The Texas Journal of Agriculture and Natural Resources 37:13-24

Arthropod Response to Fire in a Chihuahuan Desert Grassland in Texas

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ABSTRACT

A Chihuahuan Desert grassland winter prescribed burn was studied for its effect on first-year arthropod abundance, biomass, composition, and recovery at the Elephant Mountain Wildlife Management Area, Brewster County, Texas. The burn was conducted in early February 1997, resulting in a cool, discontinuous fire. Arthropods were collected pre- and postburn from January-December on unburned control and burned treatment sites with suction collection. Insects were identified to the familial level, pooled at the ordinal level; counted, dried, and weighed, whereas arachnids were grouped, counted, dried, and weighed. Preburn vegetation was comparable between treatments, and no differences were found for insect and arachnid abundance and biomass ($P \leq 0.05$). Postburn, herbaceous cover was reduced significantly, and abundance differences were found in arachnids and for nine of 15 insect orders ($P \leq 0.05$). Biomass at the ordinal level differed for the Hymenoptera, Orthoptera, and Psocodea ($P \leq 0.05$). The fire did not strongly impact diversity and evenness of arthropod populations although composition differed. Postburn arthropod abundance and biomass tracked vegetation changes over the ensuing year and were highest at peak growing season following monsoonal rains. Arthropod abundance, biomass, and diversity between treatments were comparable at the end of the first postfire year, and trends indicated that population abundance was approaching pre-burn levels, although composition remained different.

KEY WORDS: arthropods, arachnids, Chihuahuan Desert, fire, grassland, insects, Trans-Pecos, Texas

INTRODUCTION

Desert grasslands of southeastern Arizona, southern New Mexico, and southwestern Texas are the most arid North American grasslands (McClaran 1995). During European exploration in the 1500's, they were dominated by grass with relatively few shrubs, low growing trees, and succulents. Since 1870, shrubs have gained dominance with a concomitant decrease in herbaceous cover (Humphrey 1958, Bahre 1991, Turner et al. 2003). Overgrazing and fire suppression, among others, are cited as primary reasons for these changes (Humphrey 1958, Buffington and Herbel 1965, Biswell 1989, Turner et al. 2003).

Fire has continuously been an integral part of North America ecosystems (Wright and Bailey 1982, Collins and Wallace 1990). Native Americans traditionally used fire to improve forage for grazing animals and to clear brush and undergrowth (Biswell 1989). From the early to mid-1900s, fire was discouraged as a management tool, while suppression was encouraged because fire was thought to be destructive. Fire further posed a risk to property and animals (Komarek 1970, Biswell 1989). More recently prescribed fire has been used as a multifaceted management tool to clear crop residues, improve forage quality, suppress shrubs, conserve biodiversity, facilitate heterogeneity, and help maintain grassland and other ecosystems (Wright and Bailey 1982, Scifres and Hamilton 1993, Fuhlendorf and Engle 2001, Calcaterra et al. 2014). Fire is further recognized as a major driver in restoring and maintaining rangeland heterogeneity and ecosystem diversity and desirable ecosystem attributes (Fuhlendorf and Engle 2001, Fuhlendorf et al. 2006). This philosophical shift is related to economics and environmental acceptability (Welch 1982, Wright and Bailey 1982, Scifres and Hamilton 1993, Fuhlendorf and Engle 2001). In Texas, prescribed burning has been used as an effective management tool for rangelands (Scifres and Hamilton 1993), however, little has been documented on the effects of fire in the Trans-Pecos region of southwestern Texas (Schnapp and Kinucan 2010, Warnock 2010).

Fire effects on ecosystem processes and invertebrate abundance and diversity are poorly understood in subtropical savanna and grassland biomes yet are frequently used as a tool to support biodiversity conservation and management (Parmenter et al. 2011, Mukwevho et al. 2024). Arthropods are critical pollinators, food web elements, and speed nutrient cycling (Seastedt and Crossley 1984). However, comparatively few studies have investigated fire effects on soil-surface and plant insects (Warren et al. 1987, McPherson 1995, Parmenter et al. 2011), although studies are increasingly pervasive (e.g., Yekwayo et al. 2018, Butler et al. 2021, Certini et al. 2021, Mukwevho et al. 2024). Within the Texas Trans-Pecos, no arthropod fire response studies have been published.

This descriptive study documents the short-term effects of a winter prescribed burn on arthropod abundance and composition in a subtropical desert grassland at the Elephant Mountain Wildlife Management Area (EMWMA), Brewster County, Texas. Our objectives were to: (1) describe short-term impacts on vegetation and arthropod assemblages, and (2) compare pre- and post-burn arthropod abundance, biomass, composition, and diversity.

MATERIALS AND METHODS

Study Area. Our study occurred at the EMWMA, 41 km south of Alpine, Texas (30°02'42"N, 103°29'27"W). The site consisted of Igneous Hill and Mountain, Mixed Prairie and Gravelly, and Mixed Prairie Ecological Sites (EDIT 2024). The prevalent plant community at the time of the study was rose-fruited juniper (*Juniperus erythrocarpa*) – mixed grass, typical of the mixed growth forms found in southwestern desert grasslands (Burgess 1995) and is essentially the same at present. Dominant herbaceous vegetation included black grama (*Bouteloua eriopoda*), big sacaton (*Sporobolus wrightii*), blue grama (*Bouteloua gracilis*), sideoats grama (*Bouteloua curtipendula*), burrograss (*Scleropogon brevifolius*), bush muhly (*Muhlenbergia porteri*), pepperweed (*Lepidium virginicum*), plains bristlegrass (*Seteria leucopila*), Russian thistle (*Salsola iberica*), and vine mesquite (*Panicum obtusum*). Shrubs included rose-fruited juniper, honey mesquite (*Prosopis glandulosa*), white brush (*Aloysia gratissima*), four-wing saltbush (*Atriplex canescens*), catclaw acacia (*Acacia greggii*), and catclaw mimosa (*Mimosa biuncifera*). Taxonomy follows Hatch et al. (1990). Live junipers and shrubs ranged from 0.5 to 2 m tall.

The climate was characteristic of Chihuahuan Desert grasslands with mild winters and hot summers. Annual precipitation ranged from 229-594 mm; the majority occurred as monsoonal high intensity summer thunderstorms (Figure 1). Elevation ranged from 1,303 to 1,333 m (Soil Survey Staff 2024). Cattle, with occasional goat and sheep use, grazed the area since the late 1800s (Vaughan 1961). More recently only cattle were grazed, and they were removed from the EMWMA in August 1996 (C. Brewer, Texas Parks and Wildlife Department, pers. comm.).

Experimental design. Four unburned control sites and four corresponding burned treatment sites were randomly established in December 1996 within a portion of the EMWMA planned for a Texas Parks and Wildlife Department prescribed burn. For each, a representative 50 m baseline was subjectively placed, and three 50 m transects permanently located at random points perpendicular to the baseline to sample vegetation and arthropods.

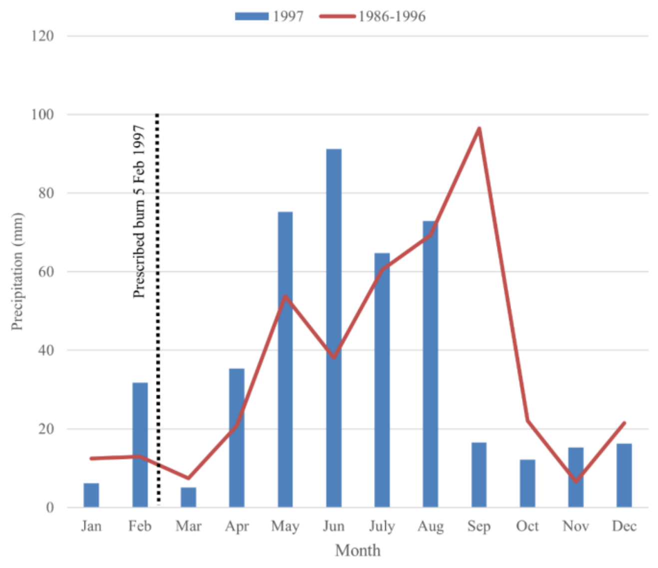

Figure 1. Mean monthly rainfall from 1986-1996 and monthly rainfall for 1997 at Elephant Mountain Wildlife Management Area (EMWMA), Brewster County, Texas. Data provided by EMWMA. The vertical dashed line represents the date of the prescribed burn.

A prescribed burn was conducted on 5 February 1997 at 0900 using a headfire ignited with drip torches. Wind speed and direction was constant throughout the burn at 9 km/h from the northeast. Mean air temperature remained 11º C during the burn. At the start of the burn, relative humidity was 41% and dropped to 32% over the first three hours. By

1200, humidity increased, and by 1500 the burn was stopped because high humidity made ignition difficult. Flame heights were estimated at 0.5 - 3 m and approximately 65% of the treatment burned. This resulted in a mosaic of burned and unburned patches, typical of fires under these conditions in desert grassland habitats.

Vegetation was stratified by growth form (herbaceous and shrub) and sampled monthly during 1997. Three 25 cm x 50 cm quadrats were randomly placed along each transect to estimate herbaceous basal cover and density, as well as bare ground and litter cover. In the same manner three 2 m x 2 m quadrats were placed to sample shrub density. Shrub foliar cover was estimated by the line intercept method along each 50 m transect.

At each site, one transect was randomly selected for arthropod sampling. Surface dwelling arthropods were collected with a D-vac Suction Machine with a 45.7 cm collector orifice on 27 sampling days starting 24 January 1997 through 29 December 1997 at roughly two-week intervals. Two preburn sample periods were included. The D-vac orifice was held 15 cm above the ground (Hurst 1972, Pavon 1995). When large or thorny shrubs were encountered, the sampler would walk around the perimeter of the shrub until the transect was intercepted. The cumulative area sampled throughout the study for the control and treatment sites was 1,782 m^2.

All samples were immediately transferred from the collection net into a 3.8 l plastic bag and placed in a cooler with ice blocks. Within four hours of collection, samples were placed in a freezer until they could be sorted and processed. Insects were identified, generally to the family level. Because counts within family were often small, insects were pooled by order. Collections were dried at 70º C for 48 hours (Bulan and Barrett 1971), counted and weighed on a Metler four-place digital balance. Arachnids were combined, dried, counted, and weighed. Taxonomy follows Borror et al. (1989).

Statistical analyses. For all data, randomized block ANOVAs, with months as the blocked factor, were used to assess differences between control and treatment sites. Because of limited numbers, arthropods were pooled by order/suborder for analysis. To address departures from normality, arthropod counts were transformed ($\sqrt{x}+0.5$) and weights were transformed [$\log_{10}(x+1)$]. For vegetation, densities were transformed ($\sqrt{x}+0.5$), and quadrat and line intercept cover values were transformed (arcsine $\sqrt{x}$). All statistics were calculated using SPSS PC+ V 7.5.2 (SPSS Base Manual 1997). A Shannon-Wiener diversity index (H'), maximum diversity (H'max), and evenness (J') were calculated within and between sites. Diversity differences were assessed using a Hutcheson t-test (Zar 1996). Nutritional values were estimated with conversion factors from Bell (1990).

RESULTS

Preburn. Bare ground (69%) had the highest relative cover for all treatment sites, followed by litter (21%), grass (8%), forb (2%), and shrub (<1%). There was no herbaceous treatment site difference between basal cover, foliar cover, and density, however shrub cover as assessed by line intercept was greater in the control (F=4.12; *d.f.*=145; $P \leq 0.05$). Growth form diversity and evenness were 1.5% greater in the control site ($P \leq 0.05$) (Table 1).

Table 1. Shannon-Wiener diversity (*H'*), maximum diversity (*H'*max), and evenness (*J'*) indices for vegetation growth forms before and after the prescribed burn for control (unburned) and treatment (burned) sites.

	Preburn			Postburn		
	H'	*H'*max	*J'*	*H'*	*H'*max	*J'*
Control	0.4704^a (n=161)	0.4771	0.9860	0.4661^a (n=877)	0.4771	0.9769
Treatment	0.4636^a (n=159)	0.4771	0.9717	0.4547^b (n=799)	0.4771	0.9530

Values with different superscripts within a burn period (preburn vs postburn) are different at $P \leq 0.05$.

The cumulative number of arthropods collected preburn was 1,422 ($\bar{x}$ = 733/sample interval) with 53% from the treatment site (Figure 2). Nine insect orders were represented, and three were unique to the treatment site (Table 2). No differences were detected between preburn treatments for insect orders and arachnid numbers ($P \leq 0.05$). The relative abundance of groups in preburn (control-treatment) sites were Homoptera (39-54%), Hemiptera (35-24%), Diptera (13-7%), Arachnids (8-9%), Hymenoptera (4-3%), and others (2-4%). The treatment site had 10% greater diversity and evenness than the control site ($P \leq 0.05$).

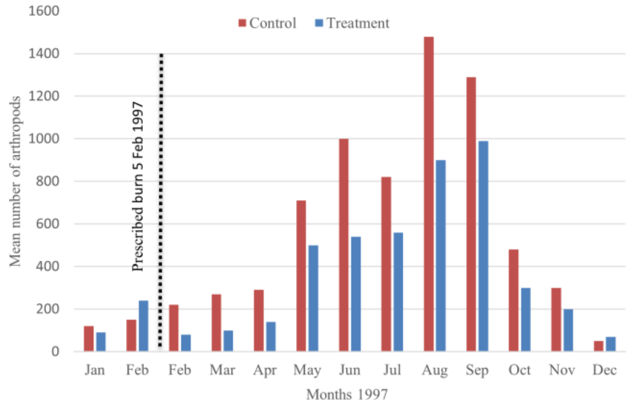

Figure 2. Mean number of arthropods collected by month starting in Jan 1997 through Dec 1997 for unburned (*n*=108) and burned (*n*=108) treatments. The vertical dashed line represents the date of the prescribed burn.

Table 2. Presence of insect orders collected before (Preburn) and after (Postburn) a winter prescribed fire for Control (unburned) and Treatment (burned) sites at Elephant Mountain Wildlife Management Area, Brewster County, Texas.

	Preburn		Postburn	
Order	Control	Treatment	Control	Treatment
Coleoptera	X	X	X	X
Collembola			X	X
Dictyoptera			X	X
Diptera	X	X	X	X
Hemiptera	X	X	X	X
Homoptera	X	X	X	X
Hymenoptera	X	X	X	X
Isoptera			X	X
Lepidoptera		X	X	X
Microcoryphia			X	X
Neuroptera		X	X	X
Orthoptera		X	X	X
Phasmida			X	X
Psocoptera			X	X
Thysanoptera	X	X	X	X

The cumulative insect and arachnid biomass was 0.4375 g with 50% from the control and treatment sites (Figure 3). Homopterans contributed the most biomass at both sites followed by Hemipterans and Dipterans. No differences ($P \leq 0.05$) were found for insect order and arachnid biomass between preburn treatments. Gross available estimated energy from insects was similar for both sites (152.12 KJ/ha for the control site and 158.49 KJ/ha for the treatment site).

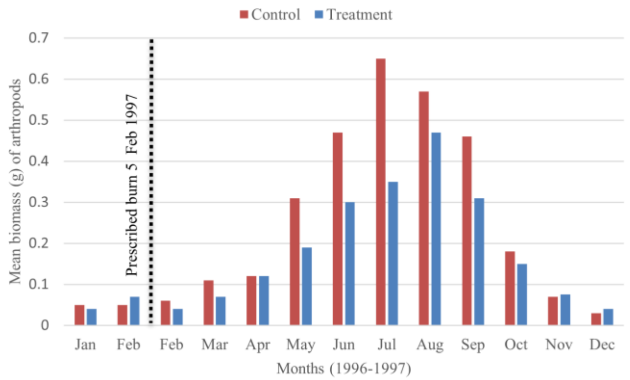

Figure 3. Mean biomass (g) of arthropods collected by month starting in Jan 1997 through Dec 1997 for unburned (*n*=108) and burned (*n*=108) treatments. The vertical dashed line is the time of the prescribed burn.

Postburn. The unburned control maintained relative vegetation cover values comparable to preburn conditions, with bare ground dominating (67%), followed by litter (23%), shrub (7%), grass (2%), and forb (<1%). In the burned treatment, relative cover for bare ground increased (78%), whereas litter (13%), shrub (6%), grass (1.8%), and forb (<1%) decreased, and shrub cover as assessed by the line intercept was lower in the burned treatment ($F=81.01$, $d.f.=1,252$; $P \leq 0.05$). Likewise, grass, forb and shrub densities also decreased in the burned treatment ($P \leq 0.05$). Growth form diversity and evenness were 2.5% greater in the control site ($P \leq 0.05$) (Table 1).

Postburn, 22,372 ($\bar{x} = 895$/sample interval) arthropods were collected with 61% from the unburned control. The largest number of arthropods were collected in August and September (Figure 2), which corresponded with the monsoon season (Figure 1). Fifteen insect orders were represented in both the control and treatment sites (Table 2). Homoptera (32-40%), Hemiptera (11-15%), Diptera (16-14%), and Hymenoptera (10-8%) continued to be the most abundant orders in the control and treatment sites, although others (32-17%) were proportionally greater, perhaps signaling increased richness in less dominant taxa as the growing season progressed. Total counts and biomass for all classes were markedly lower in the burned area until November and December 1997 (Figures 2 and 3).

Within most insect orders, multiple families were represented and each had two or three families that constituted most abundance within the order. Forty-four percent of the families collected were phytophagous (Table 3). Significant abundance differences between treatments occurred for arachnids, Coleoptera, Collembola, Diptera, Hymenoptera, Lepidoptera, Orthoptera, Phasmida, Psocoptera, and Thysanoptera. The orders Hymenoptera, Hemiptera, and Diptera remained the most abundant on the control and treatment sites. The control site had 2% greater diversity and evenness; however, both the control and treatment sites increased in diversity and evenness from winter through the growing season (Table 4).

Table 4. Shannon-Wiener diversity (*H*'), maximum diversity (*H*'max), and evenness (*J*') indices for insect orders and arachnids before and after the prescribed burn for control (unburned) and treatment (burned) sites.

	Preburn			Postburn		
	H'	*H*'max	*J*'	*H*'	*H*'max	*J*'
Control	0.7822^{a} (*n*=79)	1.2041	0.6496	1.0927^{a} (*n*=745)	1.2041	0.9075
Treatment	0.8606^{b} (*n*=78)	1.2041	0.7147	1.0702^{a} (*n*=647)	1.2041	0.8888

Values with different superscripts within a burn (preburn vs postburn) period are different at $P \leq 0.05$.

Cumulative insect and arachnid biomass was 9.8932 g, with 60% in the unburned control site (Figure 3). Differences ($P \leq 0.05$) were detected among the Hymenoptera, Orthoptera, and Psocodea. The burned treatment site had 32.7% less gross available energy (618.01 KJ/ha) than the unburned control site (918.66 KJ/ha).

CONCLUSION AND DISCUSSION

Arthropod counts for burned and unburned sites followed a seasonal trend, and increased from May through August, then decreased September through December. Herbaceous vegetation followed a similar growth pattern which corresponded with monsoonal growing season precipitation (Figure 1). Family Cicadellidae, 69% of the order Homoptera, were the most abundant insects throughout the study. Homoptera represented the highest proportion of insects preburn, as well as during postburn recovery, a tendency observed by others (e.g., Cancelado and Yonke 1970, Pavon 1995) reflecting their mobility and the proximity of unburned refugia within the treatment area. As a proportion of arthropods collected, the following were the most abundant: Arachnids (10% control, 8% burn), Diptera (16% control, 14% burn), Hemiptera (11% control, 13% burn), Homoptera (32% control, 40% burn), Hymenoptera (9% control, 8% burn), and all others combined (22% control, 17% burn). Relative arthropod biomass exhibited a similar pattern: Arachnids (6% control, 7% burn), Diptera (7% control, 11% burn), Hemiptera (15% control, 19% burn), Homoptera (26% control, 39% burn), Hymenoptera (7% control, 7% burn) and all others combined (40% control, 20% burn).

Most insects collected were plant feeders or omnivores [i.e., phytophagous (44%), omnivorous (21%), predaceous (16%), parasitic (11%), and scavenger (7%)]. The lower number of arthropods on the burned site during the growing season was likely from decreased food resources and altered habitat structure caused by the fire. Warren et al. (1987) stated fire can have an acute impact with an extended shock phase on arthropods until vegetation regrowth occurs. Shock phase length varied depending on time to regrowth, which was impacted by season of burn and subsequent weather and growing conditions. They stated burn impacts can be chronic until plant and animal population equilibrium returns.

Immediately following the prescribed burn, arthropod abundance increased in the unburned control but declined in the burned treatment. By May, four months following the fire, a noticeable increase in arthropod abundance occurred in both the control and treatment sites, although numbers and biomass remained significantly lower in burned areas. Abundance peaked in August, corresponding with plant growth and monsoonal rains. By October, total arthropod abundance decreased, however proportionally less so in the burned site (Figures 2 and 3). By December, total arthropod numbers were comparable between treatments. These dynamics appeared to follow a seasonal trend driven largely by temperature and rainfall (Cloudsley-Thompson 1991, Gilbert and Raworth 1996, Vasconcelos et al. 2009). Certini et al. (2021) noted that prescribed burns often cause less impact than wildfires and were generally less harmful for ground dwelling arthropods. Enhanced arthropod population growth on burned sites may relate to reduced resource competition and improved forage quality. Arthropod recovery in burned southwestern desert grasslands approached those of unburned sites within several growing seasons as vegetation recovered and arthropods reoccupied burned sites (e.g., Bock and Bock 1991, Parmenter et al. 2011, Day et al. 2019) as well as in South African grasslands (Haddad et al. 2015, Mukwevho et al. 2024). Given the arthropod recovery observed in burned areas of this study it is not unreasonable to anticipate comparable recovery in this subtropical desert grassland. Reed (1997) discussed four factors that influence effects a fire can have on an arthropod species: (1) burn tolerance or sensitivity during the season when the burn occurs, (2) postburn population dynamics, (3) colonization potential, and (4) response to all phases of the postburn succession. These factors were undoubtedly influential for this area, but our study resolution did not permit us to fully quantify each of them.

Season of burn can impact arthropods in multiple ways. Our winter burn likely minimized direct impacts on arthropods because they were less active and perhaps sheltered in protected areas (Parmenter et al. 2011). Spring burns in a Kansas tallgrass prairie had minimum impacts on grasshoppers because most existed as underground eggs at the time of burn (Evans 1984, 1988). In contrast, an intense summer fire in Arizona grassland drastically reduced the flightless grasshopper *Dactylotum variegatum* as evidenced by charred corpses and its disappearance for multiple years post-burn (Bock and Bock 1991). Following a June fire in New Mexico desert grassland, ground dwelling herbivore activity-density generally decreased, although predator species often increased (Parmenter et al. 2011). Crickets were essentially eliminated from the burned areas but quickly recolonized. Acridid grasshoppers, carabid beetles, tenebrionid beetles, centipedes, scorpions, and sun spiders demonstrated significant postfire activity-density increases. These groups apparently found fire refugia in the soil, under rocks, or in rodent burrows and didn't suffer immediate mortality. They also likely dispersed from adjacent unburned patches to recolonize burned areas. Parmenter et al. (2011) observed fire effects were most profound on grass-dependent species.

Table 3. Insect orders and families collected at Elephant Mountain Wildlife Management Area, Brewster County, Texas, January–December 1997. Nomenclature follows Borror et al. (1989). Families and species of Thripidae identified by Dr. Charles Cole, Texas AgriLife Extension, College Station, Texas.

Coleoptera
- Bostrichidae (Branch & Twig Borers)
- Bruchidae (Seed Beetles)
- Cantharidae (Soldier Beetles)
- Cerambycidae (Longhorn Beetles)
- Chrysomelidae (Leaf Beetles)
- Cleridae (Checkered Beetles)
- Coccinellidae (Ladybird Beetles)
- Curculionidae (Weevils or Snout Beetles)
- Melandryidae (False Darkling Beetles)
- Melyridae (Soft-winged Flower Beetles)
- Mordellidae (Tumbling Flower Beetles)
- Oedemeridae (False Blister Beetles)
- Staphylinidae (Rove Beetles)

Collembola
- Entomobryidae (Elongated Springtails)
- Sminthuridae (Springtails)

Dictyoptera
- Mantidae (Mantids)

Diptera
- Agromyzidae (Leaf-miner Flies)
- Anthomyiidae
- Asilidae (Robber Flies)
- Bombyliidae (Bee Flies)
- Calliphoridae (Blow Flies)
- Cecidomyiidae (Gall Midges)
- Ceratopogonidae (Biting Midges)
- Chamaeyiidae (Aphid Flies)
- Chironomidae (Midges)
- Chloropidae (Grass Flies)
- Culicidae (Mosquitoes)
- Dolichopodidae (Longlegged Flies)
- Empididae (Dance Flies)
- Helomyzidae
- Lauxaniidae
- Muscidae (House Flies)
- Otitidae (Picture-winged Flies)
- Phoridae (Humpbacked Flies)
- Pipunculidae (Bigheaded Flies)
- Rhagionidae (Snipe Flies)
- Sarcophagidae (Flesh Flies)
- Scenopinidae (Window Flies)
- Sciomyzidae (Marsh Flies)
- Sepsidae (Black Scavenger Flies)
- Syrphidae (Flower Flies)
- Tachinidae
- Therevidae (Stiletto Flies)

Hemiptera
- Alydidae (Broad-headed Bugs)
- Anthocoridae (Minute Pirate Bugs)
- Berytidae (Stilt Bugs)
- Coreidae (Leaffooted Bugs)
- Lygaeidae (Seed Bugs)
- Miridae (Plant Bugs)
- Nabidae (Damsel Bugs)
- Pentatomidae (Stink Bugs)
- Reduviidae (Assassin Bugs)
- Rhopalidae (Scentless Plant Bugs)
- Thyreocoridae (Negro Bugs)
- Tingidae (Lace Bugs)

Homoptera
- Acanaloniidae
- Aphididae
- Aleyrodidae (Whiteflies)
- Cicadellidae (Leaf Hoppers)
- Cixiidae (Cixiid Planthoppers)
- Delphacidae (Delphacid Planthoppers)
- Dictyopharidae
- Eriosomatidae (Woolly and gall making aphids)
- Flatidae (Flatid Planthoppers)
- Issidae (Issid Planthoppers)
- Membracidae (Treehoppers)
- Ortheziidae (Ensign Scales)
- Pseudococcidae (Mealybugs)
- Psyllidae (Jumping Plant Lice)

Hymenoptera
- Andrenidae (Andrenid Bees)
- Apidae (Bumble Bees)
- Argidae
- Braconidae
- Ceraphronidae
- Chalcididae
- Chrysididae (Cuckoo Wasps)
- Diapriidae
- Dryinidae
- Embolemidae
- Encyrtidae
- Eucharitidae
- Eulophidae
- Eupelmidae
- Eurytomidae (Seed Chalcidoids)
- Formicidae (Ants)
- Halictidae (Halictidae Bees)
- Ichneumonidae
- Megachilidae (Leafcutting Bees)
- Mymaridae (Fairyflies)
- Pteromalidae
- Sphecidae
- Tiphiidae
- Torymidae

Isoptera
- Termitidae (Termites)

Lepidoptera
- Arctiidae (Tiger Moths)
- Cosmopterigidae (Leaf Miners)
- Lasiocampidae (Tent Caterpillars)
- Lycaenidae (Blues, Coppers, and Hairstreaks)
- Lyonetiidae
- Noctuidae (Noctuid Moths)
- Pyralidae (Snout and Grass Moths)

Microcoryphia
- Machilidae (Bristletails)

Neuroptera
- Chrysopidae (Green Lacewings)
- Coniopterygidae (Dusty Wings)
- Hemerobiidae (Brown Lacewings)
- Myrmeleontidae (Antlions)

Orthoptera
- Acrididae (Slantfaced Grasshoppers)
- Gryllidae (Crickets)
- Tettigoniidae (Katydids)

Phasmida
- Phasmatidae (Walking Sticks)

Psocoptera
- Pseudocaeciliidae (Barklice)

Thysanoptera*
- Aeolothripidae (Broad-winged/banded thrips)
  - *Stomatothrips flavus*
- Phlaeothripidae
  - *Oedaleothrips baileyi*
- Thripidae (Common Thrips)
  - *Bregmatothrips sonorensis*
  - *Chirothrips falsus*
  - *Chirothrips mexicanus*
  - *Chirothrips simplex*
  - *Chirothrips texanus*
  - *Frankliniella tritici*
  - *Plesiothrips avarsi*

This appeared to occur in our winter burn where arthropod populations were initially lower postburn but increased abundance throughout the growing season and into the winter. The mild burn intensity, patchy burn pattern, and season likely minimized direct mortality at the site level. It reduced overall herbaceous vegetation cover until regrowth occurred, which decreased habitat and forage for herbivores and omnivores dependent on grasses and forbs. However, the mosaic of burned and unburned vegetation provided refugia from which rapid colonization of burned patches could emanate as the summer growing season progressed. Although a July grassland wildfire in Arizona had moderate short-term impacts on ground dwelling arthropod populations, recolonization and recruitment were generally rapid (Bock and Bock 1991). They concluded herbivores tracked postfire vegetation changes, whereas predator species responded to altered habitat structure and prey availability. Parmenter et al. (2011) concluded that arthropod response patterns in a Chihuahuan Desert grassland were comparable to those found in many studies from more mesic environments, with generally rapid postfire recovery. Andersen and Muller (2000) noted a limited fire impact for most ordinal-level taxa in an Australian tropical savannah, indicating considerable resilience of arthropod assemblages to early and late season fire regimes.

In Mississippi, Hurst (1972) attributed increased insect abundance following fire to the lush, succulent vegetative growth after a burn, an assertion supported by Warren et al. (1987). Bock and Bock (1991) found that *Melanoplus gladstoni* and *Trimerotropis pallidepennis* grasshoppers, which preferred habitats with bare ground and herbaceous foods, temporarily increased in abundance following a summer burn in an Arizona grassland. Although overall nymph and adult grasshopper abundance declined 60% on burned sites in the first year following the fire, no significant differences remained by the second year. They concluded that grasshoppers in this environment are tolerant of but not dependent on fire. Decreased ant abundance followed fire in northeastern Argentina grasslands, but was not detectible six months later, although effects on species richness and composition were unclear (Calcaterra et al. 2014)

In the short-term, vegetation and arthropod diversity and evenness decreased slightly in the burned treatment, but this was not likely ecologically meaningful in the long-term. Periodic fire often enhances biodiversity (Kim and Holt 2012, Yekwayo et al. 2018, Mukwevho et al. 2024), including community spatial and temporal heterogeneity (Fuhlendorf and Engle 2001). Burning as a disturbance is regularly used as a management tool to enhance biodiversity and improve habitat. Evans (1984) found the greatest grasshopper diversity in tallgrass prairies subjected to intermediate fire frequencies. In Mojave and Great Basin deserts, Day et al. (2019) noted that fire had little effect on invertebrate abundance, except for fewer flying foragers in Great Basin burned areas, ascribed to avoidance of burned habitats. They did observe reduced ant species richness and diversity, attributed to indirect changes in plant community composition through fire-mediated invasive species increases. Arthropod impacts can be mixed depending on response of individual species as well as plant community dynamics, as noted by Reed (1997) and others (e.g., Evans 1984, Bock and Bock 1991, Day et al. 2019), but in general biodiversity recovers quickly (Yekwayo et al. 2018) and is often enhanced (Butler et al. 2020, Mukwevho et al. 2024).

Total available energy and abundance of arthropods decreased following the burn, although it appeared to match the unburned treatment as the subsequent year progressed. Our estimation of energy from arthropods would have been more meaningful had it been based on taxa-specific values rather than ordinal-level estimations. Despite this shortcoming, our estimates provided an approximation of system-level caloric loss in the aftermath of a burn with a concomitant recovery during the following growth season. There is little doubt that insects are an important food resource for many species of amphibians, reptiles, birds, and mammals (Porter 1972, Wetzel 1983, Payne and Bryant 1994). They provide protein, energy, and water. Peters (1988) stressed their importance as a protein resource, whereas Bell (1990) emphasized their importance as an energy resource, and Cloudsley-Thompson (1991) noted their importance as a water resource in desert environments.

Cloudsley-Thompson (1991) reported that seasonal fluctuations in Chihuahuan Desert lizard communities were related directly to the relative abundance of arthropods, which was linked to rainfall. Because reptiles and amphibians tend to burrow in the ground or seek shelter under rocks during winter months, prescribed burns during the winter or early spring would have the least direct impact on their populations. However, if insect numbers did not recover for one or more years following a fire, herpetofauna could be indirectly affected. Arthropods, notably insects, provide sustenance for a greater number and variety of birds than any other plant or animal food (Brook and Birkhead 1991). In New York (Euler and Thompson 1978) and Scotland (Watson 1989), several species of insectivorous birds foraged more on burned areas than unburned areas which was attributed to better insect visibility. We did not see a fire impact on shrub structure, but when managing rangeland habitat for songbirds, Renwald (1978) noted care must be taken to preserve adequate numbers of lotebush (*Ziziphus obtusifolia*) and older mesquite trees for nesting. Winter and spring burns should have little impact on nesting or juvenile birds, and this low-intensity fire was likely favorable for insectivorous birds.

Insects are critical as a food resource and pollinators and are undoubtedly affected by fires. Prescribed burns can be used to enhance biodiversity by creating a spatial and temporal mosaic of habitats (Kim and Holt 2012, Fuhlendorf and Engle 2001, Fuhlendorf et al. 2006). This is particularly true given the irregular distribution of vegetation and patchy nature of fires in desert grasslands. The paucity of fire studies, specifically those investigating the impacts on arthropods, limits our understanding of ecosystem dynamics in Trans-Pecos desert grasslands. This study was brief in duration, affected only a single season burn, lacked different fire frequencies and intensities, and focused on ordinal-level groups, factors that limited broad interpretation. Although conducted in 1997, the lack of subsequent studies in this region makes it relevant to current and future studies of arthropod fire response in the Trans-Pecos. To fully address Reed's (1997) hypothesized factors of fire impacts on arthropods, research should address population dynamics under different fire conditions, as well as study genus and species levels of arthropods to better understand trophic-level importance and host-specific relationships (Kim and Holt 2012, Yekwayo et al. 2018). This assertion, and the difficulty identifying these relationships, was highlighted by Richerson and Boldt (1995) in Trans-Pecos Texas and Arizona. In their study they were unable to determine the host-plant associations for 41% of the insects they collected on tarbush (*Flourensia cernua*), a key floristic component of Chihuahuan Desert vegetation.

ACKNOWLEDGMENTS

We thank the Texas Parks and Wildlife Department for access to the research site and their prescribed burn. C. Cole generously assisted with thrip identification, and C. Ritzi graciously contributed insight into arthropod classification as well as discerning suggestions. Two anonymous reviewers provided constructive comments that strengthened the manuscript. We are grateful to K. Kinucan for a keen editorial eye to ensure clarity and readability.

REFERENCES

Andersen, A. N. and W.J. Muller. 2000. Arthropod responses to experimental fire regimes in an Australian tropical savannah: ordinal-level analysis. Austral Ecology. *25*:199-209.

Bahre, C. J. 1991. A legacy of change: human impacts on vegetation in the Arizona borderlands. University of Arizona Press, Tucson, AZ.

Bell, G. P. 1990. Birds and mammals on an insect diet: a primer on diet composition analysis in relation to ecological energetics. Studies in Avian Biology. 13:416-422.

Biswell, H. H. 1989. Prescribed burning in California wildlands vegetation management. University of California Press, Los Angeles, California.

Bock, C. E. and J. H. Bock. 1991. Response of grasshoppers (Orthoptera: Acrididae) to wildfire in a southeastern Arizona grassland. The American Midland Naturalist. 125:162-167.

Borror, D.J., C. A. Triplehorn, and N. F. Johnson. 1989. An introduction to the study of insects. Sixth edition. Saunders College Publishing, New York.

Brook, M. and T. Birkhead. 1991 (eds.). The Cambridge encyclopedia of ornithology. Cambridge University Press, Cambridge.

Buffington, L. C. and C. H. Herbel. 1965. Vegetation changes on a semidesert grassland range from 1858 to 1963. Ecological Monographs. 35:139-164.

Bulan, C. A. and B. W. Barrett. 1971. The effects of two acute stresses on the arthropod component of an experimental grassland ecosystem. Ecology 52:597-605.

Burgess, T. L. 1995. Desert grassland, mixed shrub savanna, shrub steppe, or semidesert shrub? The dilemma of coexisting growth forms. Pages 31-67, *in* M. P. McClaran and T. R. van Devender (eds.). The Desert Grassland. University of Arizona Press, Tucson, AZ.

Butler, A., C. A. Davis, S. D. Fuhlendorf, and S.M. Wilder. 2021. Effects of fire on ground-dwelling arthropods in a shrub-dominated grassland. Ecology and Evolution. 11:427-442.

Calcaterra, L. A., Y. Di Blanco, M. Srur, and J. Briano. 2014. Fire effect on ground-foraging ant assemblages in northeastern Argentina. Journal of Insect Conservation. 18:339-352.

Cancelado, R., and T. R. Yonke. 1970. Effect of prairie burning on insect populations. Journal of the Kansas Entomological Society. 43:274-281.

Certini, G., D. Moya, M. E. Lucas-Borja, and G. Mastrolonardo. 2021. The impact of fire on soil-dwelling biota: a review. For Ecol Manag 488:118989.

Cloudsley-Thompson, J. L. 1991. Ecophysiology of desert arthropods and reptiles. Springer-Verlag, Berlin.

Collins, S. L. and L. L. Wallace. 1990. Fire in North American Tallgrass Prairies. University of Oklahoma Press, Norman, OK.

Day, J. D., J. H. Birrell, T. J. Terry, A. Clark, P. Allen, and S. B. St. Clair. 2019. Invertebrate community response to fire and rodent activity in the Mojave and Great Basin Deserts. Ecology and Evolution. 9:6052-6067.

EDIT. Ecosystems Dynamics Interpretive Tools: Ecological Site Descriptions [Internet]. 2024 Feb 5 [cited 2024 Feb 16]. Available from: https://edit.jornada.nmsu.edu/catalogs/esd.

Euler, D. L., and D. Q. Thompson. 1978. Ruffed grouse and songbird foraging response on small spring burns. New York Fish and Game Journal 25:156-164.

Evans, E.W. 1984. Fire as a natural disturbance to grasshopper assemblages of tallgrass prairie. Oikos. 43:9-16.

_____, 1988. Community dynamics of prairie grasshoppers subjected to periodic fire: predictable trajectories or random walks in time. Oikos. 52:283-292.

Fuhlendorf, S. D., & Engle, D. M. 2001. Restoring heterogeneity on rangelands: ecosystem management based on evolutionary grazing patterns: we propose a paradigm that enhances heterogeneity instead of homogeneity to promote biological diversity and wildlife habitat on rangelands grazed by livestock. BioScience. 51:625-632.

Fuhlendorf, S. D., W. C. Harrell, D. M. Engle, R. G. Hamilton, C. A Davis, and D. M. Leslie Jr. 2006. Should heterogeneity be the basis for conservation? Grassland bird response to fire and grazing. Ecological applications. 16:1706-1716.

Gilbert, N., and D. A. Raworth. 1996. Insects and temperature -- a general theory. Canadian Entomologist. 128: 1-13.

Haddad, C. R., S. H. Foord, R. Fourie, and A. S. Dippenaar-Schoeman. 2015. Effects of a fast-burning spring fire on the ground-dwelling spider assemblages (Arachnida: Araneae) in a central South African grassland habitat. African Zoology. 50:281-292.

Hatch, S. L., K. N. Gandhi, and L. E. Brown. 1990. A Checklist of the Vascular Plants of Texas. MP 1655. The Texas Agr. Exp. Sta., College Station, TX.

Humphrey, R. R. 1958. The desert grassland: a history of vegetational change and an analysis of causes. Botanical Revue. 24:193-252.

Hurst, G. A. 1972. Insects and bobwhite quail brood habitat management. Proceeding of the first bobwhite quail symposium. Stillwater, OK. 1:65-82.

Kim, T. N. and R. D. Holt. 2012. The direct and indirect effects of fire on the assembly of insect herbivore communities: examples from the Florida scrub habitat. Oecologia: 168:997-1012.

Komarek, E. V. 1970. Insect control-fire for habitat management. Proceedings of the Tall Timbers Conference on Ecological Animal Control by Habitat Management. Tallahassee, FL. 2:157-171.

McClaran, M. P. 1995. Desert Grasslands and Grasses. Pages 1-30, *in* M. P. McClaran and T. R. van Devender (eds.). The Desert Grassland. University of Arizona Press, Tucson, AZ.

McPherson, G. R. 1995. The role of fire in desert grasslands. Pages 130-151, *in* M. P. McClaran and T. R. van Devender (eds.). The Desert Grassland. University of Arizona Press, Tucson, AZ.

Mukwevho, L., M. Ndlovu, G. Chikowore, T. Dalu, R. Mutamiswa and F. Chidawanyika. 2024. Response of ground-dwelling arthropods to long-term prescribed fire regimes in a savanna-protected area. Rangeland Ecology & Management. 95:46-55.

Parmenter, R. R., M. Kreutzian, D. I. Moore, and D. C. Lightfoot. 2011. Short-term effects of a summer wildfire on a desert grassland arthropod community in New Mexico. Environmental Entomology. 40:1051-1066.

Pavon, M. L. 1995. Diversity and response of ground cover arthropod communities to different seasonal burns in longleaf pine forests. M.S. Thesis. Florida A&M University, Tallahassee, FL.

Payne, N. F. and F. C. Bryant. 1994. Techniques for wildlife management of uplands. McGraw-Hill, Inc., New York, NY.

Peters, T. M. 1988. Insects and human society. Van Nostrand Reinhold Company, New York, NY.

Porter, K. R. 1972. Food relations of amphibians and reptiles, pp. 315-331. In: Herpetology. W. B. Saunders Company, Philadelphia, PA.

Reed, C. C. 1997. Responses of prairie insects and other arthropods to prescription burns. Natural Areas Journal. 17:380-385.

Renwald, J. D. 1978. The effect of fire on woody plant selection by nesting nongame birds. Journal of Range Management. 31:467-468.

Richerson, J. V., and P. E. Boldt. 1995. Phytophagous insect fauna of Flourensia cernua (Asteraceae: Heliantheae) in Trans-Pecos Texas and Arizona. Environmental Entomology 24:588-594.

Schnapp, L. A. and R. J. Kinucan. 2010. An abbreviated fire history of the southwestern United States. Pages 1-7 *in* K. A. Cearley (ed.). Trans-Pecos Prescribed Fire Symposium. April 8-9, 2010. Alpine, Texas. Texas AgriLife Extension Service, Amarillo, TX.

Scifres, C. J. and W. T. Hamilton. 1993. Prescribed burning for brushland management: the South Texas example. Texas A&M University Press, College Station, TX.

Seastedt, T. R. and D. A. Crossley Jr. 1984. The influence of arthropods on ecosystems. Bioscience. 34:157-161.

Soil Survey Staff, Natural Resources Conservation Service, United States Department of Agriculture. Official Soil Series Descriptions. [Internet]. 2024 Feb 5 [cited 2024 Feb 16]. Available from: https://websoilsurvey.nrcs.usda.gov/app/WebSoilSurvey.aspx.

SPSS Base Manual. 1997. SPSS Base system user's guide, version 7.5.2. SPSS Inc., Chicago, IL.

Turner, R., R. H. Webb, J. E. Bowers, and J. R. Hastings. 2003. The Changing Mile Revisited: An ecological study of vegetation change with time in the lower mile of an arid and semiarid region. The University of Arizona Press, Tucson, AZ.

Vasconcelos, H. L., R. Pacheco, R. C. Silva, P. B. Vasconcelos, C. T. Lopes, A. N. Costa, and E. M. Bruna. 2009. Dynamics of the leaf-litter arthropod fauna following fire in a neotropical woodland savanna. PloS one, 4(11), e7762.

Vaughan, J. E. 1961. A study and analysis of the vegetation on Elephant Mountain and surrounding area, Brewster County, Texas. M.S. Thesis, Sul Ross State University, Alpine, TX.

Warnock, B. J. 2010. A brief history of fire in the Trans-Pecos Region of Texas. Pages 23-25 *in* K. A. Cearley (ed.). Trans-Pecos Prescribed Fire Symposium. April 8-9, 2010, Alpine, Texas. Texas AgriLife Extension Service, Amarillo, TX.

Warren, S. D., C. J. Scifres, and P. D. Teel. Response of grassland arthropods to burning: a review. 1987. Agriculture, Ecosystems and Environment. 19:105-130.

Watson, A. 1989. Foraging birds favoring burnt ground in harvested fields. Scottish Birds 15:178-179.

Welch, T. G. 1982. Prescribed range burning in central Texas. No. 900-9-82. Texas Agricultural Extension Service, College Station, TX.

Wetzel, R. G. 1983. Limnology. Second edition. CBS College Publication, New York, NY.

Wright, H. A., and A. W. Bailey. 1982. Fire ecology, United States and Southern Canada. Wiley & Sons, New York, NY.

Yekwayo, I., J. S. Pryke, R. Gaigher, and M. J. Samways. 2018. Only multi-taxon studies show the full range of arthropod responses to fire. PLoS One. 13:e0195414.

Zar, J. H. 1996. Biostatistical Analysis. Third Edition. Prentice Hall, Upper Saddle River, NJ.