragnatha* spp.	0.75	<0.01	P
	Theredíidae		*Anelisomus* spp.	0.25	<0.01	P
			Argyrodes elevates	0.50	<0.01	P
			Latrodectus Hesperus	0.25	<0.01	P
	Thomisidae		*Mecaphesa* spp.	2.25	0.06	P
			Misumena spp.	10.50	0.26	P
			Xysticus spp.	4.50	0.10	P
			Unknown spp.	2.75	0.04	P
Blattodea	Blattellidae	Blattellinae	*Blattella vaga*	18.75	0.43	D
			Parcoblatta spp.	1.25	0.02	D
Coleoptera	Anobiidae	Anobiinae	Unknown spp.	0.25	<0.01	D
	Anthicidae	Anthicinae	*Acanthinus scitulus*	13.25	0.19	D
		Notoxinae	*Notoxus monodon*	0.25	<0.01	D
	Bostrichidae	Lyctinae	*Trogoxylon aequale*	1.25	0.01	D
	Bothrideridae		*Bothrideres geminatus*	0.50	<0.01	P
	Brentidae	Cyladinae	*Cylas formicarius*	0.25	<0.01	H
	Buprestidae	Agrilinae	*Agrilus muticus*	0.25	<0.01	Po
			Agrilus ornatulus	0.25	<0.01	H

(Appendix H continued)

Order	Family	Subfamily	Species	% of samples	% of individuals	Functional Group
Coleoptera	Buprestidae	Agrilinae	*Taphrocerus* spp.	0.50	<0.01	H
		Buprestinae	*Spectralia* spp.	0.50	<0.01	H
		Polycestinae	*Acmaeodera bowditchi*	0.25	<0.01	Po
	Brachypteridae		*Brachypterus schaefferi*	0.25	<0.01	Po
	Carabidae	Brachininae	*Brachinus alexginuus*	1.50	0.02	P
		Carabinae	*Calsoma angulatum*	0.75	<0.01	P
		Cicindelinae	*Dromochorus welderensis*	17.50	0.35	P
		Harpalinae	*Calleida punctulata*	0.25	<0.01	P
			Chlaenius orbus	0.50	<0.01	P
			Lebia spp.	0.75	<0.01	P
			Loxandrus spp.	8.00	0.12	P
			Notioba spp.	9.25	0.13	P
			Stenomorphus spp.	0.25	<0.01	P
			Syntomus americanus	0.25	<0.01	P
			Unknown spp.	0.50	<0.01	P
		Scaritinae	*Clivina* spp.	0.25	<0.01	H
			Pasimachis spp.	2.25	0.02	P
			Scarites subterraneus	0.50	<0.01	P
	Carabidae		Larvae	3.25	0.04	P
	Cerambycidae	Cerambycinae	*Megacyllene antennata*	0.25	<0.01	H
		Lamiinae	*Hippopsis lemniscata*	0.25	<0.01	H
			Spalacopsis texana	0.75	<0.01	H
	Chrysomelidae	Bruchinae	*Acanthoscelides* spp.	2.25	0.03	H
			Stator pruinosus	1.00	<0.01	H
		Cassidinae	*Cassida flaveola*	0.75	<0.01	H

(Appendix H continued)

Order	Family	Subfamily	Species	% of samples	% of individuals	Functional Group
Coleoptera	Chrysomelidae	Cassadinae	*Gratiana* spp.	0.25	<0.01	H
		Criocerinae	Unknown spp.	0.25	<0.01	H
		Cryptocephalinae	*Cryptocephalus* spp.	0.25	<0.01	H
			Dianchus auratus	0.50	<0.01	H
			Pachybrachis brevicornis	0.50	<0.01	H
			Pachybrachis duboisis	0.25	<0.01	H
		Galerucinae	*Chaetocnema* spp.	7.75	0.09	H
			Disonycha leptolineata	0.25	<0.01	H
			Epitrix fasciata	2.00	0.05	H
			Longitarsus spp.	6.00	0.14	H
			Phyllotreta aeneicollis	0.25	<0.01	H
			Triarius vittipennis	0.25	<0.01	H
			Unknown spp.	0.50	<0.01	H
		Hispinae	*Agroiconota bivittata*	0.25	<0.01	H
			Coptocycla texana	0.50	<0.01	H
			Stronglyocassis atripes	0.50	<0.01	H
			Unknown spp.	0.25	<0.01	H
	Chrysomelidae		Larvae	3.00	0.08	H
	Cleridae	Tilinae	*Clerida balteata*	0.25	<0.01	H
	Coccinellidae	Coccinellinae	*Hippodamia convergens*	0.50	<0.01	P
		Scyminae	*Brachiacantha quadrillum*	0.25	<0.01	P
	Curculionidae	Baridinae	*Apinocis* spp.	0.25	<0.01	H
		Cryptorhynchinae	*Calles cladotrichis*	0.25	<0.01	H
			Maemactes cribratus	1.50	0.02	H

(Appendix H continued)

Order	Family	Subfamily	Species	% of samples	% of individuals	Functional Group
Coleoptera	Curculionidae	Cryptorhynchinae	*Sudus* spp.	0.25	<0.01	H
			Tyloderma pseudofoveolatum	0.25	<0.01	H
			Tyloderma sphaerocarpae	0.25	<0.01	H
			Tyloderma spp.	0.75	0.01	H
		Curculioninae	*Anthonomus albopilosus*	4.25	0.06	H
			Anthonomus elongatus	0.25	<0.01	H
			Anthonomus ligatus	0.25	<0.01	H
			Anthonomus testaceosquamosus	1.25	0.02	H
			Anthonomus spp.	1.00	0.01	H
			Mymex arizonicus	0.25	<0.01	H
			Smicronyx lineolatus	0.25	<0.01	H
			Smicronyx spp.	0.25	<0.01	H
		Entiminae	*Colecerus marmoratus*	0.25	<0.01	H
			Compsus auricephalus	0.25	<0.01	H
			Cyrtepistomus castaneus	0.50	<0.01	H
			Epicaerus spp.	0.50	<0.01	H
			Thecesternus maculosus	0.25	<0.01	H
		Erirhininae	Unknown spp.	0.25	<0.01	H
		Scolytinae	*Scolytus* spp.	5.50	0.08	H
			Xyleborus spp.	1.00	0.01	H
	Dermestidae		*Attagenus* spp.	3.25	0.06	D

(Appendix H continued)

Order	Family	Subfamily	Species	% of samples	% of individuals	Functional Group
Coleoptera	Dermestidae		Unknown spp.	1.00	0.02	D
			Larvae	10.50	0.18	D
	Elateridae	Agrypninae	*Aeolus* spp.	4.75	0.06	H
		Elaterinae	*Melanotus* spp.	0.50	<0.01	H
			Unknown spp.	2.00	0.03	H
	Elateridae		Larviform	1.25	0.02	H
	Erotylidae		Unknown spp.	0.25	<0.01	D
	Hybosoridae	Hybosorinae	*Hybosorus illigeri*	0.25	<0.01	D
	Latridiidae	Corticariinae	*Melanophthalma* spp.	22.00	0.62	D
		Latridiinae	Unknown spp.	2.50	0.04	D
	Latridiidae		Larvae	3.75	0.05	D
	Mordellidae		*Mordella atrata*	1.50	0.02	Po
			Mordellistena spp.	0.75	<0.01	Po
	Nitidulidae		Unknown spp.	4.75	0.06	D
	Phalacridae		*Olibrus* spp.	4.00	0.06	Po
	Ptilidae		Unknown spp.	2.75	0.04	D
	Salpingidae		*Dacoderus sterneri*	1.25	0.01	P
	Scarabaeidae	Aphodiinae	*Ataeniopsis figurator*	0.50	<0.01	D
		Cetoniinae	*Euphoria sepulcralis*	0.25	<0.01	H
		Dynastinae	*Aphonus texanus*	1.25	0.02	D
			Dyscinetus morator	0.50	<0.01	D
			Euetheola humilis	0.25	<0.01	D
		Melonthinae	*Phyllophaga* spp.	0.25	<0.01	H
		Scarabaeinae	*Canthon vigilans*	3.00	0.03	D
			Canthon viridus	0.50	<0.01	D
			Onthophagus pennslyvanicus	1.25	0.01	D
	Scarabaeidae		Larvae	0.50	<0.01	H

(Appendix H continued)

Order	Family	Subfamily	Species	% of samples	% of individuals	Functional Group
Coleoptera	Scraptiidae	Anaspidinae	*Diclidia* spp.	0.50	<0.01	H
	Silvanidae		Unknown spp.	0.50	<0.01	H
	Sphindidae		*Odontosphindus* spp.	0.50	<0.01	D
	Staphylinidae	Aleocharinae	*Phanerota fasciata*	4.25	0.34	D
			Unknown spp.	1.00	0.01	D
		Scydmaeninae	*Euconnus* spp.	0.25	<0.01	P
		Steininae	Unknown spp.	4.00	0.04	P
	Staphylinidae		Larvae	1.25	0.02	D
	Tenebrionidae	Diaperinae	*Platydema excavatum*	1.0	0.02	D
			Poecilocrypticus formicophilus	0.75	0.01	D
			Unknown spp.	0.25	<0.01	D
		Lagriinae	*Paratenetus punctatus*	0.50	0.01	D
		Pimeliinae	*Armalia texana*	17.00	0.31	D
		Tenebrioninae	*Blapstinus fortis*	1.50	0.02	D
			Eleodes tricostatus	0.25	<0.01	D
	Tenebrionidae		Larvae	10.00	0.29	D
Coleoptera			Larvae	0.25	<0.01	
Collembola	Entomobryidae		*Entomobrya* spp.	67.50	5.49	D
			Seria bipunctata	2.75	0.03	D
	Hypogastruridae		Unknown spp.	3.00	0.08	D
	Isotomidae		*Isotoma* spp.	2.75	0.04	D
	Sminthuridae		Unknown spp.	36.00	2.03	D
	Tomoceridae		*Tomocerus minor*	8.75	0.16	D
Dermaptera	Anisolabidae		*Anisolabis maritima*	5.50	0.07	P
			Euborellia annulipes	2.00	0.03	D
	Forficulidae		*Forficula* spp.	1.00	0.01	P
Diplura	Campypoedidae		Unknown spp.	0.75	<0.01	P

(Appendix H continued)

Order	Family	Subfamily	Species	% of samples	% of individuals	Functional Group
Diplura	Japygidae			1.00	0.01	P
Diptera	Agromyzidae		Unknown spp.	6.25	0.15	D
	Anthomyiidae		Unknown spp.	0.25	<0.01	D
	Asilidae	Asilinae	*Proctacanthus* spp.	0.25	<0.01	P
		Dasypogoninae	Unknown spp.	0.25	<0.01	P
	Bombyliidae	Phthriinae	*Poecilognathus punctipennis*	3.00	0.04	Po
	Cecidomyiidae	Cecidomyiinae	Unknown spp.	2.25	0.04	D
	Ceratopogonidae		Unknown spp.	0.50	<0.01	P
	Chironomidae		Unknown spp.	0.75	<0.01	D
	Chloropidae	Chloropinae	Unknown spp.	3.50	0.05	H
		Oscinellinae	*Liohippelates* spp.	12.25	0.23	D
	Conopidae	Stylogastrinae	Unknown spp.	0.25	<0.01	Po
	Culicidae		*Anopheles quadrimaculatus*	1.25	0.01	P
	Dolichopodidae	Medeterinae	*Medetera* spp.	0.25	<0.01	P
		Scipodinae	*Condylostylus longicornis*	2.25	0.03	P
	Drosophilidae		Unknown spp.	5.75	0.09	D
	Fannidae		Unknown spp.	2.00	0.02	D
	Muscidae	Muscinae	*Musca autumnalis*	1.50	0.02	D
	Phoridae		*Apocephalus* spp.	0.75	<0.01	P
			Megaselia spp.	16.50	0.22	D
			Pseudacteon spp.	0.75	0.01	P
			Unknown spp.	2.00	0.04	D
	Pipunculidae		*Pipunculus* spp.	0.50	<0.01	P
	Sarcophagidae		Unknown spp.	0.25	<0.01	D
	Scatopsidae			16.50	0.31	D
	Scenopinidae		*Scenopinus* spp.	0.25	<0.01	Po

(Appendix H continued)

Order	Family	Subfamily	Species	% of samples	% of individuals	Functional Group
Diptera	Sciaridae		*Sciara* spp.	3.75	0.07	D
			Zygoneura spp.	4.75	0.07	D
			Unknown spp.	3.50	0.08	D
	Sciomyzidae		*Sepedon* spp.	0.25	<0.01	D
	Sepsidae		Unknown spp.	0.50	0.01	D
	Simulidae		Unknown spp.	1.00	0.02	P
	Stratomyidae		Unknown spp.	0.25	<0.01	P
	Syrphidae		*Toxomerus* spp.	0.25	<0.01	Po
	Tachinidae	Phasinae	Unknown spp.	0.25	<0.01	P
		Tachinidae	Unknown spp.	1.75	0.02	P
	Tephritidae	Tephritinae	*Campioglossa* spp.	0.50	<0.01	H
	Therevidae		Unknown spp.	0.25	<0.01	Po
	Xylophagidae		Unknown spp.	0.25	<0.01	Po
Geophilomorpha			Unknown spp.	3.50	0.04	P
Hemiptera	Aleyrodidae		*Bemisia* spp.	7.00	0.10	H
			Tretraleurodes spp.	3.25	0.06	H
			Unknown spp.	0.50	<0.01	H
	Alydidae	Alydinae	*Alydus eurinus*	3.75	0.05	H
		Micrelytrinae	*Protenor* spp	0.25	<0.01	H
	Alydidae		Immature	0.50	<0.01	H
	Anthocoridae		*Orius* spp.	6.25	0.09	P
	Aphidae		Unknown spp.	2.00	0.03	H
	Cercropidae		Unknown spp.	0.50	<0.01	H
	Cicadellidae	Cicadellinae	*Banasa* spp.	0.50	<0.01	H
			Cuerna spp.	0.25	<0.01	H
			Draeculacephala minerva	4.50	0.09	H
			Draeculacephala zaea	0.25	<0.01	H
			Draeculacephala spp.	1.00	0.01	H

(Appendix H continued)

Order	Family	Subfamily	Species	% of samples	% of individuals	Functional Group
Hemiptera	Cicadellidae	Cicadellinae	*Neokolla dolobrata*	9.25	0.20	H
			Neokolla heiroglyphica	0.50	<0.01	H
			Neokolla spp.	3.50	0.06	H
			Unknown spp.	0.25	<0.01	H
			Immature	5.50	0.09	H
		Coelidinae	Unknown spp.	13.50	0.08	H
		Deltocephalinae	*Balclutha rubrostriata*	28.00	6.44	H
			Bonneyana spp.	0.25	<0.01	H
			Chlorotettix spp.	0.25	<0.01	H
			Scaphoideus spp.	1.25	0.06	H
			Stirellus bicolor	0.25	<0.01	H
			Texanus spp.	0.75	0.02	H
			Unknown spp.	1.75	0.02	H
			Immature	0.75	<0.01	H
		Idiocerinae	*Idiocerus* spp.	2.00	0.04	H
		Ledrinae	*Xerophloea* spp.	0.50	<0.01	H
		Typhlocybinae	*Alebra* spp.	0.25	<0.01	H
			Typhlocyba spp.	0.75	<0.01	H
			Unknown spp.	0.50	<0.01	H
	Cicadellidae		Immature	19.00	0.42	H
	Cixiidae	Cixiinae	*Cixius basalis*	6.25	0.10	H
			Microledrida spp.	0.25	<0.01	H
			Immature	1.25	0.01	H
	Coccoidea			6.75	0.08	H
	Coreidae	Coreinae	*Anasa tristis*	1.25	0.03	H
			Immature	0.50	<0.01	H
	Cydnidae		*Amnestus* spp.	2.00	0.02	H
			Pangaeus bilineatus	5.50	0.07	H

(Appendix H continued)

Order	Family	Subfamily	Species	% of samples	% of individuals	Functional Group
Hemiptera	Cydnidae		Immature	1.75	0.02	H
	Delphacidae		Unknown spp.	0.50	<0.01	H
	Enicocephalidae		*Systelloderes* spp.	1.25	0.02	P
	Geocoridae		*Geocoris* spp.	0.50	<0.01	P
	Issidae	Caliscelinae	*Fitchiella* spp.	1.50	0.02	H
	Largidae		*Arhaphe carolina*	1.25	0.01	H
	Lygaeidae	Orsillinae	*Xyonysius californicus*	10.25	0.97	H
	Lygaeidae		Immature	0.75	<0.01	H
	Membracidae	Smiliinae	*Acutalis tartarea*	0.25	<0.01	H
			Micrutalis parva	9.25	0.22	H
	Miridae	Byocorinae	*Sixeonotus albicornis*	0.25	<0.01	H
		Deraeocorinae	*Hyaliodes* spp.	0.25	<0.01	H
		Mirinae	*Megaloceroea* spp.	1.00	0.01	H
			Neurocolpus spp.	1.50	0.02	H
			Oncerometopus spp.	0.25	<0.01	H
			Plagiognathus spp.	0.75	<0.01	H
			Polymerus basalis	6.25	0.22	H
			Trigonotylus spp.	0.25	<0.01	H
			Unknown spp.	0.25	<0.01	H
			Immature	2.75	0.08	H
		Orthotylinae	*Lopidea major*	0.25	<0.01	H
		Phylinae	*Plagiognathus albatus*	0.50	<0.01	H
			Pseudatomoscelis seriatus	30.25	1.86	H
			Immature	5.75	0.09	H
	Nabidae		*Nabis* spp.	2.25	0.03	P
			Pagasa spp.	0.75	<0.01	P
			Immature	0.50	<0.01	P
	Pentatomidae	Asopinae	*Podisus maculiventris*	0.25	<0.01	P

(Appendix H continued)

Order	Family	Subfamily	Species	% of samples	% of individuals	Functional Group
Hemiptera	Pentatomidae	Pentatominae	*Euchistus tristihmus*	0.25	<0.01	H
			Mecidea minor	10.50	0.64	H
			Thyanta cusator	0.25	<0.01	H
			Trichopepla semivittata	10.75	0.06	H
			Immature	2.50	0.03	H
	Pseudococcidae		Unknown spp.	0.50	0.02	H
	Psyllidae		*Heteropsylla texana*	3.75	0.05	H
			Unknown spp.	2.00	0.02	H
	Reduviidae	Emesinae	*Emesaya* spp	0.25	<0.01	P
			Gardena elkinsi	0.25	<0.01	P
		Harpactorinae	*Apiomerus sissipes*	1.25	0.01	P
			Repipta taurus	2.75	0.04	P
			Sinea spp.	1.25	0.02	P
			Zelus luridus	3.50	0.05	P
			Zelus spp.	3.75	0.06	P
		Stenopodainae	*Oncocephalus* spp.	2.50	0.04	P
	Reduviidae		Immature	1.50	0.02	P
	Tingidae		*Alveotingis brevicornis*	0.50	<0.01	H
			Atheas spp.	0.25	<0.01	H
			Corythucha spp.	10.50	0.31	H
			Gargaphia iridescens	9.75	0.19	H
			Leptodictya plana	0.25	<0.01	H
			Teleonemia spp.	0.50	<0.01	H
	Rhopalidae	Rhopalinae	*Harmostes* spp.	2.00	0.02	H
			Immature	0.50	0.04	H
	Rhyparochromidae		*Pseudopamera setosa*	0.50	<0.01	H
			Immature	0.75	0.01	H

(Appendix H continued)

Order	Family	Subfamily	Species	% of samples	% of individuals	Functional Group
Hemiptera	Thyreocoridae		*Galgupha* spp.	1.75	0.02	H
Hymenoptera	Andrenidae	Andreninae	*Andrena* spp.	0.50	<0.01	Po
	Apidae	Apinae	*Apis mellifera*	0.25	<0.01	Po
			Bombus spp.	1.00	0.01	Po
		Xylocopinae	*Ceratina* spp.	1.75	0.03	Po
	Bethylidae	Bethylinae	Unknown spp.	1.50	0.02	P
		Epyrinae	*Anisepyris* spp.	0.25	<0.01	P
			Holepyris spp.	0.50	<0.01	P
			Laelius pedatus	0.25	<0.01	P
			Scelerodermus spp.	0.25	<0.01	P
		Pristocerinae	*Pristocera hyalina*	12.25	0.18	P
	Braconidae	Braconinae	Unknown spp.	2.25	0.03	P
	Ceraphronidae		Unknown spp.	0.50	<0.01	P
	Chalcididae	Chalcidinae	*Conura dema*	0.25	<0.01	P
			Conura igneoides	2.50	0.03	P
			Conura side	0.25	<0.01	P
			Phasgonophora spp.	0.25	<0.01	P
	Colletidae	Hylaeinae	*Hylaeus* spp.	0.75	<0.01	Po
	Cynipidae	Cynipinae	Unknown spp.	0.75	<0.01	P
	Encyrtidae		*Encyrtus* spp.	5.25	0.08	P
			Unknown spp.	11.00	0.17	P
	Eucharitidae		*Orasema occidentalis*	0.25	<0.01	P
	Eulophidae	Eulophinae	Unknown spp.	11.00	0.32	P
	Eupelmidae	Eupelminae	*Anastatus* spp.	1.50	0.02	P
			Arachnophaga spp.	1.75	0.02	P
			Eupelmus spp.	1.75	0.03	P
			Unknown spp.	2.50	0.03	P
	Eurytomidae		*Scyophilia* spp.	1.00	<0.01	P
	Figitidae	Eucoilinae	Unknown spp.	0.25	<0.01	P

(Appendix H continued)

Order	Family	Subfamily	Species	% of samples	% of individuals	Functional Group
Hymenoptera	Formicidae	Dolichoderinae	*Dorymyrmex bureni*	2.75	0.07	A
			Forelius mccooki	21.00	12.03	A
			Forelius pruinosus	47.75	1.57	A
			Liometopum spp.	0.25	<0.01	A
			Tapinoma sessile	9.25	0.16	A
		Ecitoninae	*Neivamyrmex pilosus*	0.50	<0.01	A
		Formicinae	*Camponotus* spp.	0.50	<0.01	A
			Formica spp.	0.25	<0.01	A
			Nylanderia terricola	22.50	0.34	A
			Nylanderia spp.	5.75	0.08	A
		Myrmicinae	*Carebara longii*	2.50	0.03	A
			Crematogaster spp.	1.00	0.01	A
			Cyphomyrmex wheeleri	0.50	<0.01	A
			Monomorium minimum	10.75	0.15	A
			Pheidole spp.	0.50	0.03	A
			Pogonomyrmex barbatus	5.25	0.47	A
			Solenopsis geminata	63.75	5.05	A
			Solenopsis invicta	45.50	2.85	A
			Stenamma spp	6.75	0.17	A
			Tetramorium bicarinatum	2.75	0.04	A
			Tetramorium spinosum	2.75	0.07	A
			Tetramorium spp.	3.25	0.06	A
		Ponerinae	*Hypoponera* spp.	0.25	<0.01	A
			Leptogenys elongata	6.00	0.08	A

(Appendix H continued)

Order	Family	Subfamily	Species	% of samples	% of individuals	Functional Group
Hymenoptera	Formicidae	Ponerinae	*Pachycondyla villosa*	0.25	<0.01	A
		Pseudomyrmicinae	*Pseudomyrmex pallidus*	0.50	<0.01	A
	Halictidae	Halictinae	*Augochloropsis metallica*	1.25	0.02	Po
			Lasioglossum texanum	1.00	0.01	Po
			Sphecodes spp.	2.00	0.03	Po
		Rophitinae	*Dufourea* spp.	0.25	<0.01	Po
	Ichneumonidae	Brachycrytinae	Unknown spp.	0.75	0.01	P
		Cryptinae	*Gelis* spp.	6.00	0.07	P
			Unknown spp.	1.50	0.02	P
	Mutillidae		*Dasymutilla* spp.	2.50	0.02	P
			Odontophotopsis spp.	2.00	0.02	P
			Pseudomethoca frigida	0.25	<0.01	P
	Mymaridae		Unknown spp.	13.25	0.18	P
	Platygastridae		*Platygaster* spp.	4.50	0.07	P
	Pompilidae	Pepsinae	*Auplopus* spp	2.50	0.03	P
			Pepsis spp.	0.25	<0.01	P
	Pteromalidae		Unknown spp.	11.75	0.19	P
	Scelionidae	Scelioninae	*Baeus* spp.	5.00	0.05	P
			Caliscelico spp.	0.25	<0.01	P
			Gryon spp.	0.75	<0.01	P
			Idris spp.	3.00	0.03	P
			Macroteleia spp.	2.75	0.09	P
			Scelio spp.	3.00	0.03	P
		Teleasinae	*Trimorus* spp.	6.25	0.08	P
		Telenominae	*Eumicrosoma* spp.	8.25	0.11	P
			Telenomus spp.	16.75	0.28	P

(Appendix H continued)

Order	Family	Subfamily	Species	% of samples	% of individuals	Functional Group
Hymenoptera	Scelionidae	Telenominae	*Trissolcus* spp.	17.50	0.27	P
			Tritoma spp.	0.25	<0.01	P
	Sphecidae	Crabroninae	*Gorytina* spp.	0.50	<0.01	P
		Philanthinae	*Philanthus* spp.	0.25	<0.01	P
	Tenthredinidae		Unknown spp.	1.00	0.02	P
	Torymidae		Unknown spp.	2.25	0.03	P
	Trichogrammatidae		Unknown spp.	15.75	0.23	P
	Vespidae	Eumeninae	*Stenodynerus* spp.	0.50	<0.01	P
		Polistinae	*Polistes exclamans*	0.25	<0.01	P
Isoptera	Rhinotermitidae		*Reticulitermes flavipes*	9.50	0.27	D
Lepidoptera	Acrolophidae		Unknown spp.	8.00	0.50	H
	Bucculatricidae		Unknown spp.	0.25	<0.01	H
	Elachistidae		Unknown spp.	0.75	<0.01	H
	Gelechiidae		Unknown spp.	0.75	<0.01	H
	Geometridae		Unknown spp.	0.75	<0.01	H
	Hesperiidae	Pyrginae	*Pyrgus communis*	3.75	0.04	Po
	Hesperiidae		Larvae	0.25	<0.01	H
	Lycaenidae	Lycaeninae	Lycaena spp.	0.50	<0.01	Po
			Larvae	0.25	<0.01	H
	Noctuidae	Noctuinae	*Spodoptera* spp.	0.50	<0.01	Po
			Larvae	1.00	0.01	H
	Nymphalidae	Nymphalinae	*Vanessa* spp.	0.25	<0.01	Po
	Tineidae		Unknown spp.	1.75	0.07	D
			Larvae	0.50	<0.01	D
	Tortricidae		Unknown spp.	0.25	<0.01	Po
Lepidoptera			Larvae	0.50	<0.01	H
Lithobiomorpha			Unknown spp.	0.25	<0.01	P
Mantodea	Mantidae		*Oligonicella scudderi*	0.25	<0.01	P

(Appendix H continued)

Order	Family	Subfamily	Species	% of samples	% of individuals	Functional Group
Mantodea	Mantidae		*Phyllovates chlorophaea*	0.25	<0.01	P
			Stamomantis carolina	0.25	<0.01	P
			Immature	0.25	<0.01	P
Mesostigmata	Parasitidae			5.25	0.10	P
Microcoryphia	Machilidae		Unknown spp.	1.75	0.03	D
Neuroptera	Chrysopidae	Chrysopinae	*Chrysopa* spp.	0.75	<0.01	P
			Larvae	0.50	<0.01	P
	Myrmeleontidae		*Brachynemurus sackeni*	0.75	<0.01	P
			Larvae	1.00	0.01	P
Odonata	Coenagrionidae		*Argia* spp.	0.25	<0.01	P
Oniscidea	Armadillidiidae		*Armadillidium vulgare*	96.00	13.51	D
	Porcellionidae		*Acareoplastes* spp.	1.00	0.01	D
Opilioacarida	Opiloacaridae		Unknown spp.	1.00	0.01	P
Opiliones	Cosmetidae		*Vonones* spp.	44.50	2.83	P
	Scelerosomatidae		Unknown spp.	0.25	<0.01	P
Orthoptera	Acrididae	Cyrtacanthacridinae	*Schistocerca* spp.	2.00	0.03	H
		Gomphocerinae	Unknown spp.	0.25	<0.01	H
		Melanoplinae	*Aeoloplides* spp.	3.25	0.06	H
			Melanoplus spp.	6.00	0.08	H
			Immature	1.50	0.02	H
		Oedipodinae	*Chortophaga viridifasciata*	2.25	0.04	H
	Gryllidae	Gryllinae	*Gryllus* spp.	42.50	0.77	D
		Myrmecophilinae	*Myrmecophilus* spp.	0.50	<0.01	D
	Mogoplistidae		*Cycloptilum* spp.	4.00	0.05	D
			Immature	5.25	0.05	D
	Rhapidophoridae		Unknown spp.	0.25	<0.01	D

(Appendix H continued)

Order	Family	Subfamily	Species	% of samples	% of individuals	Functional Group
Orthoptera	Tettigoniidae	Conocephalinae	*Conocephalus* spp.	7.25	0.10	H
		Phaneropterinae	*Scudderia* spp.	0.50	<0.01	H
		Tettigoniinae	Unknown spp.	0.50	<0.01	H
Phasmida	Heteronemiidae		*Parabacillus* spp.	7.75	0.10	H
Protura	Eosemtomidae		Unknown spp.	0.25	<0.01	D
Pseudoscorpionida	Neobisiidae		*Microbisium* spp.	0.25	<0.01	P
	Syarinidae		*Syarinus* spp.	0.25	<0.01	P
Psocoptera	Lachesillidae		*Lachesilla* spp.	6.25	0.10	D
	Liposcelidae		*Liposcelis* spp.	22.00	0.58	D
	Pachytroctidae		*Tapinella* spp.	13.00	0.20	D
	Pseudocaeciliidae		*Pseudocaecilius citricola*	0.25	<0.01	H
Sarcoptiformes	Acaridae			0.75	<0.01	D
	Euphthiracaridae			2.50	0.04	H
	Galumnidae			25.75	0.55	D
	Lohmanniidae			0.25	<0.01	D
	Mochlozetidae			35.25	3.64	H
	Nothridae			10.25	0.12	D
Sarcoptiformes	Unknown Family			5.25	0.09	
Scorpiones	Buthidae		*Centruroides vittatus*	46.25	0.98	P
Scutigeromorpha	Scutigeridae		*Dendrothereua homa*	6.00	0.08	P
			Scutiger coleoptera	29.00	0.56	P
Strepsiptera			Unknown spp.	0.25	<0.01	P
Thysanoptera	Aeolothripidae		*Aeolothrips* spp.	40.75	4.04	P
	Phlaeothripidae		*Haplothrips* spp.	42.25	1.61	P
	Thripidae		*Scolothrips* spp.	21.00	0.64	P
Thysanura	Lepismatidae		Unknown spp.	0.25	<0.01	D
Trichoptera	Limnephilidae		Unknown spp.	0.25	<0.01	H
Trombidiformes	Anystidae			92.25	8.01	P

(Appendix H continued)

Order	Family	Subfamily	Species	% of samples	% of individuals	Functional Group
Trombidiformes	Bdellidae			10.25	0.16	P
	Erythraeidae			41.50	1.35	P
	Parantennullidae			0.25	0.02	P
	Smarididae			11.00	0.21	P
	Stigmaeidae			4.75	0.07	P
	Tetryanchidae			4.50	0.12	H
Trombidiformes	Unknown Family			0.75	0.01	