INDIRECT EFFECTS OF ENERGY DEVELOPMENT IN GRASSLANDS

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INDIRECT EFFECTS OF ENERGY DEVELOPMENT IN GRASSLANDS

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Abstract: Grassland landscapes in North America are undergoing rapid industrialization due to energy development facilitated by the growing popularity of fracking and horizontal drilling technology. Each year over 3 million hectares are lost from grassland and shrubland habitats to well infrastructure. Direct footprints from energy infrastructure cause impacts to vegetation cover, available cattle forage, carbon sequestration potential, and usable space for wildlife. However, legacy effects from well construction and noise pollution, light pollution, and altered viewsheds have the potential to impact areas beyond this direct footprint, causing additive and persistent changes to nearby grassland function. While these additional areas may be small on a well pad basis, they may have substantial cumulative impacts over time. To investigate these effects via a diversity of mechanisms, we studied the seasonal habitat selection of northern bobwhite (*Colinus virginianus, hereafter bobwhite*) in an energy-producing landscape to evaluate space use patterns relative to energy infrastructure. Habitat selection was modeled in the breeding and nonbreeding season using resource Utilization functions (RUFs). We then investigated patterns of vegetation, arthropod, and soil characteristics surrounding well pads to assess small scale environmental gradients extending away from drilling pads via a combination of multivariate and univariate techniques (i.e., Nonmetric dimensional scaling and ANOVA). We found minimal avoidance of energy structures by quail, suggesting a tolerance of moderate development levels. All small-scale effects studied except for soil moisture were impacted at the pad itself ($P < 0.01$). Off-pad impacts to arthropod abundance and biomass were spatially limited to areas close to pads, while vegetation cover was typically lower than the surrounding habitat beyond 10 m of pads. Soil surface temperature was higher at distances close to well pads, and soil moisture was not different between areas close to and far from well pads. Small-scale gradient results indicate vegetation effects around active drilling pads, potentially increasing erosion and decreasing nesting cover, decreasing carbon sequestration potential, and decreasing forage. Collectively, this research highlights the complexity and importance of impact thresholds in landscape fragmentation.

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CHAPTER I

NORTHERN BOBWHITE (*Colinus virginianus*) HABITAT SELECTION IN RESPONSE TO OIL AND GAS DEVELOPMENT

ABSTRACT Despite the fact that the northern bobwhite (*Colinus virginianus*, hereafter bobwhite) is one of the most well-studied species of wildlife in North America, little is known about how bobwhite respond to oil and gas infrastructure. We investigated the impacts of oil and gas development on third-order space use of bobwhite during the breeding and nonbreeding season in a shinnery oak (*Quercus havardii*) mixed grass prairie at Packsaddle Wildlife Management Area (PSWMA) in western Oklahoma, United States. We captured and monitored bobwhite in the breeding season (April 1 – October 1, n = 135) and nonbreeding season (October 1 – April 1, n = 30) and modeled their habitat selection in a resource utilization function (RUF) format. Generally, energy infrastructure effects on quail were neutral, but breeding season bobwhite selected for low traffic roads (β = -0.31 ± 0.15). In the nonbreeding season, areas with a limited viewshed (i.e., areas with limited visibility of well structures; β = -0.03 ± 0.02). At a home range scale of analysis, breeding season bobwhite selected for areas with a high density of low traffic roads (β = 0.05 ± 0.03). Selection did not differ between sexes except for low traffic roads (males, β = -0.18 females, β= -0.49) and well pads (males, β=-0.22, females, β = 0.37). These results indicate that habitat selection is a complex issue that depends on seasonal and demographic factors. As a generalist, bobwhite seem to be tolerant of energy infrastructure from a third-order habitat selection standpoint at moderate development levels.

INTRODUCTION

Recent trends in fossil fuel extraction, as well as the widespread advent of alternative energy development such as windfarms and solar fields, have resulted in the proliferation of industrialized landscapes in previously unfragmented areas (Allred 2015). Direct impacts to wildlife (i.e., those caused by the deterioration of habitat from the physical footprint of access roads and well pads) such as the complete denuding of vegetation are readily apparent. However, indirect impacts such as increased noise, obstructed viewsheds, and fragmented landscapes can be more subtle and are often confounded with each other (Bayne et al. 2008). The effects of noise, light, industrial activities, and dust all diminish with distance from disturbance and can be difficult to separate (Summers et al. 2011). Additionally, species responses to development are known to vary seasonally and demographically. For example, grouse have shown a greater overall sensitivity to oil and gas development during the lekking season (Hovick et al. 2014), while male grizzly bears (*Ursus arctos*) are less impacted by oil and gas structures than females (Laberee et al. 2014). The increased use of hydraulic fracturing has caused rapid development of fossil fuels in North America. These rapid developments cause numerous issues, such as landscape fragmentation, vegetation alteration, increased human traffic, and noise and light pollution. This requires an understanding of how energy development affects wildlife at large scales during critical life stages and between age classes and sexes.

The land area of oil and gas infrastructure (e.g., well pads, pump stations, access roads) in North America has resulted in the loss of approximately 3 million ha of grassland and shrubland in the Great Plains alone (Allred et al. 2015). Aside from these direct impacts of habitat loss, organisms respond behaviorally to many indirect facets of energy development such as noise, light pollution, altered fearscapes (i.e., structure-based factors affecting prey visibility and concealment; Olsoy et al. 2015), and traffic. For example, noise from natural gas compressors has been demonstrated to alter distribution of pinyon pine (*Pinus edulis*) by impacting pollinator and seed disperser behavior (Francis et al. 2012).The loss of usable space for wildlife due to energy activities through direct and indirect means represents a significant impact to grassland and shrubland wildlife species, and indirect effects may cause effective habitat loss to extend far beyond the physical footprint of development (Bayne et al. 2008).

When modeling impacts to sensitive species, it is important to evaluate effects on a seasonally- and demographically-specific basis. Species may experience differing vulnerabilities to fragmentation and loss of usable space depending on particular life stage requirements. For example, the current energy policy for the greater sage-grouse *(Centrocercus urophasianus)* is focused on development restrictions surrounding leks, but it does not address multiscale impacts to habitat selection due to the well density around critical wintering areas (Carpenter et al. 2010, Doherty et al. 2008, NRCS, 2010). Lesser prairie-chickens (*Tympanuchus pallidicinctus*) also use space differently by season and require seasonal and multi-scale study in order to adequately conduct conservation planning through a better understanding of population vital rates and distribution shifts (Hagen et al. 2016). Resistance to crossing powerline corridors by lesser prairie-chickens results in a year-round loss of usable space (Pruett et al. 2009), representing localized impacts to individuals and populations. In addition, power lines may prove particularly impactful during the fall dispersal period for lesser prairie-chickens, obstructing gene flow and diminishing population connectivity, impacting metapopulations over large geographic and temporal scales (Pruett et al. 2009). Even within specific life stages, differences in habitat selection with respect to energy development across sexes and diurnal periods may differ significantly. Mule deer (*Odocoileus hemionus*) and grizzly bears avoid roads and active well pads more strongly during the daytime, and grizzly bears avoid energy development more strongly in the fall than in the springtime (Laberee et al. 2014, Northrup et al. 2015). In addition, local extinctions and selection impacts due to energy development may only occur after a significant time lag (Gilbert and Chalfoun 2011), likely necessitating multi-year studies to fully assess impacts of energy development to species.

Despite the significant body of work providing information on the impact of energy development on grassland birds, few studies have investigated the effects of oil and natural gas extraction on northern bobwhite (*Colinus virginianus;* hereafter, bobwhite). Bobwhite are one of the most extensively studied wildlife species in North America, and their broad decline has been attributed to habitat loss and fragmentation through land use change such as industrialized forestry and crops and increases in urbanization (Brennan 1994, Guthery et al. 2000, Sandercock et al. 2008, Veech 2006). As a relatively sedentary species, bobwhite may be particularly vulnerable to the fragmentary effects of energy development.

The few existing studies documenting the effects of energy structures on quail have found mixed effects of infrastructue roads, neutral effects of well pads, and negligible impacts of energy infrastructure on hunting mortality risk (Dunkin et al. 2009, Richardson et al. 2008, Tanner et al. 2016a). However, these studies were conducted in areas with a low density of energy structures. The authors proposed that although they did not find substantial impacts from energy structures, there exists a higher threshold of energy development at which point landscape-level suitability would diminish for bobwhite. Conducting habitat selection analysis at Packsaddle Wildlife Management Area (PWMA, the same study area as Dunkin et al. [2009]) allowed us to build upon previous findings after a period of substantial energy development to further pinpoint this threshold. In addition, previous PWMA energy research did not account for the effects of vegetation cover on habitat selection. Also, many efforts to document habitat selection patterns of bobwhite may be biased towards roads due to trapping effort, which makes it difficult to draw conclusions from results. By studying bobwhite in an area of high energy density at a home range level of analysis, we are able to quantify the effects of individual structures while reducing the effects of trapping bias.

In order to better understand how bobwhite respond to anthropogenic fragmentation caused by energy development, we investigated third-order space use (i.e., space use within an individual's home range) using a multi-scale, multi-season approach.

Our objectives were to: 1) determine if anthropogenic impacts related to energy (i.e., oil and gas) development affect the habitat selection of bobwhites, 2) determine if those effects vary seasonally for both individual bobwhites and coveys, 3) determine differences in selection for anthropogenic features by sex, and 4) assess effects of anthropogenic structure density on bobwhite habitat selection.

METHODS

Study Area

PWMA encompasses 7,956 ha and is located approximately 27 km south of Arnett, Oklahoma. The study site is largely comprised of shinnery oak (*Quercus havardii*) and mixed grass prairie. Other dominant shrubs include Chickasaw plum (*Prunus angustifolia*), skunkbush sumac (*Rhus trilobata*), and sand sagebrush (*Artemisia filifolia*). Forb cover is diverse, and abundant species include prairie sunflower (*Heliolanthus petiolaris*), erect dayflower (*Commelina erecta*), Indian blanket (*Gaillardia pulchella*), Texas croton (*Croton texensis*), and western ragweed (*Ambrosia psilostachaya*). Little bluestem (*Schizachyrium scoparium*), sand big bluestem (*Andropogon hallii*), Indiangrass (*Sorghastrum nutans*), switchgrass (*Panicum virgatum*), side-oats grama (*Bouteloua curtipendula*) and hairy grama (*Boutelou hirsuta*) constitute the dominant graminoid species at the study site (DeMaso et al. 1997, Townsend 1999). Dominant soils at PWMA include Nobscot fine sand, sandy Nobscot-Brownfield, sandy Pratt-Carwile, and loamy Quinland-Woodward (Dunkin et al. 2009). Elevation on the WMA ranges from 579 m to 762 m above sea level (Townsend et al. 1999). The 30-year average annual rainfall levels for the area are 659.89 mm (Oklahoma Mesonet 2014).

The site undergoes habitat management activities in the form of cattle grazing at a stocking rate of 40 ha animal^{-1} (T. Atha, Oklahoma Department of Wildlife Conservation, personal communication), prescribed burning, and planting of wildlife food plots. Approximately 25% (2,950 ha) of the WMA has been burned each year since 2004 (S. Parry, Oklahoma Department of Wildlife Conservation, personal communication). The WMA has been subject to substantial energy development, and there are currently 70 active wells on the property. Fifty of the wells were installed after 2002. This recent development has also created a network of access roads expanding from 31 km (0.39km/km^2) in 2002 (Dunkin et al. 2009) to 97 km (1.21km/km^2) in 2016. In fact, much of the WMA was inaccessible by road prior to recent energy development.

Capture and Monitoring

In order to determine bobwhite space use, we captured bobwhite using wire walk-in funnel traps baited with milo and corn chop (Stoddard 1931). Trap locations were placed both randomly and in the vicinity of known coveys based on visual sightings for birds and sign at a range of distances from roads. Traps were monitored twice daily in the morning and evening. Once captured, bobwhite were weighed, aged, and sexed. We then attached aluminum leg bands to birds weighing >90 g, and those birds weighing >132 g were also fitted with 5.6 g necklace-style VHF radio-transmitters. Radio-transmitters had a battery life of 113-221 days and were equipped with mortality sensors (Advanced Telemetry Systems, Isanti, MN). Capture and handling of bobwhite was approved by Oklahoma State University Animal Care and Use Committee (ACUP No.: AG-11-22). Monitoring of individuals occurred ≥5 times per week during the breeding season (i.e., April 1- Oct 1) and ≥3 times per week during the nonbreeding season (i.e., Oct 1 – April

1) by homing using a Yagi antenna and Communication Specialists receiver (Communications Specialists, Inc., Orange, CA). Only one location per individual per day was recorded and monitoring was conducted at a range of diurnal times for each individual to avoid temporal bias of points.

Data Layer Creation

The vegetation map used for analyses was created using IKONOS-2 2 m imagery (DigitalGlobe, Westminster, CO). The final map was created using the Maximum Likelihood classification tool in ArcMap 10.3 (Environmental Systems Research Institute, Redlands CA). Sixty training sites were used per cover class (see below) and the final map was field verified for accuracy. Bobwhite vegetation preferences are well-documented (Guthery 2000, Tanner et al. 2016a) and therefore, we decided to focus on anthropogenic features (e.g., roads and well pads) by limiting cover classes to five coarsely-themed categories: water, bare ground, grass/forb, short woody vegetation, and tall woody vegetation . Tall woody vegetation was defined as motte-forming and riparian tree species such as shinnery/post oak (*Quercus stellata*) hybrids, elm (*Ulmus* spp.), cottonwood (*Populus deltoides*), black locust (*Robinia pseaudoacacia*), eastern red cedar (*Juniperus virginiana*), salt-cedar (*Tamarisk* spp.), willow (*Salix* spp.), hackberry (*Celtis occidentalis*), soapberry (*Sapindus saponaria*), Osage orange (*Maclura pomifera*), and persimmon (*Diospyros* spp.). Short woody species consisted of sand sagebrush, shinnery oak, smooth sumac (*Rhus glabra*), skunkbush sumac (*Rhus trilobata*) and Chickasaw plum (Table 1).

Roads and well pads were digitized from IKONOS-2 imagery and subsequently, field verified to confirm existence and activity status. Well pads were considered active following their spud date (i.e., date of commencement of drilling), which was obtained from the Oklahoma Corporation Commission. Distance rasters were created using the Euclidean distance tool in ArcMap 10.3 (Environmental Systems Research Institute, Redlands, CA). Diamond Traffic Tally 2 single road tube accumulators (Diamond Traffic Products, Oakridge, OR) were deployed at 25 sites during the study period to determine vehicle traffic. Because actual traffic volumes could not be estimated as counters only recorded axle hits and typical WMA traffic had varying numbers of axles, axle hits were used as an index of traffic volume. Separate distance rasters were created for roads with > 20 axle hits per day based on findings that this traffic volume began to negatively impact greater sage grouse lek attendance (Halloran 2005).

An anthropogenic viewshed was developed for PWMA for inclusion in habitat modeling. The heights of brine storage tanks, gas vents, pump jacks, and flame towers were measured using a clinometer. Pump jack heights were defined as the maximum structure height during the pumping cycle. Tank batteries were considered to be a single object for viewshed calculation. Structure heights were averaged within structure types for vents (7.65 m ± 0.62), pump jacks (9.90 m ± 0.70), short tank batteries (6.58 m ± 0.27), and tall tank batteries (7.98 m ± 0.28). Structure footprints (i.e., the extent of applied fill) were digitized using IKONOS-2 imagery cross-referenced with digital orthophoto quadrangles. For each structure, a binary viewshed raster was created using the Viewshed tool in ArcMap 10.3 (Environmental Systems Research Institute, Redlands, CA). Rasters for each structure were then summed to create a layer depicting the number of anthropogenic structures visible at each location on the landscape.

Data Analysis

Space use was modeled in a resource utilization function (RUF) framework (Marzluff et al. 2004, Millspaugh et al. 2006). RUF habitat selection modeling is a third-order habitat selection approach, used to analyze relative space use by the individual within its home range. Fixed-kernel home ranges are then used to create a utilization distribution (UD), which details relative use. For each year, we constructed RUFs for individuals and coveys that had ≥ 30 locations in their respective seasons (Tanner et al. 2016a). Nesting locations were censored from the analysis. Kernel home ranges were calculated using a maximum likelihood bandwidth estimator (Horne and Garton 2006). Because relative use values initially fit a right-skewed distribution, we log-transformed the values to better approximate error homoscedasticity (Winder et al. 2014). Individual RUFs were calculated using the package ruf.fit in Program R (ver. 3.1.1, R Foundation for Statistical Computing, Vienna, Austria). To allow for population-level averaging, we used standardized β coefficients. Breeding season models were based on locational data collected from April 1 to October 1 in 2015 and 2016. Nonbreeding season models were based on locational data collected from October 1 to April 1. Locations of individuals not associated with coveys during the nonbreeding season were censored from analysis. New development was limited to one pad over the course of our study, and we did not distinguish the impacts of well construction versus well production.

To build the RUF models, we then distributed sampling points with a 10 m spacing throughout each home range, and extracted use values and environmental covariates (Winder et al. 2014). Relative use within the home range was then used as the response variable to a suite of covariates in a multiple regression framework. Individual results can then be used to evaluate population-level selection effects. A third-order habitat selection approach permitted us to lessen potential trapping bias towards roads and well structures, as locations where we successfully trapped quail should have a smaller impact on where roads and well pads occur along a usage gradient within that individual's home range than on the impact of the placement of the individual's home range relative to roads and well pads on the landscape (i.e., all collected data were subject to the same bias; Phillips et al. 2009).

The final model included the Euclidean distance to the nearest oil and gas drilling pad, the Euclidean distance to the nearest low-traffic road, the Euclidean distance to the nearest high-traffic road, a viewshed layer representing the number of anthropogenic structures visible from every point on the landscape, and the Euclidean distance to each vegetation class in a thematically coarse vegetation map consisting of low woody cover, high woody cover, grass/forb, bare ground, and water. None of the studied covariates were highly correlated with each other (Pearson's $r \geq 0.6$).

Significance and directionality of selection were assessed for each individual, as well as the percentage of the population responding positively, negatively, and non-significantly to each covariate. These results provided an assessment of the consistency of use patterns within a population (Marzluff et al. 2004). Standardized RUF β estimates for each individual were averaged to obtain population-level β estimates. Population-level variance was estimated using the formula:

$$Var(\bar{\hat{\beta}}_j) = \frac{1}{n-1} \sum_{i=1}^{n} (\widehat{\beta_{ij}} - \hat{\beta}_j)^2,$$

where n is the number of individuals, $\widehat{\beta_{ij}}$ is the value of the coefficient of covariate j for individual i, and β_j is the averaged coefficient for covariate j. This is a conservative estimate of variance and incorporates individual variation (Marzluff et al. 2004). Variance was used to construct 95% confidence intervals, and population level β estimates overlapping zero were deemed nonsignificant at $\alpha=0.05$ (Tanner et al. 2016a, Winder et al. 2014). We then ran a Welch’s t-test in R to compare differences between sexes for all individual breeding season beta coefficients.

Utilization distributions were created using the above methods. Surrounding each sampling point within the UD, circular buffers were created. Two separate analyses were run using buffer radii corresponding to the radius of the average breeding season home range of bobwhite during the study period (551.00 m ± 24.61[SE]) and covey home range (432.01 m ± 31.26 [SE]). Well density (wells per km^2) and road density (km road per km^2) were calculated for each sampling radius. These values were used as explanatory variables in RUF equations. Significance, directionality, and response percentage distributions were reported as above.

RESULTS

A total of 135 individual home ranges were used in breeding season analysis (72 in 2015 and 63 in 2016), which included 7,000 unique locations (mean = 51.8 locations per individual, range = 30 to 117 locations). Our analyses included 58 females and 75 males (two juveniles with unknown sex were censored from analysis). For nonbreeding season analysis, we used 30 unique coveys (12 in 2015 and 18 in 2016) that included 1,253 unique locations (mean = 41.8 locations per covey, range = 30 to 61 locations).

In general, breeding season resource selection results for bobwhites were largely neutral. For the majority of covariates studied, 95% confidence intervals overlapped 0 (Table 2). Selection for grass/forb patches was significant at the population level (Table 2), albeit β estimates were near 0 indicating a weak selection for grass/forb patches. This result may not be biologically significant, as individual-level results revealed that only a single individual responded significantly to grass/forb patches. Similarly, bare ground was significantly avoided at the population level, but the overall effect was weak (Table 2). At the population level, quail exhibited strong selection for low traffic roads (Table 2). Unlike selection for grass/forb patches and against bare ground, the strength of selection is also reflected in individual level results with 48% of the individuals significantly selecting for areas closer to low traffic roads.

During the breeding season, there were no significant differences between sexes for the majority of selection coefficients. However, male and female quail did differ in their selection for well distance, with female quail significantly avoiding well pads and male quail significantly selecting for well pads. Well distance was one of the covariates with the largest confidence interval for breeding season selection, widely overlapping 0 (Table 2). It was also the covariate with the most evenly-divided responses at the individual level (33%, 24%, and 44% individual bobwhite responded with positive, negative, and neutral selection coefficients, respectively; Figure 1). We further investigated this relationship by modeling predicted UD values as a function of distance from well pad for each sex. Results showed that male bobwhite UD values ranged from 0.046 to 0.066 higher than female bobwhite up to 400 m away from well pads, indicating a greater preference for well pads by males at close distances. Both breeding season males and breeding season females selected for areas closer to low traffic roads, but females selected more strongly than males for these areas (Table 3).

During the nonbreeding season, covey resource selection was also largely nonsignificant. The only covariates significantly selected were short woody vegetation and visible wells at the population level. Coveys selected for areas closer to short woody vegetation at the population level and avoided areas with a high amount of visible structures (Table 4). These effects were also small, with only 10% of individuals responding significantly negatively for each case (Table 4).

Analysis at the scale of the daily movement distance was not possible using an RUF approach due to the fact that for many individuals and coveys, well density and road density were consistently 0 wells per km^2 and 0 km of road per km^2 within the home range buffer. Therefore, results are only presented for the home range level of analysis. For these reasons, only low-traffic roads were analyzed at a home range scale.

During the breeding season, bobwhite were selecting areas with high road densities within their home range (Table 5). Individually, 25.9% of the population responded positively to road density. As distance to low traffic road increased, 11.2% of individuals responded with lower use values, and 62.9% of individuals exhibited no significant habitat use response. In contrast, bobwhite did not significantly respond to road or well density during the nonbreeding season (Table 5).

DISCUSSION

At the current level of development at PWMA, bobwhite habitat selection generally does not suggest avoidance of energy infrastructure, corroborating previous results (Dunkin et al. 2009, Tanner et al. 2016a). Due to the small selection effects for known preferred cover classes in our study (Table 2, Table 4), as well as the small average distances to preferred cover types (Table 6), we postulate that seasonal availability of habitat in our study site is high and largely nonlimiting. In such landscapes, moderate levels of energy development do not appear to limit space use by bobwhite. At higher structure densities, it is likely that direct losses of space will impact bobwhite populations, however, the direct footprint of energy development currently occupies 3.19 km^2 in our study area, or roughly 3.8% of the total landscape.

Selection for low-traffic roads by quail has been previously documented, though reasons for this have only been postulated (Dunkin et al. 2009, Wellendorf et al. 2002). Past studies have suggested that this may be due to a bias incurred by trapping preferentially closer to roads, or that bobwhite may prefer roads with a low traffic volume as travel corridors (Dunkin et al. 2009). Selection for low traffic roads during the breeding season represented the strongest effect of any anthropogenic covariate we studied (Table 2). It is unlikely that trap placement influenced our results as we attempted to include a variety of distances from roads in our trap lines, and achieved good coverage of the study area. Moreover, RUF habitat selection modeling analyzes third-order (within home range) habitat selection trends, which allowed us to address trapping biases inherent in second order (i.e., placement of home range) habitat selection studies.

To our knowledge, empirical evidence documenting bobwhite use of roads as travel corridors does not exist, though we anecdotally observed bobwhite travelling along roadways often during the study period. Bare ground bordered by dense cover of grasses and forbs, such within road corridors, may be used by bobwhite during the breeding season (Rosene 1969) as forbs provide food resources as well as open foraging cover for broods to forage for arthropods. Like many avian species, breeding bobwhite and bobwhite broods depend on arthropods for abundant and easily assimilable protein (Doxon and Carroll 2007, Wiens and Rotenberry 1979). Forbs can support higher abundances of arthropods than grasses, bare ground, and woody cover (Hill 1985) as well as improved mobility for foraging individuals (Doxon and Carroll 2007).This idea is reinforced by our findings that female bobwhite select more strongly for areas closer to low traffic roads than males during the breeding season.

Selection for low-traffic roads may have negative implications as well. The use of roads by vertebrate prey species has been repeatedly documented, and is thought to be a response to improved foraging efficiency (Camacho et al. 2017, Jackson 2003, Barrientos and Bolino 2009). However, mesopredators often experience greater foraging efficiencies along road corridors (Bergin et al. 1997, Suarez-Esteban et al. 2013, Tigas et al. 2002). Prey species have been documented using risky habitats to benefit from increased foraging efficiency despite greater predator encounter rates (Camacho et al. 2017, Heithaus and Dill 2006, Wirsing et al. 2007). Thus, despite the perceived benefits of selecting for areas near roads, bobwhite may be incurring a greater predation risk leading to decreased fitness. Research in a similar system has suggested increased bobwhite mortality closer to roads independent of hunting pressure (Tanner et al. 2016a).

Our results also indicate sexual divergence in selection patterns for well pads by bobwhite. Higher use of areas near well pads by male bobwhite is not supported in the literature to our knowledge. However, disturbed areas surrounding well pads often have shorter vegetation than the surrounding landscape (Rodgers and Koper 2017), and male bobwhite are known to select for areas with high visibility for singing (Carter et al. 2002). In addition, many well pads on PWMA are on prominent ridgetops and small hills, further enhancing the detectability by nearby females of males using these areas.

Though selection for road and well density within the radius of a quail home range is still relatively small in scale, it allows us to observe that quail are making third-order selection decisions based on landscape configuration as well as the location of individual features. Though feature density and the distance to nearest feature are correlated, it is important to note that quail are compatible with well pads as well as areas with higher densities of well pads at this level of development. This suggests that additive indirect effects of multiple well pads on the landscape on bobwhite habitat selection are not negative at the level of development we evaluated.

Our results indicate that bobwhite coveys exhibit positive selection for short woody vegetation during the nonbreeding season. Bobwhite are known to select for woody cover during this period. This is thought to provide thermoregulatory benefits (Janke et al. 2013, Johnson and Guthery 1988, Yoho and Dimmick 1972). Short woody vegetation may provide superior cool weather thermal and predation cover than tall woody vegetation, explaining seasonal differences in vegetation selection (Carrascal et al. 2001).

During the nonbreeding season, bobwhite also selected for areas with low structure visibility. There may be several reasons for this. Presumably, anthropogenic structures are more visible to bobwhite during the dormant season (when leaves have senesced and dropped) than during the growing season. Our viewshed model did not include vegetation when determining the visibility of anthropogenic structures, thus potentially overstating the importance of tall structures in a landscape during the nonbreeding season. In addition, anthropogenic viewshed values are likely correlated with landscape-level exposure. Bobwhite in Illinois showed a preference for lower elevations for winter roosting cover, presumably to benefit from greater shrub cover and reduce wind exposure (Klimstra and Ziccardi 1963). That is, areas of the landscape that are sheltered from winter exposure are likely to have few visible wells. Finally, areas with a low viewshed value are more likely to be farther away from roads, and selection for small viewsheds could be a response to hunting pressure (Richardson et al. 2008). Further research is needed to parse out these individual effects.

Bobwhite have several characteristics that may make them tolerant of a landscape fragmented by energy deelopment. They are a generalist species, and likely have a degree of habitat “slack” that allows them to persist in a variety of plant communities and vegetation arrangements (Guthery 1999). The theory of habitat slack suggests that there is a range of habitat configurations that maximize bobwhite habitat suitability. Assuming that this theory is not limited to the peak of a habitat suitability model, bobwhite may have a range of habitat parameters conducive to a use probability across all levels of use.

Thus, the range of distances to grass and forb patches may all fall within an interval that allows for high habitat suitability for bobwhite, maximizing available space on the landscape. This degree of adaptability may also help to explain a tolerance for energy infrastructure. Though bobwhite are successful in mixed grass prairie, they are also adapted to open woodlands and pine savannahs. In this sense, bobwhite may be more evolutionarily adapted to coexist with tall structures and impacted viewsheds (i.e., pump jacks, gas vents, and tank batteries) than gallinaceous birds that are prairie and sagebrush obligates, such as greater prairie-chickens (*Tympanuchus cupido*), lesser prairie-chickens, greater sage-grouse, and Gunnison sage-grouse (*Centrocercus minumus*; Dunkin et al. 2009).

It is also important to note that our results dictate selection of high use areas within an individual’s home range. It is likely that bobwhite situate their home ranges using different cues than those used to select habitat within their home range. Lesser prairie chicken home range size and placement has been shown to change following energy development (Hagen et al. 2011, Winder et al. 2014). We chose to avoid investigating bobwhite habitat selection at a second order (i.e., the placement of their home range on the landscape). While it is necessary to study the impacts of energy on the placement of bobwhite home ranges, we felt that in this case, bobwhite home range location would be largely an artifact of where bobwhite were trapped on the landscape. Our capture effort was intended to distribute traps throughout the landscape while relocating traps that had been unsuccessful to maintain adequate numbers of radiomarked individuals. This resulted in a distribution of locations that was somewhat skewed towards roads and well pads, which, despite our efforts, may not represent how bobwhite were actually located on the landscape. Additionally, none of the studied covariates were highly correlated with each other, and thus it is unlikely that selection trends for the covariates studied are confounded at this scale of analysis.

Though we found that third-order habitat selection is largely unaffected by energy infrastructure, it is possible that indirect effects are of consequence to bobwhites in ways that do not affect habitat selection, such as through noise pollution. The volume of gas compressors present on drilling pads has been measured at an average of 55 dBA (Francis et al. 2011), a level considered to be an occupational hazard for humans. Compressor noise may hamper predator detection ability, leading to decreased survival and higher rates of nest depredation (Francis and Barber 2013). Furthermore, compressor sound and traffic noise, like many sources of anthropogenic noise pollution, are also the most energetic in low frequency bands, which propagate widely and may overlap with bobwhite vocalizations (Francis 2009). This may hamper mate selection and covey reformation through acoustic masking, leading to decreased fitness in populations subject to noise pollution (Barber et al. 2010).

Future energy development has the potential to reduce existing habitat quality through increased traffic on existing roads coupled with reduced patch size for remaining habitat. Bobwhite did not select for high traffic roads in our study, indicating a potential impact threshold for vehicle traffic. Additional well development may eventually exceed the traffic threshold for impacting bobwhites. It is estimated that an average fracked natural gas pad requires between 4,315 and 6,590 vehicle visits over its lifetime, 70% of which are concentrated during the initial fracking period (Goodman et al. 2016). Additionally, old wells may be refracked, contributing further to traffic disturbance.

Whether or not development requires the installation of new roadways, additional traffic to service new wells on existing roads introduces additional lethal and sublethal effects such as increased noise and collision hazards (Blickley et al. 2012, McClure et al. 2013, Summers et al. 2011).

While newly popular methods in oil and gas extraction technology such as fracking and multibore well pads have permitted the rapid industrialization of western landscapes, they also offer a unique opportunity for landscape planning to avoid indirect fragmentation impacts to sensitive species and habitats. Current technology allows well bores to extend up to 3.2 km from a drilling pad, allowing land managers to greatly influence the spatial arrangement of energy infrastructure (Thompson et al. 2015). Developers can make use of new energy planning software (e.g., Bungee Infrastructure Planning Software; Milt et al. 2016) to conduct site-specific analyses that compare the environmental impacts of well configuration to the increased development cost of mitigating those impacts (Milt et al. 2016). For example, with informed modeling approaches, researchers were able to reduce impacted space for Sprague's pipits (*Anthus spragueii*) by 20% through the manipulation of well location (Thompson et al. 2015). In a cost-benefit analysis, an average of 38% of shale gas ecosystem impacts could be avoided through well pad placement before the installation cost of the project became prohibitively expensive (> 20% increase in estimated development cost; Milt et al. 2016). These tools will allow natural resource managers to develop empirically-derived impact mitigation standards for future development to avoid the destruction of critical species and habitats by planning additional development in a way that is compatible with impact thresholds to maximize usable space to target species.

Implications

Habitat fragmentation and land use change have been identified as the key contributors to bobwhite decline **(**Brennan 1994, Guthery et al. 2000, Sandercock et al. 2008, Veech 2006), though at our scale of analysis and level of site energy development, the presence of energy infrastructure may neither alter habitat functionality nor result in fragmentary landscapes for a generalist avian species. If roads and well pads are not avoided or selected for by bobwhite, they then may simply create relatively small areas of low-suitability habitat without degrading existing available habitat. This represents space use consequences due to the direct effects, but not indirect effects of energy infrastructure. Though energy development does not seem to strongly impact bobwhite habitat selection, lack of avoidance may cause individuals to select for risky habitats leading to decreased survival (Battin 2004, Robertson and Huto 2006, Tanner et al. 2016a). By building upon the results of Dunkin et al. (2009), we have demonstrated that at a population level, quail select for low traffic roads and do not avoid well pads at our study site at low and moderate levels of development. However, our results show that selection patterns are more complex than previously thought, and differ seasonally and demographically. PWMA is a multi-use property, and our results indicate managing for quail and allowing oil and gas extraction may not be conflicting objectives.

Our results contribute to understanding the effects of energy fragmentation on grassland birds by reinforcing the idea that habitat fragmentation is a diverse and complex issue. Largely linear intrusion into an unfragmented landscape has different effects than a checkerboard matrix of suitable and less suitable habitat. The former does not result in a large amount of habitat loss, which has been implicated as a larger driver of local extinctions and population-level effects than fragmentation alone (Fahrig 2003). Even for species compatible with low and moderate densities of development, it is important to identify impact thresholds to guide future development.

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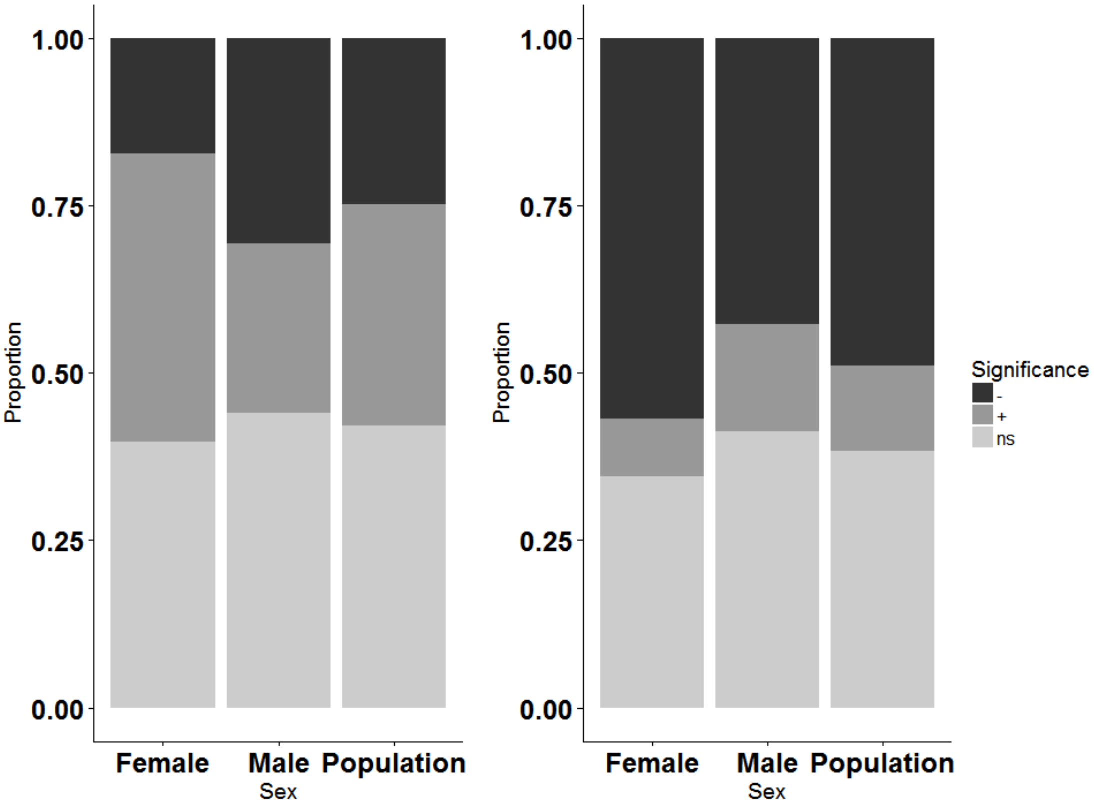

Figure 1. Proportions of female, male, and total population of northern bobwhite responding positively, negatively, and nonsignificantly to distance to wells (A) and distance to roads (B) during the breeding season at Packsaddle Wildlife Management Area in western Oklahoma, United States, 2015 and 2016. -, +, and ns indicate negative, positive, and nonsignificant beta coefficients (positive beta coefficients indicate avoidance of variable).

Table 1. Description of cover classes used in resource utilization functions (RUFs) modeling of northern bobwhite space use at Packsaddle Wildlife Management Area in western Oklahoma, United States in 2015 and 2016.

Cover Type	**Description**	**Associated Plant Species**
Water	Open or shallow water, including emergent vegetation	N/A
Grass/Forb	Mixed-grass prairie and forb cover, may be interspersed with low shrubs	*Heliolanthus petiolaris*, *Commelina erecta*, *Gaillardia pulchella*, *Croton texensis*, *Ambrosia psilostachaya*, *Schizachyrium scoparium*, *Andropogon hallii*, *Sorghastrum nutans*, *Panicum virgatum*, *Bouteloua curtipendula*, *Boutelou hirsuta*
Tall Woody Vegetation	≥ 2 m woody vegetation	*Quercus stellata*, *Quercus stellata/Quercus havardii*, *Ulmus* spp., *Populus deltoides*, *Robinia pseaudoacacia*, *Juniperus virginiana*, *Tamarisk* spp., *Salix* spp., *Maclura pomifera*, *Diospyros* spp.
Short Woody Vegetation	≤ 2 m woody vegetation	*Artemisia filifolia*, *Quercus havardii*, *Prunus angustifolia*
Bare Ground	Sparse vegetation, sand, gravel, caliche	N/A

Table 2. Selection for vegetation characteristics and anthropogenic features or individuals and the population during breeding season for northern bobwhite at Packsaddle Wildlife Management area in western Oklahoma, United States in 2015 and 2016. Covariates marked with an asterisk denote significance at α = 0.05. +, -, and ns denote the proportion of individuals responding with significant positive, significant negative, and nonsignificant selection coefficients indicating distance to anthropogenic features (m), cover types (m), or the number of visible structures.

	Population-Level Effects			**Individual-Level Effects**		
Covariate	**β**	**LCI**	**UCI**	**+**	**-**	**ns**
Water	0.00	-0.09	0.09	0.30	0.22	0.48
Grass/Forb *	-0.01	-0.02	-0.01	0.00	0.01	0.99
Tall Woody	-0.02	-0.04	0.00	0.10	0.17	0.73
Bare *	0.05	0.00	0.09	0.30	0.13	0.57
Short Woody	-0.01	-0.03	0.00	0.03	0.05	0.92
Well Distance	0.04	-0.25	0.32	0.33	0.24	0.44
High Traffic Rds	-0.13	-0.40	0.13	0.22	0.32	0.46
Low Traffic Rds*	-0.31	-0.46	-0.16	0.13	0.48	0.39
Viewshed	0.00	-0.01	0.02	0.08	0.07	0.84

Table 3. Differences in selection for vegetation characteristics and anthropogenic features by breeding season for male and female northern bobwhite at Packsaddle Wildlife Management Area in western Oklahoma, United States in 2015 and 2016.

Covariate	**Male Mean β**	**Female Mean β**	**df**	**t**	**p**
Water	-0.03	0.04	130.74	-0.79	0.43
Grass/Forb	-0.01	-0.02	128.05	1.11	0.27
Tall Woody Vegetation	-0.02	-0.02	127.60	0.24	0.81
Bare Ground	0.07	0.02	118.72	1.13	0.26
Short Woody Vegetation	-0.01	-0.02	95.96	0.91	0.36
Well Distance*	-0.22	0.37	125.54	-2.12	0.04
High Traffic Rds	-0.26	0.02	124.07	-1.05	0.30
Low Traffic Rds	-0.18	-0.49	100.17	1.91	0.06
Viewshed	0.01	0.00	122.30	0.71	0.48

Table 4. Selection for vegetation characteristics and anthropogenic features during nonbreeding season by northern bobwhite at Packsaddle Wildlife Management Area in western Oklahoma, United States in 2015 and 2016. Covariates marked with an asterisk denote significance at α = 0.05. +, -, and ns denote the proportion of individuals responding with significant positive, significant negative, and nonsignificant selection coefficients indicating distance to anthropogenic features (m), cover types (m), or the number of visible structures.

	Population-Level Effects			**Individual-Level Effects**		
Covariate	**B**	**LCI**	**UCI**	**+**	**-**	**ns**
Water	0.17	-0.19	0.54	0.20	0.30	0.50
Grass/Forb	0.00	-0.01	0.01	0.00	0.00	1.00
Tall Woody	-0.02	-0.07	0.03	0.10	0.10	0.80
Bare	0.01	-0.08	0.10	0.10	0.13	0.77
Short Woody*	-0.03	-0.06	0.00	0.03	0.10	0.87
Well Distance	-0.27	-0.75	0.22	0.13	0.27	0.60
High Traffic Rds	-0.18	-0.44	0.08	0.13	0.27	0.60
Low Traffic Rds	-0.10	-0.24	0.05	0.13	0.33	0.53
Viewshed *	-0.03	-0.05	-0.01	0.00	0.10	0.90
VIRS	0.03	-0.02	0.09	0.33	0.10	0.57

Table 5. Selection for density of well pads and low traffic roads during breeding and nonbreeding season by northern bobwhite at the scale of the average seasonal home range at Packsaddle Wildlife Management Area in western Oklahoma, United States in 2015 and 2016. Covariates marked with an asterisk denote significance at $\alpha = 0.05$. +, -, and ns denote the proportion of individuals responding with significant positive, significant negative, and nonsignificant selection coefficients indicating distance to anthropogenic features (m).

		Population-Level			**Individual-Level**		
Season	**Covariate**	**β**	**LCI**	**UCI**	**+**	**-**	**ns**
Breeding	Wells/km^2	0.02	-0.01	0.06	0.27	0.22	0.52
	Low Traffic Rd km/km^2 *	0.05	0.01	0.08	0.26	0.11	0.63
Nonbreeding	Wells/km^2	0.03	-0.05	0.10	0.43	0.24	0.24
	Low Traffic Rd km/km^2	0.12	-0.01	0.25	0.48	0.14	0.38

Table 6. Average distance to vegetation patches and anthropogenic features (in meters) ± 1 standard error by northern bobwhite for breeding season males, breeding season females, the breeding season population, and the nonbreeding season population at Packsaddle Wildlife Management Area in western Oklahoma, United States in 2015 and 2016.

	Male Breeding Season	**Female Breeding Season**	**Population Breeding Season**	**Population Nonbreeding Season**
Water	352.97 ± 4.58	375.79 ± 5.09	363.96 ± 3.41	339.20 ± 7.80
Grass/Forb	2.48 ± 0.09	2.50 ± 0.10	2.49 ± 0.07	3.36 ± 0.17
Tall Woody	28.97 ± 0.65	32.02 ± 0.72	30.44 ± 0.48	28.14 ± 1.19
Bare Ground	82.53 ± 1.22	82.06 ± 1.21	82.30 ± 0.86	79.71 ± 1.86
Short Woody	8.43 ± 0.27	8.89 ± 0.30	8.65 ± 0.20	5.38 ± 0.28
Well Distance	558.69 ± 5.04	597.42 ± 5.35	577.34 ± 3.68	477.05 ± 7.82
High Traffic Rds	2136.34 ± 26.37	1704.44 ± 22.55	1928.34 ± 17.66	2152.90 ± 36.38
Low Traffic Rds	322.88 ± 5.19	289.64 ± 5.71	306.87 ± 3.85	276.50 ± 7.77

Table S1. Covariates used in Resource Selection Function (RUF) models describing quail habitat selection during the breeding and nonbreeding seasons in 2015 and 2016 at Packsaddle Wildlife Management Area, western Oklahoma, United States.

Covariate	**Definition**	**Source**
Well Distance	Distance to the nearest active drilling pad	Field-verified IKONOS-2 imagery, Euclidean Distance Tool (ArcGIS)
High Traffic Road Distance	Distance to the nearest road with ≥ 20 axle hits/day	Field-verified IKONOS-2 imagery Road tube traffic loggers (Halloran 2005)
Low Traffic Road Distance	Distance to the nearest road with < 20 axle hits/day	Field-verified IKONOS-2 imagery Road tube traffic loggers (Halloran 2005)
Distance to: Water Bare Ground Grass/Forb Short Woody Vegetation Tall Woody Vegetation	Distance to the nearest patch of each cover class	Field-verified maximum likelihood supervised classification (ArcGIS)
Viewshed	Number of Tank Batteries, Pump Jacks, Gas Vents, and Flame Towers Visible	Clinometer, Viewshed tool (ArcGIS)

CHAPTER II

IMPACT OF OIL AND GAS DRILLING PADS ON ARTHROPOD COMMUNITIES ALONG AN ENVIRONMENTAL GRADIENT

ABSTRACT

Despite a rapid proliferation of energy development in the western United States, little is known about the effects of oil and gas infrastructure on the surrounding landscape. In order to characterize small-scale gradients in well pad impacts, we investigated the effects of oil and natural gas well pads on patterns of arthropod (i.e., composition, biomass, and abundance), vegetation (i.e., cover of functional groups, maximum vegetation height, and litter cover), and soil characteristics (i.e., soil surface temperature and soil moisture) in a shinnery oak (*Quercus havardii*) mixed grass prairie. We sampled along distance gradients surrounding active well pads at 64 locations at Packsaddle Wildlife Management Area (PWMA) in western Oklahoma during 2015 and 2016. We found significant positive relationships between distances from well pads and arthropod communities based on abundance and biomass. Ordination of the eight most common arthropod orders we collected (Araneae, Coleoptera, Diptera, Hemiptera, Lepidoptera, Orthoptera, and Hymenoptera) were correlated with vegetation height, litter depth, forb cover, grass cover, bare ground cover, and distance from well pad ($p < 0.05$). Our findings indicate that well pads affect vegetation and arthropod distributions beyond the direct footprint of the pad itself, especially for vegetation characteristics, where impacts typically extended 10 m from the pad. These results may be used to predict the impacts of oil and gas to grassland organisms, as well as to model the full effects of energy development on landscapes.

INTRODUCTION

A major potential cause of anthropogenic gradients in grassland and shrubland ecosystems is the rapid proliferation of energy development, which in North America has created novel fragmented and industrialized landscapes (Allred et al. 2015). In addition to the obvious physical footprint of energy development (i.e., the immediate area that is denuded of vegetation and subjected to regular vehicle traffic), more subtle changes, such as alterations of microtopography and moisture regimes, introduced gases, and sediment deposition may have widespread and synergistic landscape-level effects (Apeagyei et al. 2011, Reid and Dunne 1984). Grading and compaction of soil during well pad construction, as well as the application of fill to the surrounding area may cause erosion, alter hydrology, reduce soil nutrients, and delay reestablishment of natural vegetation communities (Fink and Drohan 2015, Matherne 2006). These alterations may be long-lasting; vegetation community change has been shown to persist from three to five decades after well abandonment (Johnstone and Kokelj 2007, Nasen et al. 2011).

Therefore, off-pad impacts from well construction activities may outlast the extraction activity itself. As future energy demand expands networks of roads, compressors, pump jacks, and storage tanks, small scale changes in microclimate and soil characteristics due to energy extraction activities will cause increasingly large areas of influence. Consequently, it is important to quantify and understand these small scale off-pad effects in order to fully understand the landscape-level effects of energy extraction.

The effects of well pads beyond the well pad footprint may also have cascading consequences. For example, Rogers and Koper (2017) reported that vegetation around shallow natural gas pumps in Alberta was shorter than the surrounding grassland, likely due to construction effects on soil and increased cattle grazing next to fencing. Research has also documented increased bare ground cover, decreased litter, decreased forb cover, and decreased diversity of native vegetation around oil and gas wells compared to reference sites (Nasen et al. 2011). Such changes in vegetation have the potential to then affect soil characteristics, as decreased vegetation cover may elevate soil surface temperatures and decrease soil moisture levels (Odriozola et al. 2014). Alterations to vegetation cover and structure as well as topsoil additions around well pads during construction may also increase erosion and alter arthropod communities (Dennis et al. 1998, Loch 2000, Matherne 2006). Collectively these changes may create areas of reduced arthropod and vegetation forage for grassland birds and ungulates, as well as reducing nesting cover and carbon sequestration potential around well pads. Studies of gas development at broad spatial scales often focus on direct footprint impacts to landscapes (i.e., direct loss of vegetative cover, primary productivity, carbon sequestration potential, etc.; Allred et al. 2015), neglecting to address the potential for enlarged impact footprints beyond the denuded well pad.

In order to better understand how biotic and environmental factors change with proximity to oil and gas drilling pads, we investigated potential changes in the arthropod community, changes in vegetation characteristics, and changes in soil characteristics with distance away from well pads to assess small-scale effects extending beyond the boundaries of well pads. Specifically, we studied: 1.) the effects of distance from well pads on arthropod biomass, abundance, and community composition, 2.) the effects of distance from well pads on vegetation cover classes, litter depth, and vegetation height, and 3.) the effects of distance from well pads on soil moisture and soil surface temperature at four distances from active well pads.

METHODS

Study Area

We conducted this study at Packsaddle Wildlife Management Area (PWMA) between May and October, 2015 and 2016. PWMA encompasses 7,956 ha and is located approximately 27 km south of Arnett, Oklahoma. The study site is largely comprised of shinnery oak (*Quercus havardii*) and mixed grass prairie. Other dominant shrubs include Chickasaw plum (*Prunus angustifolia*), three-leaf sumac (*Rhus trilobata*), and sand sagebrush (*Artimisia filifolia*). Forb cover is diverse, and abundant species include prairie sunflower (*Heliolanthus petiolaris*), erect dayflower (*Commelina erecta*), firewheel (*Gaillardia pulchella*), Texas croton (*Croton texensis*), and western ragweed (*Ambrosia psilostachaya*). Little bluestem (*Schizachyrium scoparium*), big bluestem (*Andropogon*

hallii), Indiangrass (*Sorghastrum nutans*), switchgrass (*Panicum virgatum*), side-oats grama (*Bouteloua curtipendula*) and hairy grama (*Boutelou hirsuta*) constitute the dominant grass species at the study site (DeMaso et al. 1997, Townsend 1999). Dominant soils at PWMA include Nobscot fine sand, sandy Nobscot-Brownfield, sandy Pratt-Carwile, and loamy Quinland-Woodward (Dunkin et al. 2009). Elevation on the WMA ranges from 579 m to 762 m above sea level (Townsend et al. 1999). The 30-year average rainfall levels for the area are 659.89 mm annually (Oklahoma Mesonet 2016).
PWMA has been subject to substantial energy development, and there were 70 active wells on the property during the study. Fifty of the wells were installed after 2002. This recent development has also created a network of access roads expanding from 31 km (0.39km/km^2) in 2002 (Dunkin et al. 2009) to 97 km (1.21km/km^2) in 2015. Much of the WMA was inaccessible by road prior to recent energy development.

Study Design

To assess the effects of oil and gas pads on soil characteristics, arthropods, and vegetation, we set up 100 m transects running perpendicular to the edge of active oil and gas pads. Transects were chosen each year by randomly selecting the well pad and cardinal direction of 30 sampling transects from a pool until 30 unique locations were drawn for each year. Multiple transects were allowed on the same well pad, but no more than two transects per well pad were permitted in a given year. We did not set up transects on the same side of the well pad as the access road for the pad, and transects were also not situated where a road or other well pad was within 200 m of the target pad. Based on the surrounding topography, we did not establish transects on well pads that were likely to be submerged during the study period. Sampling locations were 0.5 m inside the edge of the well pad (i.e., on the pad), at 1 m from the edge, at 10 m from the edge, and at 100 m from the edge. These distances were determined to include a distance where well pad conditions are likely fully governing measured processes (on the pad) and a distance at which environmental variables are likely no longer affected by the presence of the nearest well pad (100 m).

Arthropod Sampling

Arthropod community changes are used as an indicator of disturbance due to their rapid population response to disturbance, high rates of endemism, and large breadth of niche occupancy and specialization. Past demonstrated changes to arthropod functional groups across gradients of grassland anthropogenic use intensity include selective pressure toward smaller body size, enhanced dispersal ability, and greater degree of specialization (Simons et al. 2016).

We collected arthropods at four distances (-0.5 m, 1 m, 10 m, 100 m) from active well pad edges. We used a combination of pitfall traps and sweep nets to obtain a more complete representation of arthropod abundance and diversity (Spafford and Lortie 2013).

At each distance, three pitfall traps (473 mL Solo cups, 9.55 cm diameter at mouth) were aligned perpendicular to the pad edge with 3 m spacing. Pitfall traps consisted of a double layer of solo cups that were set flush with the ground and filled with 5 cm of 50% ethyl alcohol. Prior to sampling, we installed pitfall traps and covered them for a week to reduce the “digging in” effect, a period of inflated capture rates immediately following trap installation (Greenslade 1973). Pitfall sampling occurred during two 24-hour sampling periods from 15 June to 1 September. We pooled pitfall traps within distances in each transect. Traps were covered between sampling periods. Following collection, specimens were transferred to sampling jars containing 70% ethyl alcohol and stored for later sorting and identification.

We conducted sweep net sampling (38 cm diameter net) from 15 June and 1 September between 1000 and 1600 hr at the same sampling distances as pitfall trapping. To reduce sampling bias and variability, all samples were collected by the same person. At each distance, sweep netting occurred along a 20 pace (16 m) transect parallel to the distance gradient transect. At each pace, one back-and-forth sweep of the net was conducted. Between sweeps, the net was twisted 180° to avoid escape. After finishing each transect, all arthropods were removed and transferred to 3.8 L freezer bags (Buffington and Redak 1998, Doxon et al. 2011a). Following collection, invertebrate specimens were frozen for later sorting and identification.

We identified invertebrates to order and morphospecies (i.e., classifying arthropods into unique categories by nonspecialists) to determine invertebrate richness and diversity. This method is comparable to specialist classification for diversity and richness metrics (Oliver and Beattie 1996). Shannon-Weiner Index, invertebrate richness, and dry biomass were determined for each sample (Doxon et al. 2011a, Doxon et al. 2011b, Krebs 1989). To determine biomass, we dried invertebrates at 75° C for 24 hours and subsequently weighed them to ± 0.0001 g.

Vegetation Sampling

Litter depth, plant community composition, and maximum plant height were measured to determine variation in vegetation with distance from well pads. At each plot, a modified Daubenmire frame (Daubenmire 1959) with dimensions of 0.5 m by 0.5 m was centered 0.5 m to the right of the transect at each sampling point (i.e., -0.5 m, 1 m, 10 m, and 100 m in 2015 and -0.5 m, 1 m, and every 10 m up to 100 m in 2016). During the 2016 field season, additional vegetation sampling distances were added to create a clearer picture of the extent of the effects of oil and gas pads on vegetation communities. Litter depth was measured at the four corners and at the center of each Daubenmire frame. Litter depth measurements did not include standing dead vegetation. Maximum vegetation height was defined as the droop height of the tallest live vegetation rooted within the frame. We visually estimated cover for warm season grasses, cool season grasses, forbs, woody vegetation, legumes, sedges, bare ground, and litter using the following cover classes: 0%-5%, 5%-25%, 25%-50%, 50%-75%, 75%-95%, and 95%-100%. For data analyses, we used the midpoint values for each cover class (Daubenmire 1959).

Soil Characteristics

We measured soil temperature using a Forward-Looking Infrared gun (FLIR systems, Wilsonville, OR). At each sampling distance, a Daubenmire frame was placed with the center located 0.5 m to the right of the central pitfall trap. Four temperature readings were taken from 6 cm outside the four corners of the Daubenmire frame at a height of 3 cm from the soil surface. Vegetation cover was parted to expose bare soil to accurately record soil surface temperature. We recorded temperatures from 1100 to 1700 h on days with no cloud cover to capture maximium daily temperatures and insolation

(Limb et al. 2009). These data were compared to ambient air temperatures collected by an on-site weather station (Weatherhawk, Logan UT) for the same period to standardize results across a range of daytime temperatures, and results were expressed via the following equation:

$$T_d = T_s - T_a$$

Where T_d is the temperature differential and T_s and T_a are the soil surface temperature and ambient air temperatures, respectively. Similar to vegetation sampling in 2016, we included additional sampling distances to record soil surface temperatures.

To measure soil moisture, we conducted soil sampling at the -0.5 m, 1 m, 10 m, and 100 m from active well pads. Cores were drilled to a depth of 30 cm using an 8.25 cm diameter auger. At each sampling distance, the three cores were emptied into an 18.9 L (5 gal) bucket and thoroughly mixed with a spade. Approximately 800 g of soil were subsampled from the bucket and transferred to a 3.8 L (1 gal) freezer bag. Samples were weighed in the field (± 0.01g) to determine initial moisture content. Samples were then dried at 105° C for 24 hours and reweighed (Mullins and Smith 2001). We then determined thermogravimetric water content (*w*) based on the following equation:

$$w = \frac{Mass\ of\ Wet\ Soil - Mass\ of\ Dry\ Soil}{Mass\ of\ Dry\ Soil}$$ (Mullins and Smith 2001).

Data Analysis

We used multivariate and univariate analyses to describe the relationship of invertebrates to distance from well pads and vegetation and soil characteristics. Nonmetric multidimensional scaling (NMDS) was used to evaluate patterns of invertebrate communities relative to distance from well pads as well as along gradients in soil and vegetation characteristics. Analysis was performed using the vegan package in Program R (ver. 3.1.1, R Foundation for Statistical Computing, Vienna, Austria) (Doxon et al. 2011b, Kruskal 1964). Sampling distance and soil and vegetation variables were used to explain community composition in ordinations and their significances were assessed using permutational analysis of variance (PERMANOVA, McArdle, and Anderson 2001). Biomass and abundance were pooled by order. Bray-Curtis dissimilarity measures were used in NMDS analysis (Doxon et al. 2011b). Each ordination was performed in 3 dimensions. Abundance measures were square root and Wisconsin-transformed while biomass measures were untransformed following recommendations based on species maxima (Ellison 2011).

Following multivariate analysis, univariate associations between distances from well pads and arthropod biomasses, arthropod orders, arthropod morphospecies richness, and Shannon-Weiner diversity were investigated using analysis of variance (ANOVA). Abundance data were square root transformed to better approximate a normal distribution. Following significant ANOVAS, we used Tukey’s honest significant difference test to distinguish differences among distances. Arthropod data were pooled within years between sampling periods. In total, we collected representatives of 24 arthropod orders. For analysis, we focused on the 8 most abundant orders: Araneae, Coleoptera, Diptera, Hemiptera, Hymenoptera, Orthoptera, and Lepidoptera. Because analysis of our first year of data did not reveal significant relationships between pitfall sample arthropod communities and sampling distance, we did not pursue further analyses of the pitfall data. This may be due to tighter associations between vegetation and arthropods than soil characteristics and arthropods. We decided to focus on sweep net samples, as we felt that these data better reflected responses to changes in the surrounding landscape due to well pads.

We conducted a multiple analysis of variance (MANOVA) to test for differences in soil and vegetation between sampling distances. Soil temperature, soil moisture, maximum vegetation height, litter depth, and percent cover for warm season grasses, forbs, shrubs, litter, and bare ground were analyzed for a total of 64 transects in 2015 and 2016. A Hotelling-Lawley test showed a significant difference in groups by distance ($F_{3,252} = 17.36$ $P < 0.001$). Individual ANOVA tests were subsequently run in order to assess post-hoc differences between sampling distances via Tukey's Honest Significant Distance Test. We did not include cool season grass cover, sedge cover, and legume cover in vegetation analysis due to minimal variability in the dataset.

RESULTS

Arthropod Community

Correlation analysis from NMDS ordination indicates that arthropod community abundance (Figure 1) was significantly correlated with distance from oil pad ($P = 0.001$, $r^2 = 0.06$), as was arthropod community biomass ($P = 0.001$, $r^2= 0.30$, Figure 1). PERMANOVA results indicated that vegetation height ($P < 0.01$, $r^2 = 0.08$), litter depth ($P < 0.01$, $r^2 = 0.06$), and bare ground ($P < 0.01$, $r^2 = 0.06$) were also significantly correlated with arthropod abundance (Table 1). NMDS axis 1 was highly positively correlated (≥ 0.7) with maximum vegetation height, and grass cover and highly negatively correlated with bare ground. NMDS axis 2 was highly negatively correlated with litter depth and forb cover (Table 1). Distance separation most closely reflected changes in Coleoptera, Orthoptera, Hemiptera, and total abundance, and was equally explained by axis 1 and axis 2 (Figure 1). PERMANOVA analysis of environmental covariates and arthropod order biomass revealed significant correlation with maximum vegetation height ($P < 0.01$, $r^2 = 0.21$), litter depth ($P < 0.01$, $r^2 = 0.14$), grass cover ($P < 0.01$, $r^2 = 0.07$), forb cover ($P < 0.01$, $r^2 = 0.07$), and bare ground ($P < 0.01$, $r^2 = 0.18$; Table 2). NMDS axis 1 was highly positively correlated with maximum vegetation height and forb cover. NMDS axis 2 was highly negatively correlated with bare ground cover (Table 2). NMDS axis 1 reflected the majority of the variance in site rank biomass, and reflected increases in the biomasses of Diptera, Hymenoptera, Orthoptera, Araneae, and total biomass with distance from well pad (Figure 2).

The presence of oil and gas pads decreased the abundance of all orders studied (all orders: $F_{3,252} \geq 4.03$, $P < 0.01$). For both Araneae and Hymenoptera, abundances were highest at the 1, 10, 100 m distances and lowest at the -0.5 m distance (Figure 3). Hymenoptera abundance was over 4X greater at the 100 m distance than the on-pad distance (Figure 3). Coleoptera and Hemiptera abundance were lowest at the on-pad distance and significantly higher at the 10 m than the 1 m distance. The 100 m abundances were not significantly different from the 1 m and the 10 m distance (Figure 3). Hemiptera and Coleoptera abundance were 6X and 5X greater at the 10 m distance than the on-pad distance respectively. Diptera and Lepidoptera abundance was significantly lower in the 1 m sample than the 10 m and 100 m sample (Figure 3). Orthoptera abundance and total abundance were highest in the 10 m and 100 m samples and lowest in the -0.5 m sample (Figure 3). Orthoptera abundance was nearly 10X greater at the 10 m distance than the on-pad distance, and total abundance was nearly 5X greater at the 10 m distance than the on-pad distance (Figure 3).

Arthropod biomass also was impacted by the presence of well pads for all orders studied (all orders: $F_{3,252} \geq 3.10$, $P < 0.03$). Araneae and Hemiptera biomass were higher at the 1 m distance than the on-pad distance, and was highest at the 10 m and 100 m distances (Figure 4). Biomasses of Araneae and Hemiptera were respectively nearly 6X and 7X in the 10 m and 100 m distances than the on-pad distance. Orthoptera biomass and total biomass were higher at the 10 m and 100 m distances than the on-pad and 1 m distance (Figure 4). Biomasses of these orders were respectively 15X and 9X higher at the 10 m distance than the on-pad distance (Figure 4). Coleoptera biomass was higher in the 1 m distance than the on-pad distance, and highest in the 10 m and 100 m distances (Figure 4). Coleoptera biomass varied three-fold between the on-pad and 10 m distances (Figure 4). Diptera biomass was highest at the 100 m sampling distances (Figure 4). Hymenoptera biomass was lowest at the -0.5 m distance, with no significant differences between the 1 m, 10 m, and 100 m distances (Figure 4). Biomass was almost 7X higher at the 1 m distance than the on-pad distance. Lepidoptera was higher at the 10 m distance than the on-pad and 1 m distance (Figure 4).

Shannon-Weiner diversity and morphospecies richness were significantly impacted by well pads (Diversity: $F_{3, 252} = 23.07$, $p < 0.01$; Richness: $F_{3,252} = 12.6$, $p < 0.01$). The on-pad distance and 1 m distance had significantly lower morphospecies diversity than the 10 m and 100 m distance and morphospecies richness was higher in the 10 m sampling distance than the on-pad and 1 m distances, with almost 4X higher richness in the 10 m distance than the -0.5 m distance (Figure 5).

Environmental Covariates

In general, effects on vegetation extended farther from the well pad than effects on arthropods. All vegetation characteristics differed between sampling distances ($F_{3, 252} \geq 6.07$, $P < 0.01$). Maximum vegetation height was lowest at the on-pad distance and was significantly higher at each successive distance from the pad. Maximum vegetation height at the 100 m distance was over 6X higher at the 100 m distance than the on-pad distance. (Figure 6). Bare ground followed the opposite trend, decreasing significantly with each successive distance away from the pad by a factor of 4X (Figure 6). Litter depth and litter cover were lower in the on-pad and 1 m distances than the 10 m and 100 m distances (Figure 6). Grass cover was the lowest at the on-pad distance, and grass cover was higher at the 10 m distance and higher still at the 100 m distance (4X higher at 100 m distance than -0.5 m, Figure 6). Forb cover was highest at the 10 m distance, with nonsignificant differences between the other three sampling distances (Figure 6). Forb cover was over 5X higher at the on-pad distance than the 10 m distance. Cover of shrubs was higher at the 100 m sampling distance than the other three distances (Figure 6).

Temperature differences were impacted by well pads $F_{3, 252} = 7.24$, $p < 0.01$, and on-pad and 1 m sampling distances had significantly higher soil temperature differentials (i.e., soil temperatures hotter than ambient air temperatures) than the 100 m distance (Figure 7). The 10 m distance was intermediate and not significantly different from either group. Soil moisture did not differ significantly among groups ($F_{3, 252} = 0.54$, $p = 0.65$; Figure 7).

DISCUSSION

Our results indicate that active well pads have an influence that extends beyond the existing physical footprint of the infrastructure. In general, the presence of an oil and gas drilling pad does not affect arthropod and soil characteristics at distances greater than 1 m from the pad, but vegetation effects may extend significantly beyond the well footprint. Maximum vegetation height, grass cover, shrub cover, and bare ground cover were impacted at distances in excess of 10 m away from well pads (Table 3, Figure 6). Forb cover was highest at intermediate distances from well pads, and additional sampling distances revealed a peak in cover at 20 m from the pad (Table 3, Figure 6, Figure 8). Vegetation height was impacted close to well pads, reinforcing the decreased vegetation height around shallow gas wells documented in Rogers et al. (2017).

Greater forb cover at intermediate distances from well pads compared to the surrounding landscape may be due to increased erosion and disturbance which inhibits shrub establishment. A lack of arthropod impacts coupled with enhanced forb cover and bare ground cover may enhance foraging opportunities for insectivorous birds, (Doxon and Carroll 2007, McIntyre et al. 2003). Research on insectivorous grassland birds has revealed a strong preference for sparsely vegetated areas for foraging, despite decreased arthropod abundances in these areas compared to nearby patches with enhanced vegetative cover (Martinez et al. 2010). Though not statistically significant, maxima in Araneae, Coleoptera, Diptera, Hemiptera, Lepidoptera, Orthoptera, and total abundance and Araneae, Coleoptera, Lepidoptera, Orthoptera, and total biomass mirrored the statistically significant 10 m forb cover maximum (Figure 3, Figure 4). This may signify small benefits to arthropods from increased coverage of forbs at intermediate distances from well pads (Hill 1985).

Our arthropod sampling may have failed to document community changes extending more than 1 m from the pad for several reasons. Arthropods may be responding to resources at a smaller scale than that at which we conducted sampling. For example, arthropod community analysis has revealed a greater response due to the presence of individual shrubs than due to landscape-level gradients in land use (Liu et al. 2013). It is possible that by sampling in a perpendicular transect (sweep nets) or by averaging pitfall traps at each sampling distance, we obscured associations between arthropods and vegetation. Also, arthropod vagility is directly impacted by vegetation structure (Crist et al. 1992), and we may have been effectively limiting the effective sampling radius of pitfall traps of far distances from well pads as compared to nearer distances due to increases in vegetative cover. Similarly, sweep net sampling effectiveness may have been reduced in plots far from well pads due to higher vegetation structural complexity, as sweep nets tend to only sample the outer surfaces of vegetation (Doxon et al. 2011a).

Vegetation cover effects were more extensive, and generally cover was lower in excess of 1 m from the pad in all cases except litter. Decreased shrub cover at distances in excess of 10 m from well pads may have significant effects on soil chemistry (Ravi et al. 2007, Stavi et al. 2009). In grazed semiarid rangelands, spatial patterning of nutrients such as soil organic carbon and nitrogen results in “islands of fertility” under shrub cover (Minnick and Alward 2015, Morris et al. 2013). During well pad creation, well pads undergo vertical and horizontal soil movement, and this has been shown to result in an absence of mineral and organic matter patterning that may take in excess of 50 years to redevelop (Minnick and Alward 2015). Additionally, shrub islands have been shown to increase soil porosity, increase soil infiltration, decrease erosion rates, and contribute to soil stability (Stavi et al. 2009).

The documented decreases in cover of grasses and shrubs surrounding well pads have several consequences. Warm season grass cover increased with distance from well pad at least until 10 m from the pad. This indicates that forage lost to oil and gas wells in North America could be larger than the 10 Tg of dry biomass or 5 million animal unit months (AUM) calculated in direct footprint projections (Allred et al. 2015). Reduced grass cover also decreases the amount of available nesting cover for grassland birds, and more sparse grass and shrub coverage coupled with shorter vegetation may make nests more visible to predators (Gregg et al. 1994, Lusk et al. 2006).

Though we found decreased cover of vegetation around well pads, these changes may align well with management objectives in disturbance-driven systems, such as PWMA. Prescribed fire is used at the site to promote heterogeneity and limit woody encroachment, particularly of eastern red cedar. Our results indicate that areas surrounding oil and gas drilling pads may resist woody encroachment and provide structural heterogeneity by preventing a shinnery oak monoculture without the need for continual treatment.

We did not find significant differences in soil moisture between sampling distances, corroborating the results of Nasen et al. (2011). However, soil moisture standard errors did decrease with distance from well pad indicating potential instability in soil moisture caused by well pads. Vegetative cover and composition are known to be drivers of soil moisture through the reduction of Aeolian forces and through precipitation partitioning (Legates et al. 2011). Plant-soil feedback has also been implicated in moisture partitioning in semi-arid grasslands, which is important for diversity in these ecosystems (Chamizo et al. 2013). However, these relationships are unproven at larger spatial scales due to spatial heterogeneity in plant community composition, precipitation, and soil (Wetzel and Chang 1987). In addition to affecting local soil moisture, decreased vegetative cover can also increase erosion in grasslands (Loch 2000). Though we did not evaluate sediment transport around well pads, rill and gully erosion during the study period destroyed several pitfall trap installations, and sediment flow frequently buried pitfall traps following rain events. Rates of erosion surrounding well pads have been measured at 11X that of similarly-sloped hills, and 4X that of similarly-sloped hills fragmented by roads (Matherne 2006).

Now that we have documented external effects of oil and gas drilling pads on vegetation, it is important for future studies to document temporal trends in vegetation relative to well age. In this study, we only considered active well pads to include effects to arthropods and vegetation from pad vehicle traffic and site maintenance. Though we sampled active wells that ranged from 1 to 35 years in age, the majority of active wells were recently constructed. Including inactive well pads would allow us to compare vegetation and arthropod composition across a gradient of well age. Additionally, our linear sampling method did not permit us to study self-organizing spatial patterning of vegetation and soil nutrients, which has been shown to be an important aspect of vegetation recovery following disturbance (Johnstone and Kokelj 2008, Minnick and Alward 2015, Morris et al. 2013). Documenting the changes in patterning through time has potential to serve as an indicator of successful reclamation efforts in arid and semiarid grasslands and shrublands, and temporal assessment of well pads effects would allow land managers to develop dynamic tools to document site-wide energy impacts to resources through time to achieve goals such as increasing community resiliency or preventing invasive plant or shrub invasion (Sankey et al. 2012, Stavi et al. 2009).

Though newer oil and gas technologies have allowed the rapid expansion of energy development in western landscapes, these same technologies allow unprecedented landscape planning opportunities. Horizontal boreholes allow wide-ranging access of fossil fuels up to 3.2 km away from the surface well (Thompson et al. 2015). This means that wells can be clustered together to minimize vegetation impacts while still accessing resources beneath the intact landscape. The advent of oil and gas site planning impact mitigation software (Milt et al. 2016) allows land managers to quantify the effects of proposed oil and gas development and to test the effects of various spatial arrangements of well pads. Conversely, areas around well pads may have vegetation structure consistent with management objective in disturbance-driven systems. Our research highlights the feasibility of including community-specific vegetation effects into site planning and mitigation activities.

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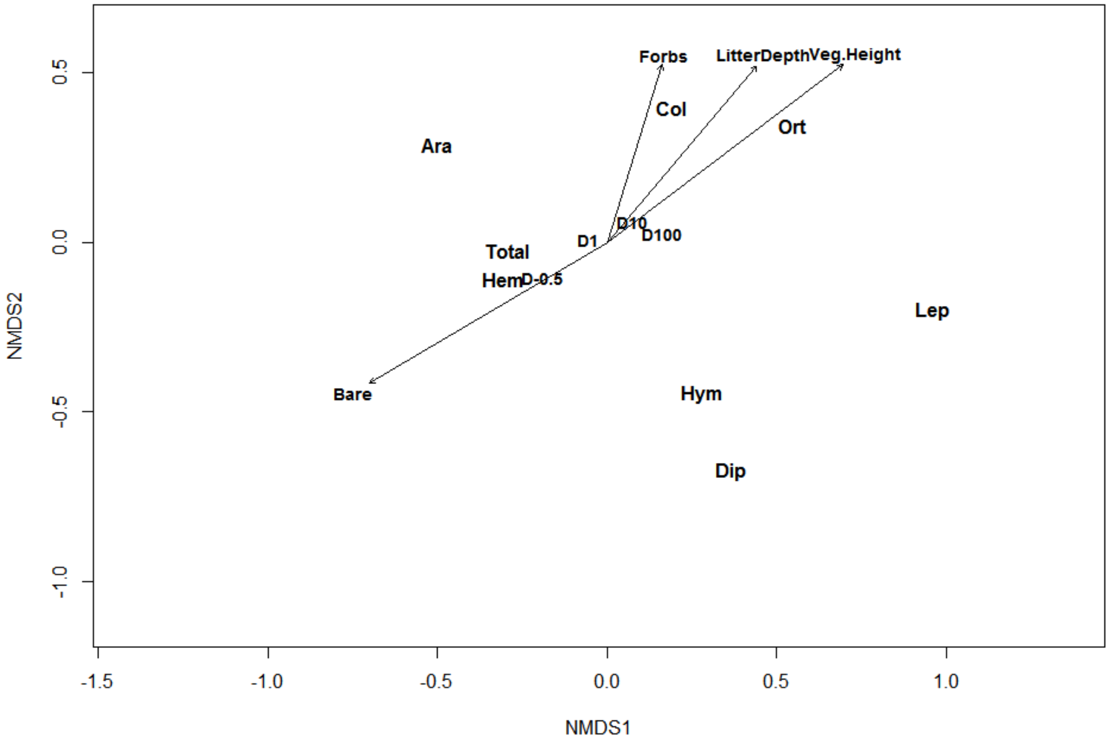

Figure 1. Nonmetric multidimensional scaling (NMDS) ordination plot for abundances of seven invertebrate orders sampled by sweep netting at -0.5 m, 1 m, 10 m, and 100 m from active oil and gas pads in 2015 and 2016 at Packsaddle Wildlife Management Area in western Oklahoma, United States. Only environmental variables that are significant at $P < 0.05$ are presented. $K = 3$, Stress = 0.13. Abbreviations are as follows: Hym: Hymenoptera, Dip: Diptera, Lep: Lepidoptera, Ort: Orthoptera, Col: Coleoptera, Ara: Araneae, Hem: Hemiptera, D-0.5: -0.5 m distance, D1: 1 m distance, D10: 10 m distance, D100: 100 m distance.

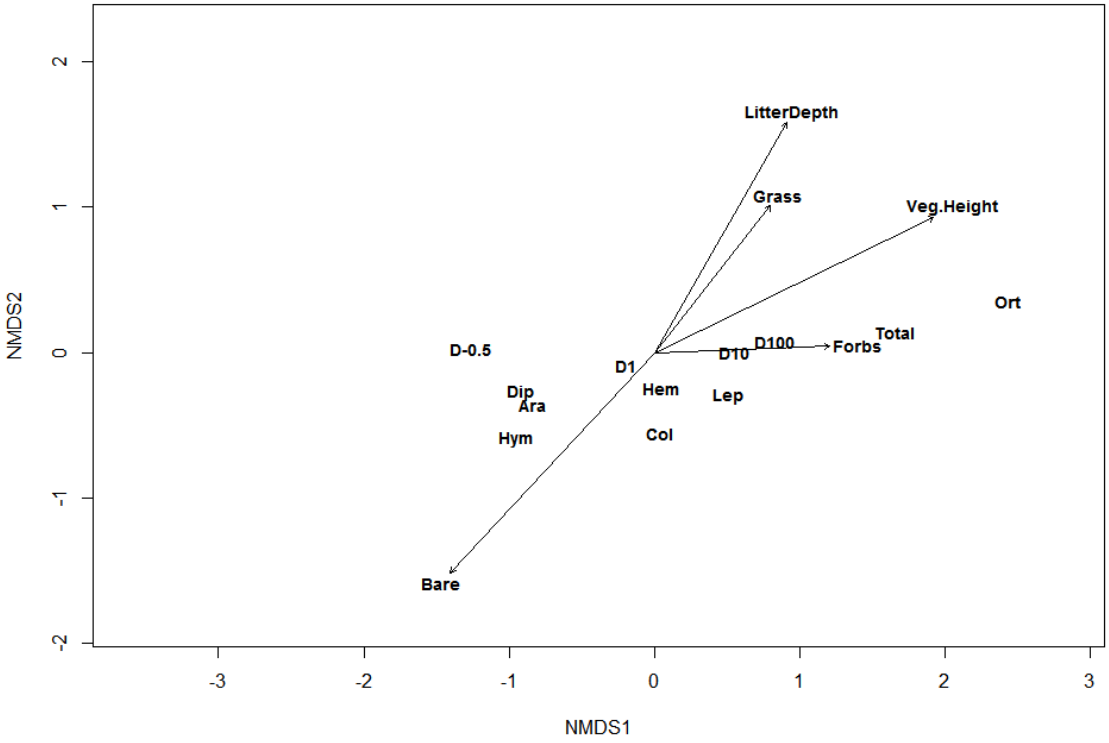

Figure 2. Nonmetric multidimensional scaling (NMDS) ordination plot for biomasses of seven invertebrate orders (Araneae, Coleoptera, Hemiptera, Diptera, Orthoptera, Hymenoptera, Lepidoptera, and total abundance) sampled by sweep netting at -0.5 m, 1 m, 10 m, and 100 m from active oil and gas pads in 2015 and 2016 at Packsaddle Wildlife Management Area in western Oklahoma, United States. Only environmental variables that are significant at $P < 0.05$ are presented. $K = 3$, Stress= 0.05. Abbreviations are as follows: Hym: Hymenoptera, Dip: Diptera, Lep: Lepidoptera, Ort: Orthoptera, Col: Coleoptera, Ara: Araneae, Hem: Hemiptera, D-0.5: -0.5 m distance, D1: 1 m distance, D10: 10 m distance, D100: 100 m distance.

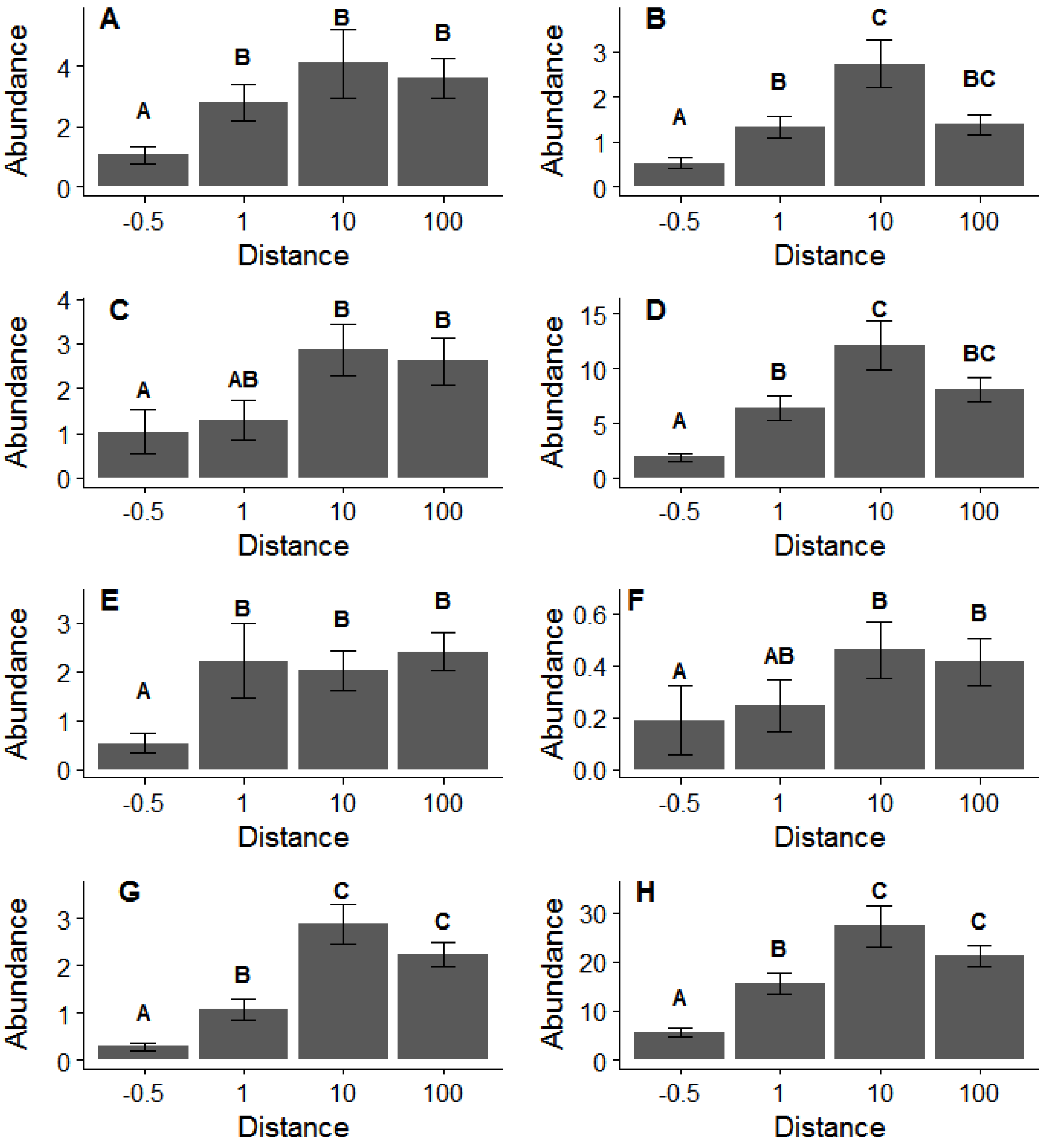

Figure 3. Average abundance/sample of invertebrate taxa (order-level) collected by sweep netting at -0.5 m, 1 m, 10 m, and 100 m from well pads at Packsaddle Wildlife Management Area in western Oklahoma, United States during 2015 and 2016: **A,** Araneae. **B,** Coleoptera. **C,** Diptera. **D,** Hemiptera. **E,** Hymenoptera. **F,** Lepidoptera. **G,** Orthoptera. **H,** Total Abundance. Error bars represent ± 1 standard error. Distances with the same letter are not significantly different ($P \geq 0.05$).

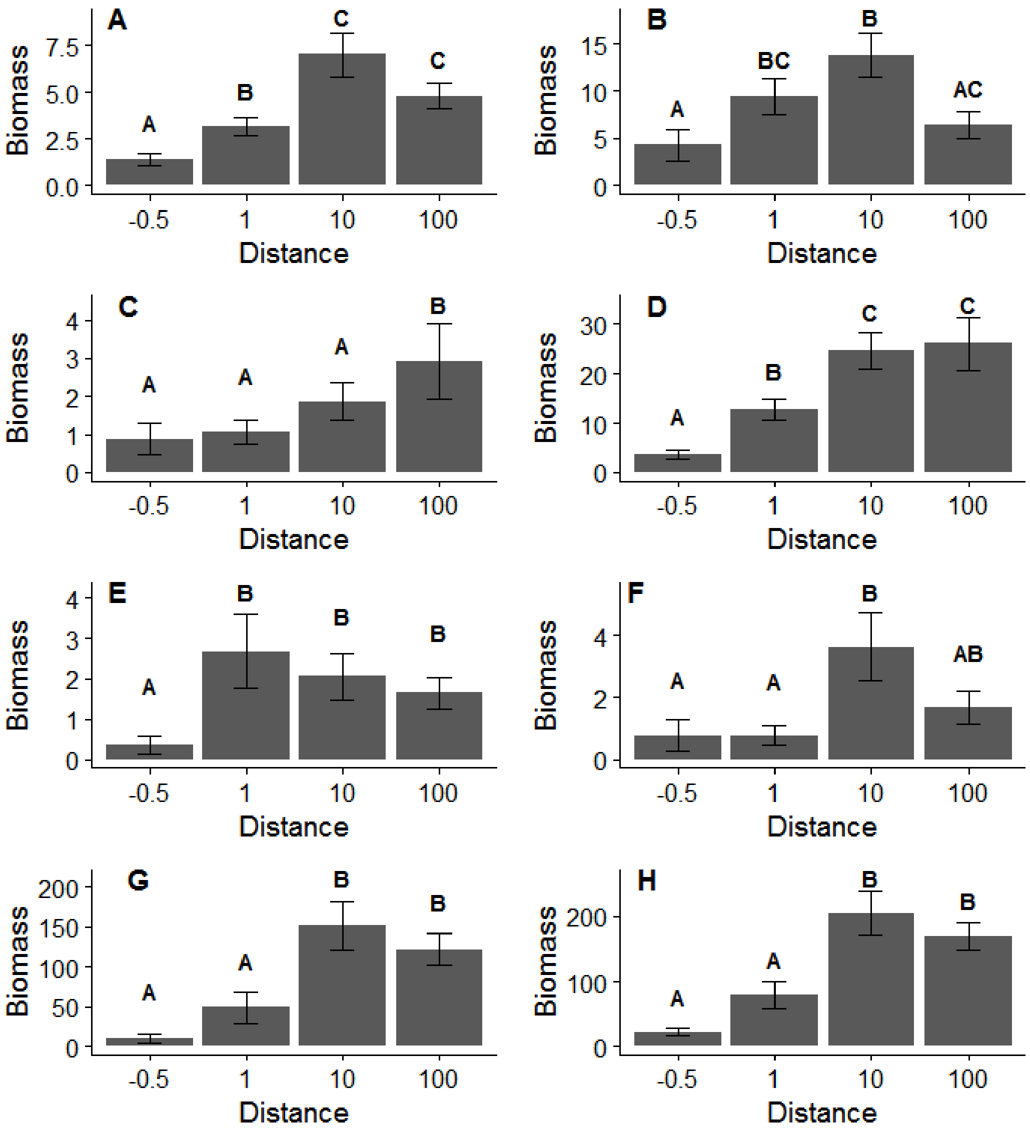

Figure 4. Average biomass/sample (mg) of invertebrate taxa (order-level) collected by sweep netting at -0.5 m, 1 m, 10 m, and 100 m from well pads at Packsaddle Wildlife Management Area in western Oklahoma, United States during 2015 and 2016: **A,** Araneae. **B,** Coleoptera. **C,** Diptera. **D,** Hemiptera. **E,** Hymenoptera. **F,** Lepidoptera. **G,** Orthoptera. **H,** Total Biomass. Error bars represent ± 1 standard error. Distances with the same letter are not significantly different ($P \geq 0.5$).

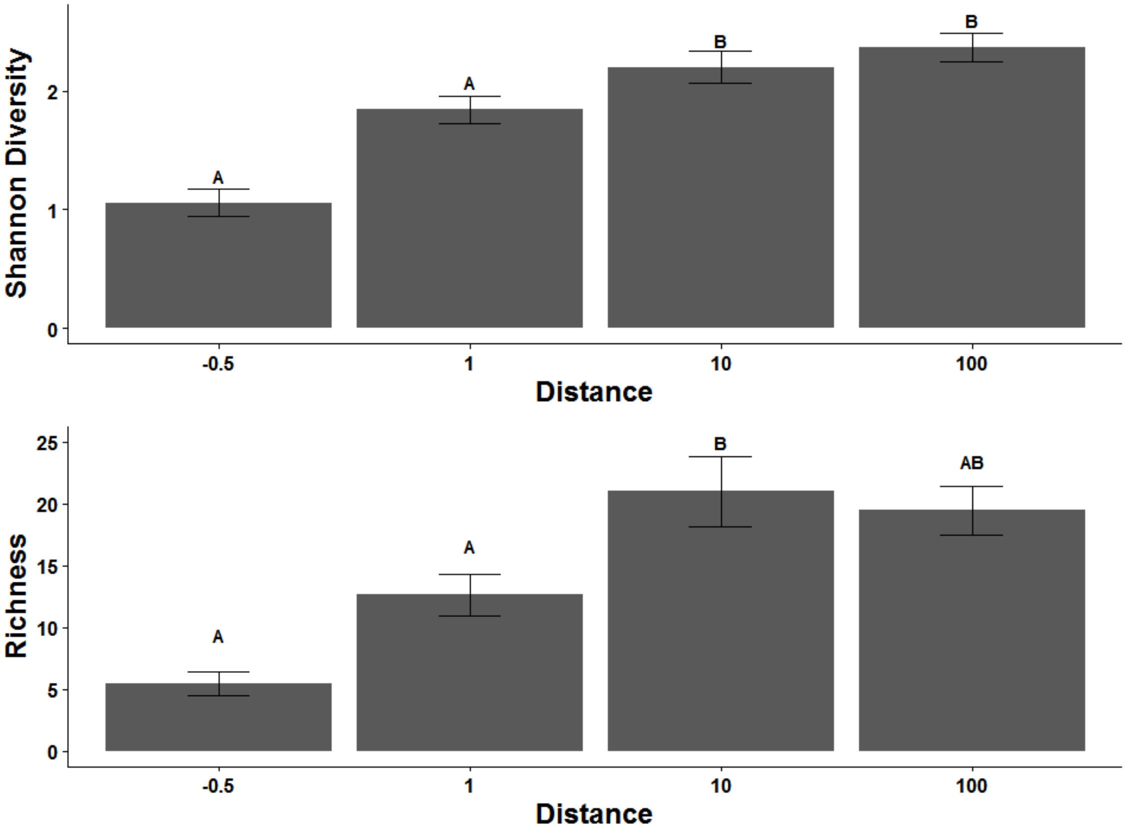

Figure 5. Shannon Diversity and richness of arthropod community at -0.5 m, 1 m, 10 m, and 100 m from well pads at Packsaddle Wildlife Management Area in western Oklahoma, United States, 2015 and 2016. Error bars represent ± 1 standard error. Distances with the same letter are not significantly different ($p \geq 0.05$).

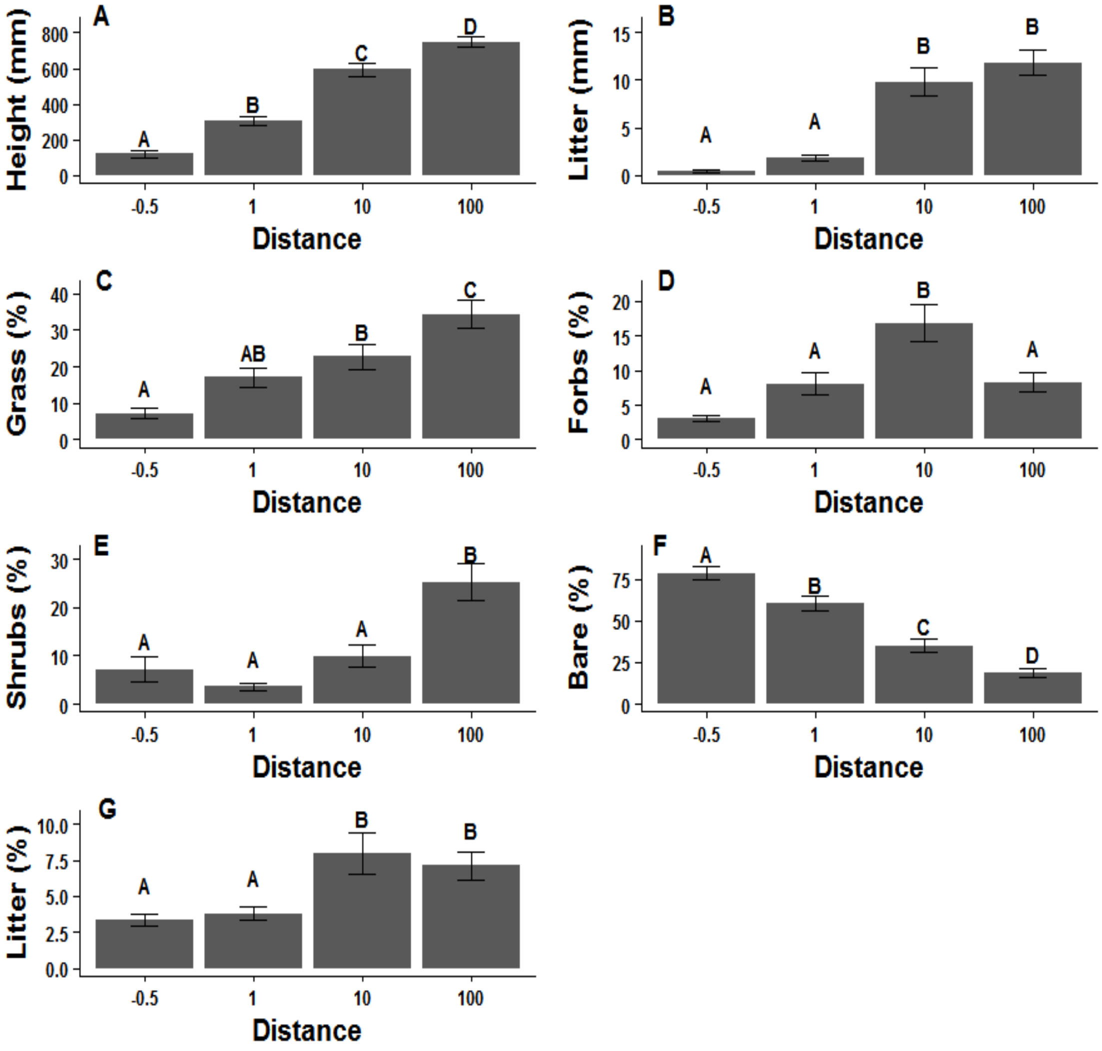

Figure 6. Vegetation and soil characteristics at -0.5 m, 1 m, 10 m, and 100 m from well pads at Packsaddle Wildlife Management Area in western Oklahoma, United States during 2015 and 2016: **A**, Maximum Vegetation Height (mm). **B**, Litter Depth (mm). **C**, Warm Season Grass % Cover. **D**, Forb % Cover. **E**, Shrub % Cover. **F**, Bare Ground % Cover. **G**, Litter % Cover. Error bars represent ± 1 standard error. Distances with the same letter are not significantly different ($p \geq 0.05$).

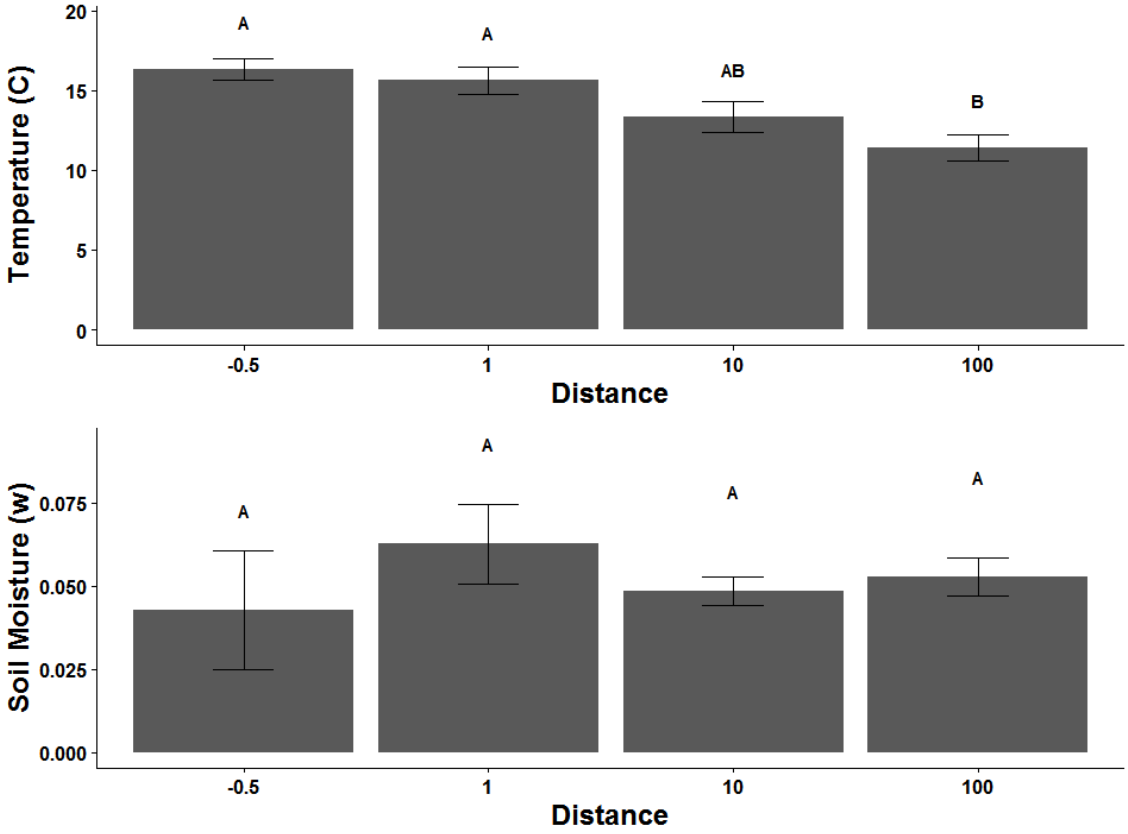

Figure 7. Soil characteristics at -0.5 m, 1 m, 10 m, and 100 m from well pads at Packsaddle Wildlife Management Area in western Oklahoma, United States in 2015 and 2016: **Top,** Ambient Air Temperature – Soil Surface Temperature (°C). **Bottom,** Thermogravimetric Soil Moisture (*w*). Error bars represent ± 1 standard error. Distances with the same letter are not significantly different.

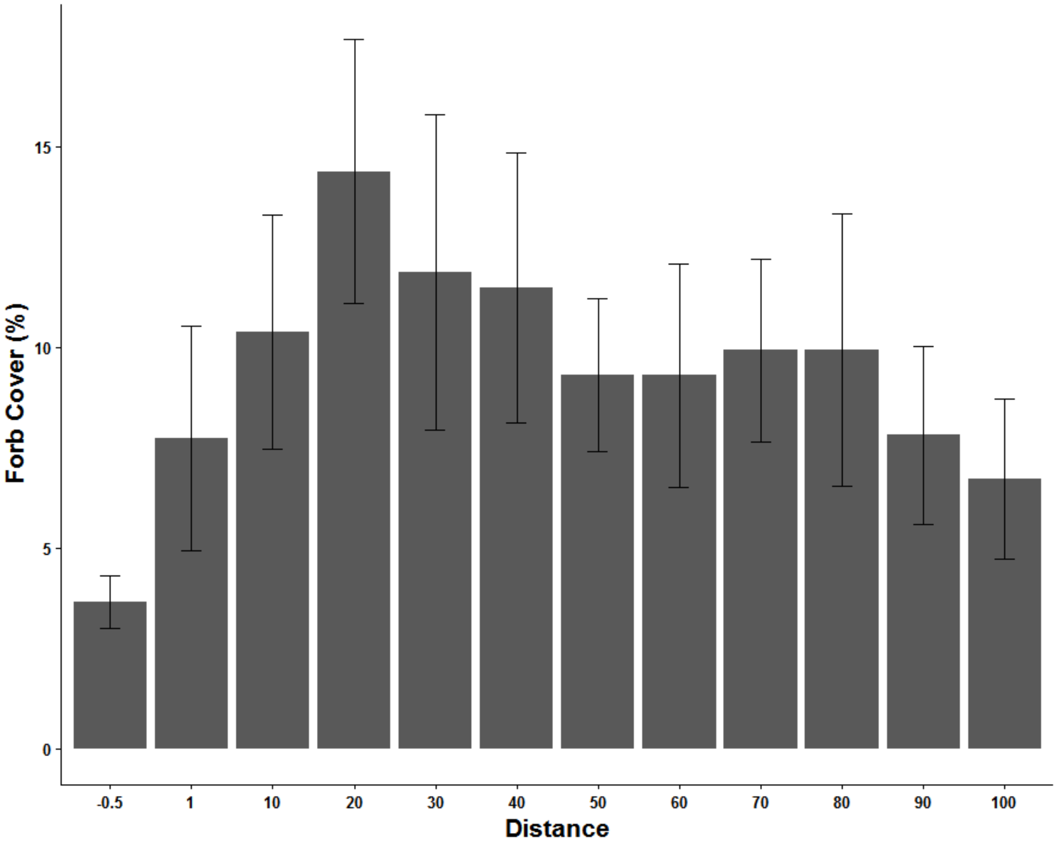

Figure 8. Forb cover at -0.5 m, 1 m, 10 m, 20 m, 30 m, 40 m, 50 m, 60 m, 70 m, 80 m, 90 m, 100 m from active well pads at Packsaddle Wildlife Management Area in western Oklahoma, United States in 2016. Error bars represent ± 1 standard error.

Table 1. Correlations of environmental variables and vegetation characteristics with each nonmetric dimensional scaling (NMDS) axis and associated significance for sweep net arthropod order abundances at -0.5 m, 1 m, 10 m, and 100 m from active oil and gas pads in 2015 and 2016 at Packsaddle Wildlife Management Area in western Oklahoma, United States. * indicates significance at α = 0.05. Axis values for transect distances (a factor variable) indicate centroid locations for each level.

	NMDS1	**NMDS2**	**r^2**	**P value**
Maximum Vegetation Height	0.80	-0.60	0.09	< 0.01*
Litter Depth	0.65	-0.76	0.06	< 0.01*
Warm Season Grass Cover	0.87	-0.48	0.01	0.21
Forb Cover	0.29	-0.96	0.04	0.01*
Shrub Cover	0.99	-0.03	0.01	0.28
Bare Ground	-0.86	0.51	0.08	< 0.01*
Litter Cover	0.69	-0.72	0.01	0.18
Temperature	0.19	-0.98	0.01	0.42
Distance	N/A	N/A	0.07	< 0.01*

Table 2. Vector cosines of environmental variables and vegetation characteristics with each nonmetric dimensional scaling (NMDS) axis and associated significance for arthropod order biomass collected by sweep net sampling in 2015 and 2016 at Packsaddle Wildlife Management Area, western Oklahoma, United States. * indicates significance at α=0.05. Axis values for transect distances indicate centroid locations for each factor level.

Biomass	**NMDS1**	**NMDS2**	**r^2**	**P value**
Maximum Vegetation Height	0.87	0.49	0.21	< 0.01*
Litter Depth	0.49	0.87	0.15	< 0.01*
Warm Season Grass Cover	0.56	0.83	0.08	< 0.01*
Forb Cover	0.99	-0.02	0.07	< 0.01*
Shrub Cover	0.41	0.91	0.02	0.18
Bare Ground	-0.65	-0.76	0.19	< 0.01*
Litter Cover	0.94	0.33	0.02	0.09
Temperature	-0.59	-0.80	0.02	0.12
Distance	N/A	N/A	0.29	< 0.01*

Table 3. Mean value ±1 SE for temperature and vegetation characteristics at -0.5 m, 1 m, 10 m, and 100 m from active oil and gas pads in 2015 and 2016 at Packsaddle Wildlife Management Area in western Oklahoma, United States.

Covariate	**Sampling Distance**			
	-0.5	**1**	**10**	**100**
Temperature (T_s - T_a °C)	16.41 ± 0.69	15.62 ± 0.83	13.34 ± 0.94	11.31 ± 0.81
Maximum Vegetation Height (mm)	118.82 ± 20.42	306.42 ± 24.30	594.74 ± 37.35	747.98 ± 30.46
Average Litter Depth (mm)	0.39 ± 0.13	1.84 ± 0.34	9.82 ± 1.50	11.72 ± 1.31
Warm Season Grass Cover (%)	7.18 ± 1.41	17.05 ± 2.54	22.73 ± 3.41	33.89 ± 3.79
Forb Cover (%)	3.10 ± 0.34	8.10 ± 1.56	16.81 ± 2.65	8.37 ± 1.44
Shrub Cover (%)	7.10 ± 2.61	3.55 ± 0.73	9.96 ± 2.28	25.48 ± 3.90
Bare Ground Cover (%)	78.71 ± 3.98	60.41 ± 4.35	34.85 ± 3.99	18.81 ± 2.67
Litter Cover (%)	3.31 ± 0.39	3.81 ± 0.47	7.96 ± 1.47	6.98 ± 0.99
Soil Moisture (% by weight)	4.28 ± 1.80	6.26 ± 1.18	4.84 ± 0.42	5.29 ± 0.57

Table S1. Species scores for orders in NMDS analysis of sweep net arthropod abundance at -0.5 m, 1 m, 10 m, and 100 m from active oil and gas pads in 2015 and 2016 at Packsaddle Wildlife Management Area in western Oklahoma, United States.

	NMDS1	**NMDS2**
Araneae	-0.50	0.25
Coleoptera	0.07	0.28
Diptera	0.37	-0.66
Hemiptera	-0.31	-0.10
Hymenoptera	0.28	-0.49
Lepidoptera	0.96	-0.22
Orthoptera	0.57	0.54
Total Abundance	-0.30	-0.01

Table S2. Species scores for orders in NMDS analysis of sweep net arthropod biomass at -0.5 m, 1 m, 10 m, and 100 m from active oil and gas pads in 2015 and 2016 at Packsaddle Wildlife Management Area in western Oklahoma, United States.

	NMDS1	**NMDS2**
Araneae	-0.82	0.41
Coleoptera	0.03	0.51
Diptera	-0.90	-0.03
Hemiptera	0.06	0.25
Hymenoptera	-0.94	0.52
Lepidoptera	0.51	0.38
Orthoptera	2.43	-0.29
Total Abundance	1.66	-0.10

VITA

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