



Original research article

Effects of conventional energy infrastructure on leporid and sciurid density in a sagebrush-steppe ecosystem

Lucretia E. Olson^{a,*}, John R. Squires^a, Mike Lockhart^b, Robert J. Oakleaf^c,
Chad Olson^d, Lauren E. Throop^e, Zachary P. Wallace^c

^a Rocky Mountain Research Station, United States Forest Service, 800 E Beckwith Ave, Missoula, MT 59801, USA

^b Wildlands Photography and Bio-Consulting, 1104 Renshaw Street, Laramie, WY 82072, USA

^c Wyoming Game and Fish Department, 260 Buena Vista, Lander, WY 82520, USA

^d HWA Wildlife Consulting, LLC, 2308 South 8th St, Laramie, WY 82070, USA

^e Wyoming Game and Fish Department, 1212 S. Adams St, Laramie, WY 82070, USA



ARTICLE INFO

Keywords:

Oil and gas development
Prey abundance
Ferruginous hawk
Small mammal
Jackrabbit
Ground squirrel
Wyoming

ABSTRACT

Shrub and grassland ecosystems are globally threatened, and loss and degradation continue to increase due to a variety of human activities. Many species of conservation concern live in these ecosystems and rely on small mammals, such as rabbits (Leporidae) and squirrels (Sciuridae) for prey. In North America, Wyoming contains one of the largest remaining intact shrubland ecosystems but is also the location of extensive energy development. We determined the response of small mammal abundance to surface disturbance (oil or gas well pads and roads), and to vegetation and topographic indices. We surveyed leporids and sciurids from 2012 to 2016 across Wyoming near occupied nests of ferruginous hawks (*Buteo regalis*), a raptor of conservation concern. We then predicted leporid and sciurid abundance given the existing amount of surface disturbance and for a scenario with no disturbance to quantify the landscape-scale effects of energy development on prey density. Finally, as a potential mechanism for differences in prey abundance, we tested for differences in remotely-sensed measures of vegetation between disturbed and undisturbed areas. We found that above-ground density of both leporids and sciurids was highest at undisturbed areas and lowest at roads. Current levels of disturbance result in 18% fewer leporids and 29% fewer sciurids compared to undisturbed sample sites. Vegetation cover differed significantly between undisturbed areas and disturbed, but not between disturbance types (roads and well pads). We conclude that energy infrastructure has an influence on fine-scale prey abundance but may be mediated by the maintenance of undisturbed areas in energy fields and revegetation of surface disturbance.

1. Introduction

Temperate grasslands and shrublands are the most globally at risk biome, with ~50% converted to human use and only 4.6% receiving habitat protection (Hoekstra et al., 2004). Grassland and sage-steppe ecosystems provide an important array of ecosystem services, including key habitat for a variety of game (Fedy et al., 2014; Stewart et al., 2010) and non-game species (Knick et al., 2003;

* Corresponding author.

E-mail address: lucretia.olson@usda.gov (L.E. Olson).

<https://doi.org/10.1016/j.gecco.2026.e04332>

Received 12 February 2026; Received in revised form 4 June 2026; Accepted 13 July 2026

Available online 17 July 2026

2351-9894/Published by Elsevier B.V. This is an open access article under the CC BY-NC-ND license (<http://creativecommons.org/licenses/by-nc-nd/4.0/>).

Wallace, Kennedy, Squires, Olson, et al., 2016), grazing for livestock (Davies et al., 2024), as well as water purification (Sirimarco et al., 2018) and carbon sequestration (Chang et al., 2021). In North America, an estimated 60–80% of mixed-grass prairie (Samson and Knopf, 1994) and 55% of sagebrush biome (Remington et al., 2021) have been converted or degraded. Of the remaining sagebrush dominated areas, approximately 70% in the United States are publicly owned (Knick et al., 2003). Thus, the preservation of these ecosystems in the US rests primarily on the actions of public land management agencies.

Native sagebrush-steppe and grassland ecosystems face many threats, with habitat loss and degradation caused by conversion to agriculture, invasive species, altered fire regimes, juniper encroachment, and human development (Davies et al., 2011). In North America, this ecotype has also been extensively developed for current and future energy production through fossil fuel extraction and solar and wind energy facilities (Copeland et al., 2009; Maguire et al., 2024; Trainor et al., 2016). The use of unconventional extraction methods (hydraulic fracturing and horizontal drilling) is also increasing (McMahon et al., 2024), which can result in larger well pads but potentially fewer wells (Grushecky et al., 2022). Energy production activities increase rangeland fragmentation (Finn and Knick, 2011), with cascading effects on the entire ecosystem, including changes in habitat permeability leading to loss of migration routes (Sandoval Lambert et al., 2022; Sawyer et al., 2009), increased human footprint leading to an increase in direct mortality through shooting, vehicle collision, and other anthropogenic factors (Katzner et al., 2020; Millsap et al., 2022; Northrup and Wittermyer, 2013; Thomason et al., 2023), and increased invasive vegetation leading to a loss of wildlife and livestock forage (DiTomaso, 2000).

Structure and composition of vegetation is a primary influence on animal community diversity and abundance (Thompson and Gese, 2013), and thus the changes resulting from energy related surface disturbance may have implications for species occurrence or abundance. Energy development frequently clears native vegetation (Allred et al., 2015) and inadvertently introduces non-native invasive vegetation, often resulting in altered plant communities or open, unvegetated areas (Nasen et al., 2011; Walsh and Rose, 2022). This can alter rangeland communities, as some rangeland species are known to prefer greater vegetation cover (Jimenez et al., 2020) or exhibit declines when native vegetation is replaced with non-native species (Ostoja and Schupp, 2009), while others are associated with open areas (Baker et al., 2013) or are positively influenced by non-native species (Litt and Pearson, 2013). Generally, native wildlife have been shown to respond negatively to the presence (Chalfoun, 2021), density (Green et al., 2017), or noise (Kleist et al., 2021) of oil and gas development, although some generalist or non-native species may benefit (Fiebler et al., 2017).

In the US, the state of Wyoming contains some of the largest remaining intact sagebrush steppe ecosystem (Scholtz and Twidwell, 2022) as well as some of the highest density and fastest growing energy development in the country (Copeland et al., 2009; Trainor et al., 2016). This ecosystem is also home to a variety of species of conservation concern, including ferruginous hawks (*Buteo regalis*), an at-risk raptor with potentially declining abundance (Hoffman and Smith, 2003; Houston and Bechard, 1984). Ferruginous hawks are dependent on small mammals as prey, with approximately 66% of their diet biomass composed of leporids (animals from the family Leporidae, including rabbits and hares) and 25% composed of sciurids (animals from the family Sciuridae, including prairie dogs, ground squirrels, and chipmunks) (Collins and Reynolds, 2005). Wyoming supports a large nesting population of ferruginous hawks (Olson et al., 2015) and their primary prey: white- and black-tailed prairie dogs (*Cynomys leucurus* and *C. ludovicianus*, respectively), cottontail rabbits (*Sylvilagus* spp.), white-tailed jackrabbits (*Lepus townsendii*), Wyoming ground squirrels (*Urocitellus elegans*), and least chipmunks (*Neotamias minimus*). The habitat needs of these prey species vary but all may be affected by habitat change resulting from oil and gas disturbance. Leporid species in Wyoming often prefer dense cover from shrubs or tall grass (Jimenez et al., 2020; Simes et al., 2015). Sciurid species in this ecosystem are primarily burrowing mammals that require vegetation for food and cover. Prairie dogs and ground squirrels, however, often prefer reduced cover near their burrows (Baker et al., 2013), to enable earlier detection of predators. The response of these prey species to the altered vegetation conditions resulting from energy development is not well studied.

We studied the influence of oil and gas development on leporid and sciurid abundance in a sagebrush steppe ecosystem in Wyoming from 2012 to 2016. We focused on these species due to their importance to ferruginous hawks, a species of conservation concern. Our research questions for this study were: 1) Does prey density differ at a fine scale around conventional energy disturbance (energy wells and roads) and in areas that are relatively undisturbed by development? 2) Does this difference impact prey abundance at a landscape scale? 3) Are there differences in vegetation across disturbance types (wells, roads, and undisturbed areas) that may drive differences in prey abundance? To answer these questions, we surveyed leporids and sciurids at 24 sample sites across Wyoming and quantified abundance around energy wells, roads, and in undisturbed areas using line and point transect distance sampling. We used the best performing distance models to predict prey abundance given current energy disturbance and an alternate scenario of no disturbance to quantify the landscape-scale effects of energy development infrastructure on prey density. We examined differences in remotely sensed measures of vegetation cover around roads, well pads, and undisturbed transects to understand the mechanism behind differences detected in prey abundance. We predict that prey density will be greater in undisturbed areas, that this will contribute to an overall decrease in prey abundance at a landscape scale, and that well pads and roads will have decreased vegetation cover compared to undisturbed areas.

2. Methods

2.1. Study area

We sampled leporids and sciurids as part of a larger study on ferruginous hawks in Wyoming (Johnson et al., 2019; Olson et al., 2015, 2017; Squires et al., 2020; Wallace, Kennedy, Squires, Oakleaf, et al., 2016; Wallace, Kennedy, Squires, Olson, et al., 2016), and thus our surveys were focused on circular areas with a 1.5-km radius centered on an occupied ferruginous hawk nest (Wallace, Kennedy, Squires, Oakleaf, et al., 2016), hereafter referred to as ‘sample sites’. We focused on two main geographic areas, in

south-central and north-eastern Wyoming, USA (Fig. 1), in the Wyoming Basin and Northwestern Great Plains ecoregions, respectively. Minimum convex polygons (MCP) drawn around all sample sites in each region span 1196 km² in the Wyoming Basin and 420 km² in the Northwestern Great Plains.

The Wyoming Basin study area is characterized by sagebrush-steppe vegetation, with mean winter temperature averaging -5°C and mean summer temperature $\sim 18^{\circ}\text{C}$, mean annual precipitation 24 cm, and elevation between 1950 – 2262 m asl (AdaptWest, Project. 2015; Wang et al., 2016). The Northwestern Great Plains are dominated by grasslands with some sagebrush-steppe vegetation, mean winter temperature -4°C , mean summer temperature $\sim 20^{\circ}\text{C}$, slightly more precipitation than the Wyoming Basin (39 cm), and lower elevation between 1330 – 1590 m asl (AdaptWest, Project. 2015; Wang et al., 2016). Both areas have significant oil and gas extraction activities, with the Wyoming Basin MCP containing approximately 1810 producing gas wells and 7 producing oil wells at the time of data collection, and the Northwestern Great Plains MCP containing 151 producing gas wells and 72 producing oil wells (WOGCC, 2015). The Northwestern Great Plains also contains extensive coal mining operations on the east side of the study area. The regions also differ in the mammalian prey communities present. The Wyoming Basin study area is within the distributions of Wyoming ground squirrels, white-tailed prairie dogs, and pygmy rabbits (*Brachylagus idahoensis*), while the Northwestern Great Plains is within the distributions of black-tailed prairie dogs, Eastern cottontails (*Sylvilagus floridanus*), and black-tailed jackrabbits (*Lepus californicus*). Both contain white-tailed jackrabbits, desert cottontails (*Sylvilagus audubonii*), thirteen lined ground squirrels (*Ictidomys tridecemlineatus*), and least chipmunks (WYNDD, 2024).

2.2. Field methods

We surveyed for prey species from 2012 to 2016. Surveys typically began in July and continued through the first week of August prior to estivation of Wyoming ground squirrels (Zegers, 1984), and was carried out between the hours of 06:00 and 11:00 AM, when prey were expected to be most active due to cooler temperatures. We delimited energy infrastructure for sampling using the temporally closest NAIP (National Agriculture Imagery Program) imagery, visually assessing sample sites for presence of surface disturbance. We focused on two types of common anthropogenic disturbance from energy development: energy well pads and energy roads, as well as a randomly placed transect in undisturbed areas with little visual sign of anthropogenic disturbance. We sampled leporids on line-transects, since these species are difficult to detect unless flushed while walking. Technicians surveyed 3–4 km of roads and 3–4 km of linear transects in undisturbed areas per sample site, spatially distributed evenly over the entire area in sub-transects of 1 km each (Fig. 2). Along each linear transect, observers moved 10 m from the road edge or followed a straight-line path in undisturbed areas. We sampled all well pads in the sample site with two circular transects each, one 20 m from the edge of the visually disturbed well pad area, and a second 100 m from this edge; the length of well pad transects varied depending on the size of the disturbed area. We sampled sciurids on points, since these species were more likely to emerge from hiding when the observer stood still. Technicians

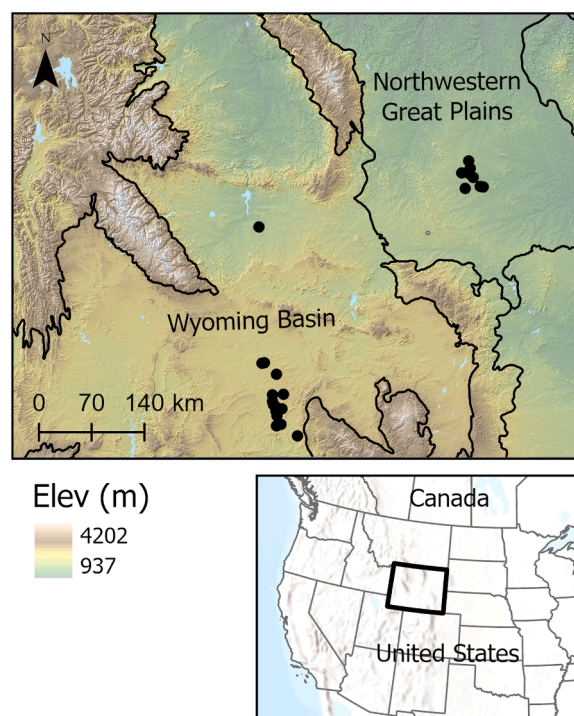


Fig. 1. The location of sample sites (black dots) for leporid and sciurid prey abundance within Wyoming, USA, 2012–2016. Inset shows the location of the study region within North America.

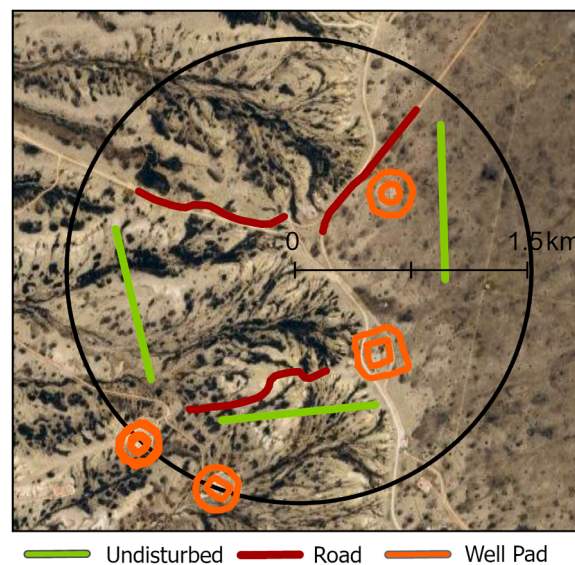


Fig. 2. An example of a prey sample site in Wyoming, USA, 2015, with the layout of the three types of survey transects shown: undisturbed, roads, and well pads. The black line shows the outline of a 1.5 km radius circle around an occupied ferruginous hawk nest.

sampled 4, 5-minute point surveys per transect, spacing them every 333 m along linear transects or at each of the cardinal directions along circular transects. Technicians avoided double-counting potential prey species to the extent possible, particularly on well pad point samples, which could be close together. Technicians recorded the distance and azimuth to each detection, as well as species and group size, if more than one individual was in the same location, and the ambient temperature ($^{\circ}\text{C}$) at the start of each sampling period.

3. Analysis

3.1. Leporid and sciurid density across disturbance types

We used distance sampling methods to determine the relative density of prey species since these models allow estimation of abundance response to environmental covariates while accounting for imperfect detection (Buckland, 1985; Buckland et al., 2001). We modeled prey at each surface disturbance type using the program ‘unmarked’ v.1.3.2 (Fiske and Chandler, 2011) in program R v.4.3.2 (R Core Team, 2023). The ‘unmarked’ package estimates the detection function and the abundance response simultaneously, allowing the estimation of covariate effects on both detection and abundance. We performed separate analyses for the leporid family (line transects) and sciurid family (point transects). We truncated detections at 100 m for leporids and 350 m for sciurids based on visual examination of detection distance histograms to avoid including low-probability detections far from the observer (Buckland et al., 2001). We did not attempt to model detection for individual leporid or sciurid species due to sample size limitations (Table 1). We acknowledge that the sciurids in our sample are semi-fossorial, so that our density estimate is relative and includes only those

Table 1

Raw counts of leporids detected on transect surveys and sciurids detected on point surveys in eastern (NW Great Plains) and western (Wyoming Basin) Wyoming, USA, from 2012 to 2016. Values of ‘NA’ indicate no sampling took place for those species in a given year.

Ecoregion	Species	2012	2013	2014	2015	2016	Total
<i>Leporids</i>							
NW Great Plains	Desert Cottontail	NA	NA	22	80	23	125
NW Great Plains	White-Tailed Jackrabbit	NA	NA	10	41	4	55
Wyoming Basin	Desert Cottontail	15	16	33	66	NA	130
Wyoming Basin	White-Tailed Jackrabbit	2	10	3	8	NA	23
Wyoming Basin	Pygmy Rabbit	0	3	0	3	NA	6
	TOTAL	17	29	68	198	27	339
<i>Sciurids</i>							
NW Great Plains	Least Chipmunk	NA	NA	0	5	0	5
NW Great Plains	Thirteen-Lined Ground Squirrel	NA	NA	1	7	1	9
Wyoming Basin	Least Chipmunk	8	16	13	14	NA	51
Wyoming Basin	Thirteen-Lined Ground Squirrel	0	0	0	1	NA	1
Wyoming Basin	White-Tailed Prairie Dog	57	548	262	206	NA	1073
Wyoming Basin	Wyoming Ground Squirrel	2	150	229	13	NA	394
	TOTAL	67	714	505	246	1	1533

individuals that are above ground and available for detection at any given time.

For each prey group (leporid or sciurid) we used a multi-stage candidate approach to determine the best-performing overall model (Morin et al., 2020). Each model contained three components to be selected: the shape of the detection function, the detection model (p), and the abundance model (λ). First, we considered the shape of the detection function and fit 4 null ($p(\cdot)\lambda(\cdot)$) models, varying only in the detection distribution ('half-normal', 'exponential', 'hazard rate', and 'uniform'). We used AICc (Akaike Information Criterion for small sample sizes; Akaike, 1974; Burnham and Anderson, 2002) to select the best-performing model. Second, we carried this best-performing detection function forward and fit a set of candidate models with varying detection covariates and null abundance ($p(\text{cov})\lambda(\cdot)$) parameters. To avoid model-overfitting, our detection (p) candidate models consisted of univariate models of each covariate (see next paragraph for covariate details). We used AICc to select the top-performing detection models, using a liberal cutoff by which either all models $\leq 5 \Delta\text{AIC}$ or the top three models (if ΔAIC was > 5) advanced to the next stage. Third, we fit a set of candidate models which varied in their abundance covariates while holding the detection covariates at their null ($p(\cdot)\lambda(\text{cov})$). For abundance (λ) models, we first ran univariate models with all possible covariates, then ranked these by AICc and selected all uncorrelated ($|r| > 0.6$) covariates that performed better than a null model. We then used these uncorrelated covariates to construct multivariate abundance candidate models for each species group and again used AICc to select the top-performing three models or $\leq 5 \Delta\text{AICc}$ models. Finally, we ran all combinations of the chosen detection models with the chosen abundance models ($p(\text{cov})\lambda(\text{cov})$) and used AICc to determine the single best-performing model for each species group. For final models, we considered quadratic forms of all continuous covariates and retained these terms if supported by likelihood ratio tests comparing a model with the linear form to one with the linear and quadratic form of the covariate.

We considered 10 covariates for detection and abundance candidate models, from field-collected and remotely-sensed data sources. Vegetation covariates included remotely-sensed percent cover (0–100%) of herbaceous vegetation, shrubs, sagebrush, bare ground, and plant litter as well as mean shrub height (cm) at a 30 m resolution (Homer et al., 2012). Topographic covariates included site exposure, integrated moisture, and the central longitude or latitude of points or line transects. Site exposure is an index of localized heat exposure to an area, and is calculated as a combination of aspect and slope (Balice et al., 2000; Evans et al., 2014), with higher values indicating warmer conditions. Similarly, integrated moisture is an index of localized soil moisture estimated using the curvature and topography of the landscape (Evans et al., 2014; Iverson et al., 1997), with higher values indicating greater soil moisture. Both rely on a digital elevation model, which we accessed at 30-m resolution from the publicly available USGS website (U.S. Geological Survey, 2019). We included the average longitude and latitude of transects as covariates since prey communities and environmental conditions are likely to vary within and between our two study regions. Finally, we considered a categorical covariate of disturbance type ('road', 'well pad', or 'undisturbed'). Additionally, for detection models we considered covariates that we hypothesized would be likely to affect prey detectability, including the temperature recorded by technicians at the start of surveys and the day of the year. We standardized covariates for ease of model fitting and interpretation.

We surveyed a total of 24 unique sample sites, with some sites surveyed in successive years depending on whether the ferruginous hawk nest was currently occupied (see Wallace et al. (2016a) for details on determining nest occupancy). Out of 24 sample sites, 14 were surveyed once, 5 twice, and 5 three times. Since the majority of sample sites were surveyed only once, we opted to consider each site-year combination as a unique sample unit, resulting in a total of 39 site-years (Burnett and Roberts, 2015; Linden et al., 2017). To control for non-independence among sample sites and years, we considered year as either a random or fixed effect in candidate detection models (Supplementary Tables 2 and 6) and year as a fixed effect and sample site as a random effect in abundance models (Supplementary Tables 4 and 8).

For sciurid species, raw prey counts indicated very few individuals (9 thirteen-lined ground squirrels, 5 least chipmunks) detected in the Northwestern Great Plains study area (Table 1). While black-tailed prairie dogs are found in this area, we encountered them in only two sample sites, both times in groups of over 250. Since most sciurids were infrequently encountered and black-tailed prairie dogs were patchily distributed and occurred as numeric outliers, we opted to limit our sciurid models to only sites in the Wyoming Basin.

We validated model fit using a bootstrap procedure which randomly resampled the data 100 times, refit the model, and calculated a fit statistic by subtracting the modeled abundance from the observed abundance at each sample site. We compared the fit statistic from the original model to the bootstrapped distribution and interpreted values within the 95% range to indicate good model fit (Fiske and Chandler, 2011). Additionally, we used our top-performing models to calculate site-specific leporid and sciurid density (individuals/ha) at sampled locations and compared these values to unmodeled raw prey counts to verify general model accuracy.

3.2. Leporid and sciurid abundance at a landscape scale

To quantify the landscape-scale effects of energy development infrastructure on prey abundance, we assessed the proportional loss of prey abundance due to current levels of development by comparing the observed abundance at our sample sites to a counterfactual scenario (in the sense of Katzner et al., 2022) in which all roads and well pads were replaced by undisturbed area. Counterfactual analyses provide a method for assessing the biological impact of an action or existing condition compared with a theoretical alternative scenario (Baylis et al., 2016; O'Garra et al., 2025). We predicted prey abundance for each sample site (1.5 km radius) using the modeled density values from the top-performing models selected above. We held all other covariates at their means and quantified the area within each sample site that was impacted by wells and roads by buffering each point (well) or linear feature (road) by its distance-model truncation distance. We used the Wyoming Oil and Gas Conservation Commission data (WOGCC, 2015) to delineate all active oil or gas wells within each sample site and a publicly available road layer from the Bureau of Land Management (BLM) to delineate all road features, including dirt roads, gravel roads, or paved roads (O'donnell et al., 2014). We used a parametric bootstrap

procedure with $n = 1000$ repetitions to generate confidence intervals for each abundance estimate. For each sample site, we calculated prey abundance at current disturbance levels and with no disturbance, to simulate prey abundance if the landscape were completely undisturbed. We calculated the difference in prey abundance between these two scenarios at every bootstrap iteration, calculated the 95% CI, and considered the difference in prey abundance between scenarios significant if the confidence interval did not overlap 0 (Nakagawa and Cuthill, 2007).

3.3. Vegetation across disturbance types

To understand a potential mechanism behind prey differences or similarities across disturbance types, we assessed differences in vegetation between roads, well pads, or undisturbed areas. We used one-way ANOVA to test for differences among means of each remotely-sensed vegetation metric (bare ground, shrubs, shrub height, herbaceous cover, plant litter, and sagebrush) measured at points where prey was sampled. We compared means across disturbance types ('road', 'well pad', and 'undisturbed') and used post-hoc Tukey's HSD tests to further determine which pairs differed when ANOVA results were significant.

4. Results

4.1. Leporid and sciurid density across disturbance types

For leporid species, we surveyed 24 sample sites and 39 site-years, for a total of 423 km of transects over all 5 study years: 220 km around well pads, 91 km along roads, and 113 km in undisturbed locations. Per sample site, we surveyed a mean of 11 km (range: 6–19 km) of transects. For sciurid species, 13 of these sample sites were in the Wyoming Basin (24 site-years), and a mean of 63 points (range: 32–96 points) were surveyed per sample site. Line-transects detected a total of 339 leporids and point-count surveys detected a total of 1533 sciurids (Table 1). Sample sites contained a mean of 8 active oil or gas wells (range: 1–21 oil or gas wells, WOGCC) at the start of the study and 11.7 km of roads (range: 2.1–28.0 km).

Model selection results for the leporid preliminary candidate models on distribution shape, detection, and abundance are given in Supplementary Tables 1–4. The hazard function was best fitting for the shape of the leporid detection function (Supplementary Table 1). There was some model uncertainty in the final leporid candidate models (Table 2); the top two models were similar in AICc and identical in abundance covariates but differed in detection covariates. We selected the model with the fewest parameters as the most parsimonious (Burnham and Anderson, 2002). Thus, leporid detection probability included a random effect of year and was greater at cooler start temperatures (range of start temperatures: 7–30°C); average detection probability across all sample sites was 0.17 (95% CI: 0.16–0.18). Leporid abundance was best modeled with topographic covariates; no univariate vegetation covariates performed better than the null. The top performing abundance model for leporids included disturbance type, longitude, day of year, survey year, and an interaction between site exposure and disturbance type (Table 2, Fig. 3). Based on this model, leporids were similarly abundant at well pads and undisturbed transects, and less abundant near roads (Fig. 4). When site exposure was low (resulting in cooler conditions), abundance was higher at well pads and lower at undisturbed transects, while in warmer conditions, abundance was higher at undisturbed transects and lower at well pads (Fig. 4). Additionally, leporids were more abundant towards the western portion of the study area and in years 2015 and 2016 (Fig. 4). Between-site variation in abundance was high, based on modeled random intercepts, while between-year variation in detection was slightly lower (Fig. 3). Modeled site-level density was 0.36 (95% CI: 0.18–0.75) leporids per ha at random transects, 0.31 (95% CI: 0.18–0.57) leporids per ha at well pads, and 0.16 (95% CI: 0.07–0.37) leporids per ha at roads, compared to raw (unmodeled) estimates ranging from 0.01 to 0.11 leporids/ha (Table 3). Bootstrap validation indicated good model fit, with the model-estimated fit statistic within the 95th percentile of bootstrap estimates ($p = 0.17$).

Model selection results for the sciurid preliminary candidate models on distribution shape, detection, and abundance are given in Supplementary Tables 5–8. The half-normal function was best fitting for the shape of the sciurid detection function (Supplementary

Table 2

Final abundance and detection distance sampling models for leporid detections on line transects in Wyoming, USA, 2012–2016. Table shows the number of model parameters (K), AICc (Akaike Information Criteria for small sample size), delta AICc, AICc weight, and the model log likelihood (LL). All abundance models included a random intercept of individual sample site, not shown in table. Covariate abbreviations include: temperature (Temp), disturbance type (Type), longitude (Lon), day of year (DOY), site exposure (SiteExp), integrated moisture index (IntMoist). Most parsimonious top performing model is shown in bold.

Detection Model (p)	Abundance Model (λ)	K	AICc	Δ AICc	AICcWt	LL
Year	Type + Lon + DOY + Type:SiteExp + IntMoist + Year	20	926.15	0	0.5	–438.36
Temp + (1 Year)	Type + Lon + DOY + Type:SiteExp + IntMoist + Year	18	926.31	0.15	0.47	–441.39
Temp + (1 Year)	Type + Lon + DOY + SiteExp + IntMoist + Year	16	933.78	7.63	0.01	–447.97
Year	Type + Lon + DOY + SiteExp + IntMoist + Year	18	933.81	7.66	0.01	–445.15
Year	Type + Type:Lon + DOY + SiteExp + IntMoist + Year	20	935.1	8.95	0.01	–442.83
Temp + (1 Year)	Type + Type:Lon + DOY + SiteExp + IntMoist + Year	18	937.19	11.04	0	–446.84
Type + (1 Year)	Type + Lon + DOY + Type:SiteExp + IntMoist + Year	19	937.98	11.83	0	–445.77
Type + (1 Year)	Type + Lon + DOY + SiteExp + IntMoist + Year	17	945.72	19.57	0	–452.53
Type + (1 Year)	Type + Type:Lon + DOY + SiteExp + IntMoist + Year	19	946.88	20.72	0	–450.22

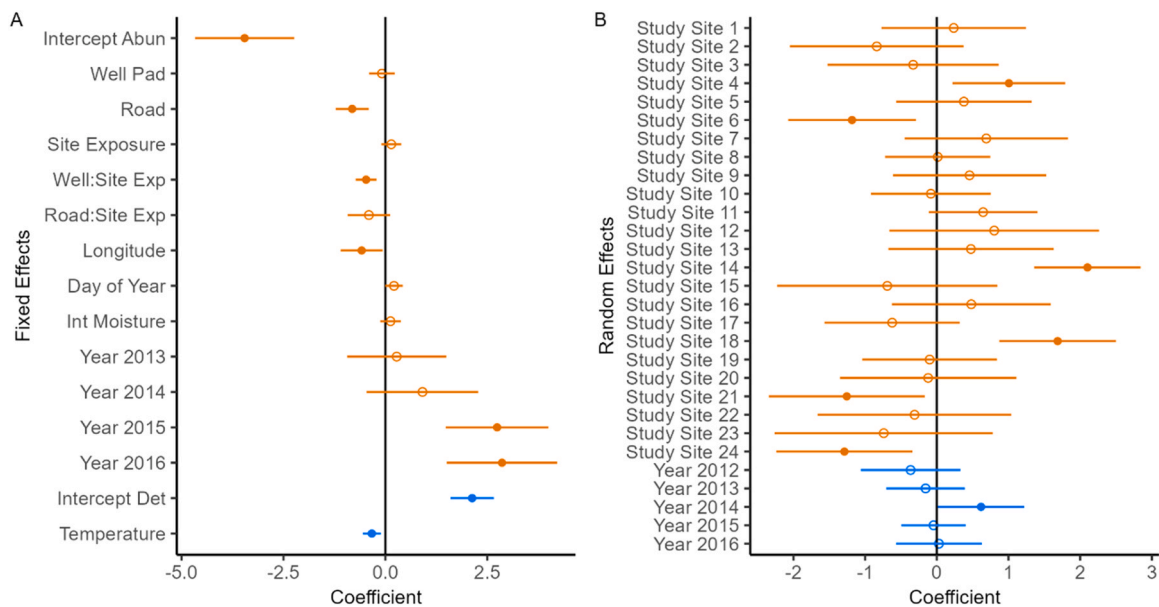


Fig. 3. Model coefficients for A) fixed and B) random effects in the top-performing leporid abundance (orange dots) and detection (blue dots) distance sampling model. Whiskers show 95% confidence intervals; filled circles indicate significant covariates that do not cross the vertical line at 0 and open circles indicate non-significant covariates. Abbreviations for fixed effects include abundance model intercept (Intercept Abun), site exposure (Site Exp), integrated moisture (Int Moisture), and detection model intercept (Intercept Det).

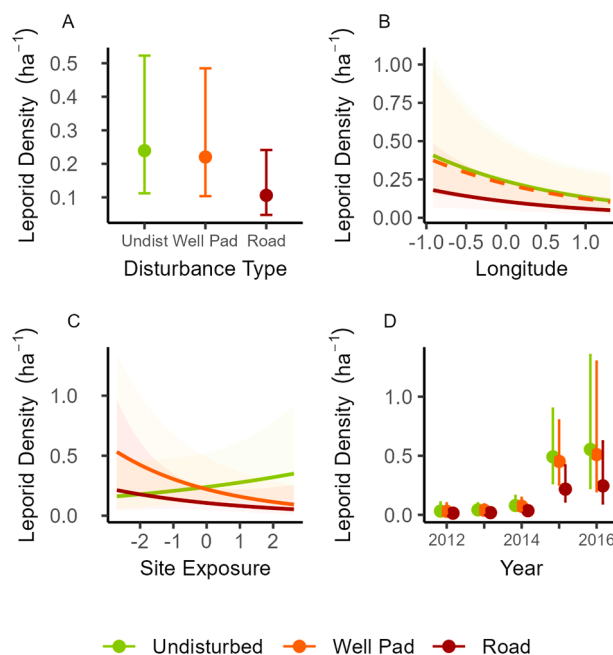


Fig. 4. Significant covariates from the top-performing distance sampling model (Table 2) for leporid density in Wyoming, USA, 2012–2016. Plots show marginal response (and 95% CI) of leporid density to A) disturbance type, B) longitude, C) site exposure (index of temperature), and D) year, for each disturbance type (undisturbed areas (Undist), well pads, and roads) while holding all other covariates at mean values. Panel B shows well pads as a dashed line to increase visibility of overlapping lines.

Table 5). Sciurid detection on point transects included a random effect of year and was higher with lower shrub cover; average detection probability across all sample sites was 0.20 (95% CI: 0.18–0.22). Sciurid above-ground abundance was best predicted by disturbance type, day of year, percent cover of shrubs and herbaceous plants, integrated moisture, and survey year, with a quadratic effect of day of year and herbaceous cover (Table 4, Fig. 5). Sciurid abundance was higher at undisturbed transects and lower at well

Table 3

Raw and modeled density of leporids on line transects sampled at energy well pads, roads, or undisturbed areas in the Northwestern Great Plains and the Wyoming Basin, Wyoming, USA, 2012–2016. Raw and modeled densities are in count per ha, and the 95% confidence interval is provided for modeled density.

Ecoregion	Disturbance type	Total length (km)	Area (ha)	Raw Count	Raw Density	Modeled Density
NW Great Plains	Undisturbed	56.82	1136.36	60	0.05	0.46 (0.23–0.99)
NW Great Plains	Well Pad	44.89	897.82	96	0.11	0.57 (0.36–0.92)
NW Great Plains	Road	43.44	868.74	24	0.03	0.21 (0.10–0.50)
Wyoming Basin	Undisturbed	56.06	1121.14	38	0.03	0.27 (0.14–0.57)
Wyoming Basin	Well Pad	174.81	3496.16	108	0.03	0.19 (0.11–0.40)
Wyoming Basin	Road	47.14	942.78	13	0.01	0.11 (0.05–0.27)

pads and roads (Fig. 6). Sciurid abundance also increased with lower shrub cover, moderate herbaceous cover, and in the middle of the survey season (Fig. 6). Between-site variation in abundance was high, based on modeled random intercepts, while between-year variation in detection was lower (Fig. 5). Modeled above-ground site-level density of sciurids averaged 0.16 (95% CI: 0.12–0.20) sciurids per ha across all study sites in the Wyoming Basin, with 0.17 (95% CI: 0.13–0.23) sciurids per ha at undisturbed transects, 0.16 (95% CI: 0.13–0.20) sciurids per ha at well pads, and 0.15 (95% CI: 0.11–0.21) sciurids per ha around roads, compared to raw (unmodeled) estimates of 0.03 sciurids/ha for all surface disturbance types (Table 5). Bootstrap validation indicated good model fit, with the model-estimated fit statistic within the 95th percentile of bootstrap estimates ($p = 0.27$).

4.2. Leporid and sciurid abundance at a landscape scale

Sample sites were made up of an average of 53.1% undisturbed area (range: 4.1–94.3%), with 28.8% impacted by roads (range: 5.7–61.4%) and 21.7% impacted by well pads (range: 1.8–49.1%), based on the area of each disturbance type calculated using buffers around features (100 m around roads, 350 m around wells). Given existing conditions, there were an average of 451 (95% CI: 194–770) leporids and 346 above-ground sciurids (95% CI: 191–560) per sample site. At a theoretically undisturbed sample site, the top-performing models predicted an average of 549 (95% CI: 228–949) leporids and 486 sciurids (95% CI: 260–785). Thus, oil and gas surface disturbance at the levels found in our study areas led to a probable average loss of 98 (95% CI: 18–203) leporids and 140 (95% CI: 60–238) sciurids per sample site, a 17.8% and 28.8% decrease, respectively. Based on confidence intervals, differences are statistically significant at $p < 0.05$.

4.3. Vegetation across disturbance types

Based on remotely-sensed vegetation data, vegetation was significantly different ($p < 0.05$) for at least one pairwise comparison between roads, well pads, and undisturbed transects for all remotely sensed vegetation covariates except shrub cover (Fig. 7). All significant differences were between undisturbed and disturbed transects and there were no significant vegetation differences between well pads and roads. Specifically, undisturbed transects had less bare ground and greater shrub height and percent litter compared to roads and well pads, and greater percent herbaceous cover and sage cover than well pads, while there was no difference between roads or well pads for any vegetation type (Figure 7, Supplementary Table 9).

5. Discussion

Our results indicate that the abundance of leporids and sciurids is influenced by conventional oil and gas infrastructure, but this response may be mitigated by site-level characteristics. Both leporids and sciurids were more abundant at undisturbed transects and

Table 4

Final abundance and detection distance sampling models for sciurid detections on point transects in Wyoming, USA, 2012–2016. Table shows the number of model parameters (K), AICc, delta AICc, and AICc weight, and the model log likelihood (LL). All abundance models included a random intercept of individual sample site, not shown in table. Covariate abbreviations include: disturbance type (Type), day of year (DOY), integrated moisture index (IntMoist), % shrub cover (Shrub), % herbaceous cover (Herb). Top performing model is shown in bold.

Detection Model (p)	Abundance Model (λ)	K	AICc	Δ AICc	AICcWt	LL
Shrub+(1 Year)	DOY² +Type+Shrub+Herb+IntMoist+Herb² +Year	16	7874.69	0	1	–3921.15
Shrub+(1 Year)	DOY+Type+Shrub+Herb+IntMoist+IntMoist ² +Year	15	7920.5	45.81	0	–3945.08
Shrub+(1 Year)	DOY+Type+Shrub+Herb+IntMoist+Year	14	7923.39	48.7	0	–3947.54
Shrub+(1 Year)	DOY+Shrub+Type*Herb+IntMoist+Year	16	7924.39	49.70	0	–3946.00
Litter+(1 Year)	DOY+Type+Shrub+Herb+IntMoist+Year	14	7926.05	51.36	0	–3948.87
Shrub+(1 Year)	DOY+Type*Shrub+Herb+IntMoist+Year	16	7926.07	51.38	0	–3946.84
Litter+(1 Year)	DOY+Type+Shrub+Type*Herb+IntMoist+Year	16	7926.66	51.97	0	–3947.13
Litter+(1 Year)	DOY+Type*Shrub+Herb+IntMoist+Year	16	7929.22	54.53	0	–3948.41
Type+(1 Year)	DOY+Type+Shrub+Herb+IntMoist+Year	15	7945.02	70.33	0	–3957.34
Type+(1 Year)	DOY+Shrub+Type*Herb+IntMoist	17	7945.79	71.10	0	–3955.67
Type+(1 Year)	DOY+Type*Shrub+Herb+IntMoist+Year	17	7947.38	72.69	0	–3956.47

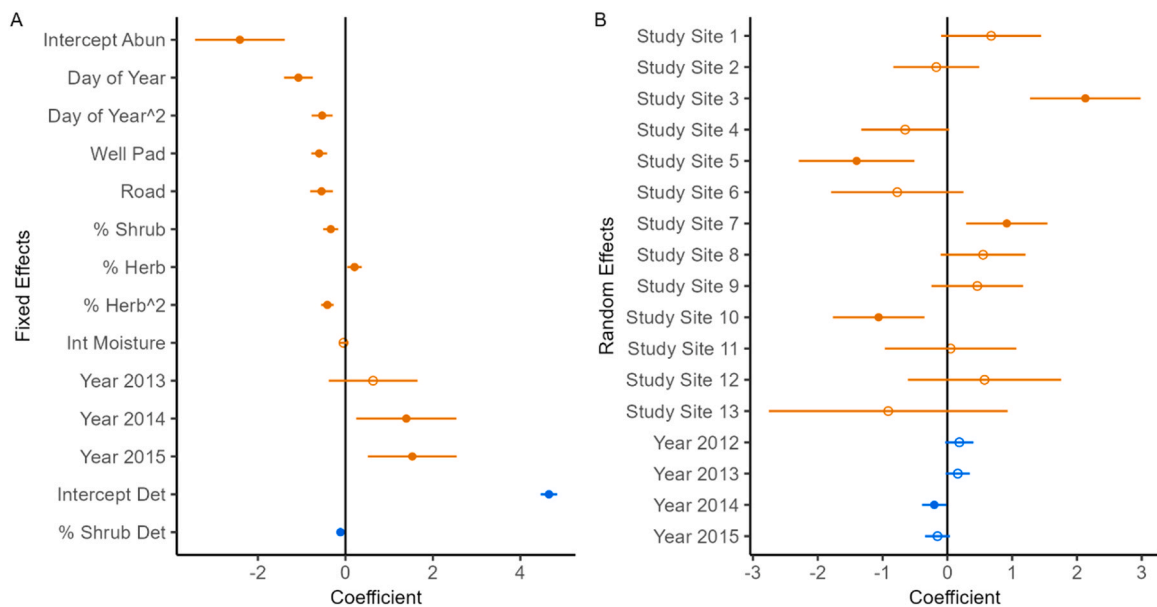


Fig. 5. Model coefficients for A) fixed and B) random effects of the top-performing sciurid abundance (orange dots) and detection (blue dots) distance sampling model. Whiskers show 95% confidence intervals; filled circles indicate significant covariates that do not cross the vertical line at 0 and open circles indicate non-significant covariates. Abbreviations for fixed effects include abundance model intercept (Intercept Abun), integrated moisture (Int Moisture), detection model intercept (Intercept Det), and detection model percent shrub (% Shrub Det).

less abundant at roads, although this relationship was mediated with changes in site conditions including heat exposure and vegetation cover. At a landscape scale, existing amounts of surface disturbance from energy well pads and roads are associated with a predicted 18% decrease in leporids and a 29% decrease in sciurids, compared to theoretically undisturbed areas. We found differences in vegetation around energy infrastructure disturbance compared to undisturbed areas, which may drive abundance differences for sciurids but potentially not leporids. These results highlight the importance of considering the local vegetation conditions as well as the type of disturbance if managing for increased prey abundance.

Oil and gas surface disturbance was consistently among the top univariate predictors for prey detection and abundance, and undisturbed transects had higher abundance than disturbed transects, consistent with existing literature, which tends to find a negative response to oil and gas disturbance in many species (Chalfoun, 2021). We found low prey abundance along road verges for both prey groups. The roads in our study sites were primarily energy-haul roads or local roads, both of which are generally unpaved, relatively narrow, and with comparatively low traffic volume. Our low prey densities at roads may be due to open conditions and revegetation characteristics, which generally include mainly herbaceous annuals or perennials (Hufford et al., 2023), resulting in a lack of hiding cover which may deter small mammals. In Wyoming, a study on vegetation around roads found that shrub height was greater in undisturbed areas, while grasses were taller around roads (Hufford et al., 2023). Alternatively, the unpredictable nature of the timing and frequency of passing vehicles in an otherwise undisturbed landscape may make it more difficult for animals to habituate to disturbance (Miller et al., 2001; Tablado and Jenni, 2017).

We found support for the prediction that vegetation differed between undisturbed and disturbed transects. Undisturbed areas in general had greater vegetation cover than disturbed areas, with less bare ground and more herbaceous cover, litter, and sagebrush, as well as taller shrubs. Only percent shrub cover didn't differ between disturbance types. A reduction in vegetation cover at roads (Hufford et al., 2023) and at oil and gas infrastructure (Jones et al., 2014; Nasen et al., 2011) compared to undisturbed sample sites has been found in other studies in arid or grassland ecosystems, both within a similar fine-scale footprint of disturbance (~0–25 m). There were no detected differences in vegetation between disturbance types, indicating that these disparate types of construction resulted in similar fine-scale habitat disturbance. Our sampling around roads and well pads was spaced such that the transects were primarily in revegetated area surrounding surface disturbance (10 m from road edges and 20–100 m from the unvegetated footprint of a well pad); thus it may be that the vegetation similarities between disturbance types are the result of removal or disturbance of native vegetation and replacement with similar reclamation seed mixes. Sanders and Chalfoun (2018) found that reclaimed areas around natural gas wells contained more bare ground, grass, and shrub cover and less sagebrush, and Hufford et al. (2023) found that reclaimed areas near roads contained more grass species and less sagebrush. Modeled sciurid abundance reflects vegetation differences, with greater abundance at undisturbed areas, similar abundance at roads and wells, and positive effects of percent herbaceous and shrub cover, suggesting that vegetation changes may be the driver behind these species' response to oil and gas, similar to other species in grassland (Ludlow et al., 2015) or shrub ecosystems (Fiebler et al., 2017). The main sciurid species in our surveys were white-tailed prairie dogs and Wyoming ground squirrels, both of which primarily forage on forbs and grasses (Clark et al., 1971; Zegers, 1984), and may perch in shrubs to aid in predator detection, as observed in other ground squirrel species (Hannon et al., 2006; van der Marel et al., 2021).

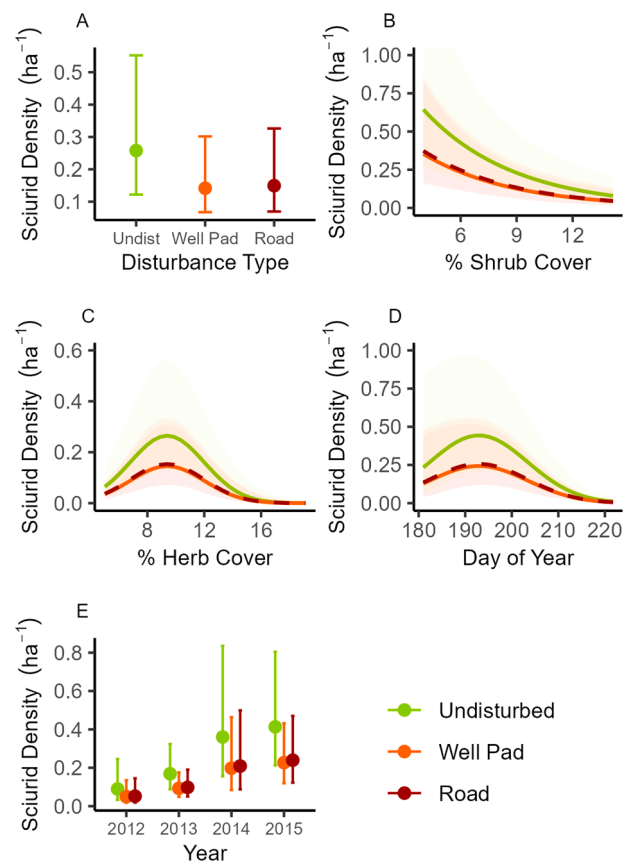


Fig. 6. Significant covariates from the top-performing distance sampling model (Table 4) for sciurid density in Wyoming, USA, 2012–2016. Plots show marginal response (and 95% CI) of sciurid density to A) disturbance type, B) percent shrub cover, C) percent herbaceous cover, D) day of the year, and E) year for each disturbance type (undisturbed areas (Undist), well pads, and roads) while holding all other covariates at mean values. The line for roads is shown dashed in panels B, C, and D to increase visibility of overlapping lines.

Table 5

Raw and modeled density of sciurids on point transects sampled at energy well pads, roads, or undisturbed areas in the Wyoming Basin, Wyoming, USA, 2012–2016. Raw and modeled densities are in count per ha, and the 95% confidence interval is provided for modeled density.

Ecoregion	Disturbance type	N Points	Area (ha)	Raw Count	Raw Density	Modeled Density
Wyoming Basin	Undisturbed	206	7931	237	0.03	0.17 (0.13–0.23)
Wyoming Basin	Well Pad	942	36267	1035	0.03	0.16 (0.13–0.20)
Wyoming Basin	Road	176	6776	216	0.03	0.15 (0.11–0.21)

Leporid abundance was not strongly predicted by any vegetation covariate, a somewhat surprising finding given leporid use of shrubs and grasses for food (Hansen and Gold, 1977; Simes et al., 2015) and cover (Jimenez et al., 2020). Instead, the most supported predictors of leporid abundance were site exposure (heat index), and longitude of the transect as well as year and disturbance type. Differences in site exposure result from fine-scale topographic features, such as differences in slope, aspect, or surface roughness, and may influence local temperature and vegetation (Bennie et al., 2008; Moeslund et al., 2013). For leporids, lower heat exposure appeared to facilitate use of well pads, while sites with greater heat exposure had low abundance when disturbed but higher abundance when undisturbed. Given our finding that well pads had lower vegetation cover than undisturbed transects, this may be a thermo-regulatory choice (Hinds, 1973), with animals selecting undisturbed sites with more cover in hot areas (Harrison, 2019; Milling et al., 2017) and tolerating more open conditions at well pads only when site conditions were cooler. Given the apparent lack of leporid response to vegetation, however, the driver behind the otherwise similarly high abundance at undisturbed and well pad transects, and low abundance at roads, is not clear. One possibility is that structures on the well pad, such as tanks, pumps, or sheds, provide good places to hide (Bock et al., 2006) in areas where vegetative cover is low, an ideal situation for detecting and avoiding predators (Palomo-Munoz et al., 2024). In support of this idea, during surveys we noted 10 leporid detections around culverts, shipping containers, or storage containers on well pads.

Differences in prey density around roads and well pads contributed to a significant cost of energy infrastructure disturbance on prey

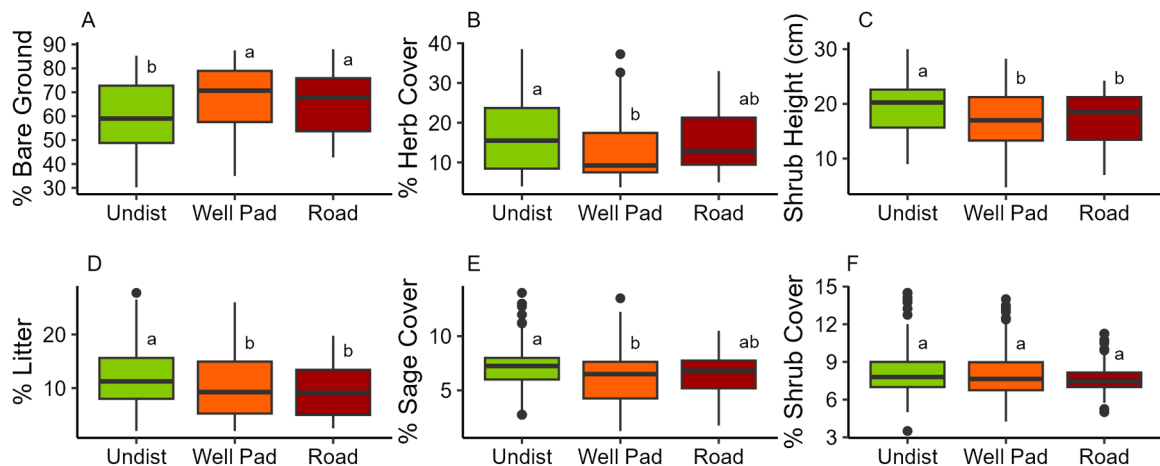


Fig. 7. Boxplots showing ANOVA comparisons of remotely-sensed vegetation metrics for A) percent bare ground, B) percent herbaceous cover, C) shrub height, D) percent vegetative litter, E) percent sage cover, and F) percent shrub cover at prey sample points on transects at undisturbed areas (Undist), well pads, and roads in Wyoming, USA. Lowercase letters above the boxplots indicate no significant pairwise difference if the letters are the same between disturbance types.

abundance, compared to theoretically undisturbed areas, when calculated at the larger sample site-level scale. Our models predict that leporid abundance was 18% lower given current energy disturbance conditions than it would be in an undisturbed area, and sciurid species were approximately 29% lower. Counterfactual methods, such as we used here, are well suited for assessing impacts of anthropogenic factors like energy development (Katzner et al., 2022). While counterfactual scenarios do not show a direct cause and effect relationship between disturbance and prey abundance, they provide a way to quantify the cost of an action, such as continued development, on a biological metric, such as prey availability (Grace et al., 2021). The amount of energy-related surface disturbance in our sample sites covered roughly half of their area, resulting in a significant decrease in prey abundance and demonstrating the already substantial cost of existing energy development in the area. Future development that further reduces undisturbed areas will likely continue to increase this loss.

Prey abundance was particularly different between the western and eastern halves of our study area given differences in the distribution of some prey species. We detected few sciurids in eastern sample sites, and the known distribution of sciurids differed between east and west Wyoming (Keinath et al., 2010). The primary species of sciurids detected in the west, white-tailed prairie dogs and Wyoming ground squirrels, do not occur in the eastern half of the study, while the species that do occur, least chipmunks and black-tailed prairie dogs, are either physically small or extremely clumpy in distribution, which we speculate may make it harder for predators to make their living off these species. Western sample sites may contain more readily available food resources for species like ferruginous hawks (Collins and Reynolds, 2005; Olson et al., 2017), which can have consequences for their successful reproduction and continued occupancy (Wallace, Kennedy, Squires, Oakleaf, et al., 2016). Anecdotally, we witnessed a bout of predation on lark bunting (*Calamospiza melanocorys*) fledglings by ferruginous hawks in the eastern portion of our study area, with 3 nests that we visited during June containing bunting remains (J. Squires, personal communication), indicating that ferruginous hawks in eastern Wyoming may pursue more varied prey due to a lack of small mammals. Abundance between individual sample sites also differed due to unexplained variation for both leporids and sciurids, as quantified by the variance in model random effects, indicating that prey may be patchily distributed across the landscape in relation to heterogeneity in factors we did not include in our study.

Site-level abundance was lower for sciurids (0.16 animals/ha) than leporids (0.28 animal/ha), averaged across all sample sites in which they occurred. Our estimates of sciurid abundance are limited to those individuals that were above ground and available for detection at the time of sampling, only a portion of the total population; thus, the actual density of sciurids in our study area is likely higher than our relative estimate. For sciurids, few estimates of density for these species in this study area are available. A 1970 study of Wyoming ground squirrels near Laramie in southeastern Wyoming found local densities of 0.21/ha to 1.18/ha (Clark, 1970), our previous study from 2010 to 2012 found 0.09/ha for Wyoming ground squirrels, 0.20/ha for white-tailed prairie dogs, and 0.02/ha for least chipmunks (Olson et al., 2017), and a 1988 study at 18 white-tailed prairie dog towns near Laramie, WY, found a range of densities from < 1/ha to 12/ha (Menkens et al., 1990). These studies used a variety of methods, however, including simple number captured per acre, distance sampling, and mark-recapture, respectively, which may result in very different estimates (Iijima, 2020). While most of our prey species are not highly social (Armitage, 1981) and detections occurred singly, white-tailed prairie dogs are loosely colonial (Hoogland, 1981), and thus our detections of them were patchy and infrequently occurred in large groups (groups ≥ 10 were detected 8 times out of 612 unique detections). Therefore, our relative sciurid density estimate is meant as an average of several species over a large area and does not reflect localized highs. Our estimated leporid density appears to be in line with historical estimates of leporid species, but may be higher than current estimates, given vast declines in multiple leporid species' abundance since the 1980s (Brown et al., 2018). Black-tailed jackrabbits in a 1959 study in Nevada had a density of 0.13/ha to 0.22/ha in June and 0.05/ha in August (Hayden, 1966), and an average spring density of 0.14/ha from 1963 to 1970 in Utah (Gross et al., 1974). Our

previous work (Olson et al., 2017) at a larger spatial extent in Wyoming from 2010 to 2012 found a combined density of all detected leporid species of 0.03/ha.

5.1. Implications

As primary prey species for multiple raptors of conservation concern (MacLaren et al., 1988) as well as mammalian mesopredators (Pechacek et al., 2000) our study has implications for raptor and carnivore management in Wyoming in areas impacted by oil and gas infrastructure (Wiggins et al., 2017). Both leporids and sciurids had lowest estimated densities along roads, and sciurids had low densities around well pads as well. Current energy-related disturbance levels compared to a simulated lack of disturbance at a given sample site resulted in an estimated 18% fewer leporids and 29% fewer sciurids, a substantial potential difference for predators. Thus, designing energy fields to minimize surface disturbance and maintain undisturbed areas is important for maintaining prey populations. Maintaining a matrix of adequate undisturbed areas in energy fields may be particularly important for predators with small home ranges, such as black-footed ferrets (Jachowski et al., 2010), since these may be affected by increased surface disturbance at a home range level. The roads used in our study were largely unpaved, energy-related or local roads, and we observed inconsistent traffic on them. Further work is needed to determine whether lower abundance was due to avoidance of traffic noise (Kleist et al., 2021), decreased habitat suitability (Gelbard and Harrison, 2003), or direct mortality (Dean et al., 2019). Abundance around well pads was also weakly lower for leporids and lower for sciurids. Our work supports the hypothesis that this reduced abundance may be due to vegetation alteration around disturbances, and thus revegetation with native shrub and herbaceous species around disturbed areas may help maintain a more abundant prey population.

Funding

This work was funded by the Bureau of Land Management, Wyoming Game and Fish Commission, and United States Forest Service Rocky Mountain Research Station.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgements

We thank the many technicians that helped collect this data and the Wyoming Game and Fish for logistical support.

Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at [doi:10.1016/j.gecco.2026.e04332](https://doi.org/10.1016/j.gecco.2026.e04332).

Data availability

Data will be made available on request.

References

- AdaptWest Project, 2015. Gridded current and projected climate data for North America at 1km resolution, interpolated using the ClimateNA v5.10 software. adaptwest.databasin.org.
- Akaike, H., 1974. A new look at the statistical model identification. *IEEE Trans. Autom. Control* 19 (6), 716–723.
- Allred, B.W., Smith, W.K., Twidwell, D., Haggerty, J.H., Running, S.W., Naugle, D.E., Fuhlendorf, S.D., 2015. Ecosystem services lost to oil and gas in North America. *Science* 348 (6233), 401–402. <https://doi.org/10.1126/science.aaa4785>.
- Armitage, K.B., 1981. Sociality as a life-history tactic of ground squirrels. *Oecologia* 48 (1), 36–49.
- Baker, B.W., Augustine, D.J., Sedgwick, J.A., Lubow, B.C., 2013. Ecosystem engineering varies spatially: a test of the vegetation modification paradigm for prairie dogs. *Ecography* 36 (2), 230–239. <https://doi.org/10.1111/j.1600-0587.2012.07614.x>.
- Balice, R.G., Miller, J.D., Oswald, B.P., Edminister, C., Yool, S.R., 2000. Forest surveys and wildfire assessment in the Los Alamos region; 1998–1999.: Vol. LA-13714-M.
- Baylis, K., Honey-Rosés, J., Börner, J., Corbera, E., Ezzine-de-Blas, D., Ferraro, P.J., Lapeyre, R., Persson, U.M., Pfaff, A., Wunder, S., 2016. Mainstreaming impact evaluation in nature conservation. *Conserv. Lett.* 9 (1), 58–64. <https://doi.org/10.1111/conl.12180>.
- Bennie, J., Huntley, B., Wiltshire, A., Hill, M.O., Baxter, R., 2008. Slope, aspect and climate: Spatially explicit and implicit models of topographic microclimate in chalk grassland. *Ecol. Model.* 216 (1), 47–59. <https://doi.org/10.1016/j.ecolmodel.2008.04.010>.
- Bock, C.E., Jones, Z.F., Bock, J.H., 2006. Abundance of cottontails (*Sylvilagus*) in an exurbanizing southwestern savanna. *Southwest. Nat.* 51 (3), 352–357. [https://doi.org/10.1894/0038-4909\(2006\)51\[352:AOC5IA\]2.0.CO;2](https://doi.org/10.1894/0038-4909(2006)51[352:AOC5IA]2.0.CO;2).
- Brown, D.E., Beatty, G., Brown, J.E., Smith, A.T., 2018. History, status, and population trends of cottontail rabbits and jackrabbits in the Western United States. *West. Wildl.* 5, 16–42.
- Buckland, S.T., 1985. Perpendicular distance models for line transect sampling. *Biometrics* 41 (1), 177–195. <https://doi.org/10.2307/2530653>.

- Buckland, S.T., Anderson, D.R., Burnham, K.P., Laake, J.L., Borchers, D.L., Thomas, L., 2001. *Introduction to distance sampling*. Oxford University Press.
- Burnett, R.D., Roberts, L.J., 2015. A quantitative evaluation of the conservation umbrella of spotted owl management areas in the sierra nevada. *PLoS. ONE* 10 (4), 1–18. <https://doi.org/10.1371/journal.pone.0123778>.
- Burnham, K.P., Anderson, D.R., 2002. *Model selection and multimodel inference: a practical information-theoretic approach*. Springer-Verlag.
- Chalfoun, A.D., 2021. Responses of vertebrate wildlife to oil and natural gas development: patterns and frontiers. *Curr. Landsc. Ecol. Rep.* 6 (3), 71–84. <https://doi.org/10.1007/s40823-021-00065-0>.
- Chang, J., Clais, P., Gasser, T., Smith, P., Herrero, M., Havlík, P., Obersteiner, M., Guenet, B., Goll, D.S., Li, W., Naipal, V., Peng, S., Qiu, C., Tian, H., Viovy, N., Yue, C., Zhu, D., 2021. Climate warming from managed grasslands cancels the cooling effect of carbon sinks in sparsely grazed and natural grasslands. *Nat. Commun.* 12 (118), 1–10. <https://doi.org/10.1038/s41467-020-20406-7>.
- Clark, T.W., 1970. Richardson's ground squirrel (*Spermophilus richardsonii*) in the Laramie Basin, Wyoming. *Gt. Basin Nat.* 30, 55–70.
- Clark, T.W., Hoffmann, R.S., Nadler, C.F., 1971. *Cynomys leucurus*. *Mamm. Species* 7, 1–4.
- Collins, C.P., Reynolds, T.D., 2005. Ferruginous Hawk (*Buteo regalis*): a technical conservation assessment. USDA Forest Service, Rocky Mountain Region. http://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5182004.pdf.
- Copeland, H.E., Doherty, K.E., Naugle, D.E., Pocewicz, A., Kiesecker, J.M., 2009. Mapping oil and gas development potential in the US intermountain west and estimating impacts to species. *PLoS. ONE* 4 (10), e7400. <https://doi.org/10.1371/journal.pone.0007400>.
- Davies, K.W., Boyd, C.S., Beck, J.L., Bates, J.D., Svejcar, T.J., Gregg, M.A., 2011. Saving the sagebrush sea: an ecosystem conservation plan for big sagebrush plant communities. *Biol. Conserv.* 144 (11), 2573–2584. <https://doi.org/10.1016/j.biocon.2011.07.016>.
- Davies, K.W., Boyd, C.S., Bates, J.D., Svejcar, L.N., Porensky, L.M., 2024. Ecological benefits of strategically applied livestock grazing in sagebrush communities. *Ecosphere* 15 (5), 1–15. <https://doi.org/10.1002/ecs2.4859>.
- Dean, W.R.J., Seymour, C.L., Joseph, G.S., Foord, S.H., 2019. A review of the impacts of roads on wildlife in semi-arid regions. *Diversity* 11 (5), 1–19. <https://doi.org/10.3390/D11050081>.
- DiTomaso, J.M., 2000. Invasive weeds in rangelands: species, impacts, and management. *Weed Sci.* 48, 255–265.
- Evans, J.S., Oakleaf, J., Cushman, S.A., Theobald, D., 2014. An ArcGIS toolbox for surface gradient and geomorphometric modeling, version 2.0. <https://evansmurfphy.wixsite.com/evansspatial/arcgis-gradient-metrics-toolbox>.
- Fedy, B.C., Doherty, K.E., Aldridge, C.L., O'Donnell, M., Beck, J.L., Bedrosian, B., Gummer, D., Holloran, M.J., Johnson, G.D., Kaczor, N.W., Kirol, C.P., Mandich, C.A., Marshall, D., McKee, G., Olson, C., Pratt, A.C., Swanson, C.C., Walker, B.L., 2014. Habitat prioritization across large landscapes, multiple seasons, and novel areas: an example using greater sage-grouse in Wyoming. *Wildl. Monogr.* 190 (1), 1–39. <https://doi.org/10.1002/wmon.1014>.
- Fiehrer, C.M., Cypher, B.L., Saslaw, L.R., 2017. Effects of oil and gas development on vertebrate community composition in the southern San Joaquin Valley, California. *Glob. Ecol. Conserv.* 9, 131–141. <https://doi.org/10.1016/j.gecco.2017.01.001>.
- Finn, S.P., Knick, S.T., 2011. Changes to the Wyoming Basins landscape from oil and natural gas development. In: Hanser, S.E., Leu, M., Knick, S.T., Aldridge, C.L. (Eds.), *Sagebrush Ecosystem Conservation and Management: ecoregional assessment tools and models for the Wyoming Basins*. Allen Press, pp. 69–87.
- Fiske, I., Chandler, R.B., 2011. Unmarked: an R package for fitting hierarchical models of wildlife occurrence and abundance. *J. Stat. Softw.* 43, 1–23. <http://www.jstatsoft.org/v43/i10/>.
- Gelbard, J.L., Harrison, S., 2003. Roadless habitats as refuges for native grasslands: Interactions with soil, aspect, and grazing. *Ecol. Appl.* 13 (2), 404–415. [https://doi.org/10.1890/1051-0761\(2003\)013\[0404:RHARFN\]2.0.CO;2](https://doi.org/10.1890/1051-0761(2003)013[0404:RHARFN]2.0.CO;2).
- Grace, M.K., Akçakaya, H.R., Bull, J.W., Carrero, C., Davies, K., Hedges, S., Hoffmann, M., Long, B., Lughadha, E.M.N., Martin, G.M., Pilkington, F., Rivers, M.C., Young, R.P., Milner-Gulland, E.J., 2021. Building robust, practicable counterfactuals and scenarios to evaluate the impact of species conservation interventions using inferential approaches. *Biol. Conserv.* 261, 109259. <https://doi.org/10.1016/j.biocon.2021.109259>.
- Green, A.W., Aldridge, C.L., O'Donnell, M.S., 2017. Investigating impacts of oil and gas development on greater sage-grouse. *J. Wildl. Manag.* 81 (1), 46–57. <https://doi.org/10.1002/jwmg.21179>.
- Gross, J.E., Stoddard, C.L., Wagner, F.H., 1974. Demographic analysis of a northern Utah jackrabbit population. *Wildl. Monogr.* 40, 3–68.
- Grushecky, S.T., Zinkhan, F.C., Strager, M.P., Carr, T., 2022. Energy production and well site disturbance from conventional and unconventional natural gas development in West Virginia. *Energy Ecol. Environ.* 7 (4), 358–368. <https://doi.org/10.1007/s40974-022-00246-5>.
- Hannon, M.J., Jenkins, S.H., Crabtree, R.L., Swanson, A.K., 2006. Visibility and vigilance: behavior and population ecology of Uinta ground squirrels (*Spermophilus armatus*) in different habitats. *J. Mammal.* 87 (2), 287–295. <https://doi.org/10.1644/05-MAMM-A-081R2.1>.
- Hansen, R.M., Gold, I.K., 1977. Blacktail prairie dogs, desert cottontails and cattle trophic relations on shortgrass range. *J. Range Manag.* 30 (3), 210. <https://doi.org/10.2307/3897472>.
- Harrison, R.L., 2019. A comparison of diurnal resting sites used by *Sylvilagus audubonii* and *Lepus californicus* in the Chihuahuan Desert. *West. North. Am. Nat.* 79 (2), 170–184. <https://doi.org/10.3398/064.079.0204>.
- Hayden, P., 1966. Seasonal Occurrence of Jackrabbits on Jackass Flat, Nevada. *J. Wildl. Manag.* 30 (4), 835–838.
- Hinds, D.S., 1973. Acclimatization of thermoregulation in the desert cottontail, *Sylvilagus audubonii*. *J. Mammal.* 54 (3), 708–728. <https://doi.org/10.2307/1378969>.
- Hoekstra, J.M., Boucher, T.M., Ricketts, T.H., Roberts, C., 2004. Confronting a biome crisis: global disparities of habitat loss and protection. *Ecol. Lett.* 8 (1), 23–29. <https://doi.org/10.1111/j.1461-0248.2004.00686.x>.
- Hoffman, S.W., Smith, J.P., 2003. Population trends of migratory raptors in western North America, 1977–2001. *Condor* 105 (3), 397–419. <https://doi.org/10.1650/7146>.
- Homer, C.G., Aldridge, C.L., Meyer, D.K., Schell, S.J., 2012. Multi-scale remote sensing sagebrush characterization with regression trees over Wyoming, USA: laying a foundation for monitoring. *Int. J. Appl. Earth Obs. Geoinf.* 14 (1), 233–244. <https://doi.org/10.1016/j.jag.2011.09.012>.
- Hoogland, J.L., 1981. The evolution of coloniality in white-tailed and black-tailed prairie dogs. *Ecology* 62 (1), 252–272.
- Houston, C.S., Bechard, J., 1984. Decline of the Ferruginous Hawk in Saskatchewan. *Am. Birds* 38 (2), 166–170.
- Hufford, K., Moine, J., Gamo, R.S., 2023. *Revegetation success and weed resilience of Wyoming right-of-way reclamation (Issue Final Report WY2304F)*. Wyoming Department of Transportation.
- Iijima, H., 2020. A review of wildlife abundance estimation models: comparison of models for correct application. *Mammal. Study* 45 (3), 177–188. <https://doi.org/10.3106/ms2019-0082>.
- Iverson, L.R., Dale, M.E., Scott, C.T., Prasad, A., 1997. A GIS-derived integrated moisture index to predict forest composition and productivity of Ohio forests (USA). *Landsc. Ecol.* 12, 331–348.
- Jachowski, D.S., Millsaugh, J.J., Biggins, D.E., Livieri, T.M., Matchett, M.R., 2010. Home-range size and spatial organization of black-footed ferrets *Mustela nigripes* in South Dakota, USA. *Wildl. Biol.* 16 (1), 66–76. <https://doi.org/10.2981/09-034>.
- Jimenez, A., Schmalz, J., Wright, M.N., Skopec, M.M., 2020. Sagebrush characteristics influencing foraging patterns of pygmy rabbits. *J. Wildl. Manag.* 84 (7), 1306–1314. <https://doi.org/10.1002/jwmg.21923>.
- Johnson, T.N., Nasman, K., Wallace, Z.P., Olson, L.E., Squires, J.R., Nielson, R.M., Kennedy, P.L., 2019. Survey design for broad-scale, territory-based occupancy monitoring of a raptor: Ferruginous hawk (*Buteo regalis*) as a case study. *PLoS. ONE* 14 (3), 1–22. <https://doi.org/10.1371/journal.pone.0213654>.
- Jones, I.L., Bull, J.W., Milner-Gulland, E.J., Esipov, A.V., Suttle, K.B., 2014. Quantifying habitat impacts of natural gas infrastructure to facilitate biodiversity offsetting. *Ecol. Evol.* 4 (1), 79–90. <https://doi.org/10.1002/ece3.884>.
- Katzner, T.E., Carlisle, J.D., Poessel, S.A., Thomason, E.C., Pauli, B.P., Pilliod, D.S., Belthoff, J.R., Heath, J.A., Parker, K.J., Warner, K.S., Hayes, H.M., Aberg, M.C., Ortiz, P.A., Amdor, S.M., Alsup, S.E., Coates, S.E., Miller, T.A., Duran, Z.K., 2020. Illegal killing of nongame wildlife and recreational shooting in conservation areas. *Conserv. Sci. Pract.* 2 (11), 1–15. <https://doi.org/10.1111/csp2.279>.
- Katzner, T.E., Allison, T.D., Diffendorfer, J.E., Hale, A.M., Lantz, E.J., Veers, P.S., 2022. Counterfactuals to assess effects to species and systems from renewable energy development. *Front. Conserv. Sci.* 3 (May), 1–4. <https://doi.org/10.3389/fcoss.2022.844286>.

- Keinath, D.A., Andersen, M.D., Beauvais, G.P., 2010. Range and modeled distribution of Wyoming's species of greatest conservation need. Report Prepared for the Wyoming Game and Fish Department. Cheyenne, Wyoming and the U.S. Geological Survey, Fort Collins, Colorado. <http://www.uwyo.edu/wyndd/reports-and-publications/index.html>.
- Kleist, N.J., Buxton, R.T., Lendrum, P.E., Linares, C., Crooks, K.R., Wittemyer, G., 2021. Noise and landscape features influence habitat use of mammalian herbivores in a natural gas field. *J. Anim. Ecol.* 90 (4), 875–885. <https://doi.org/10.1111/1365-2656.13416>.
- Knick, S.T., Dobkin, D.S., Rotenberry, J.T., Schroeder, M.A., Vander Haegen, W.M., van Riper, I.L.I., C., 2003. Teetering on the edge or too late? Conservation and research issues for avifauna of sagebrush habitats. *Condor* 105 (4), 611–634. <https://doi.org/10.1650/7329>.
- Linden, D.W., Fuller, A.K., Royle, J.A., Hare, M.P., 2017. Examining the occupancy–density relationship for a low-density carnivore. *J. Appl. Ecol.* 54 (6), 2043–2052. <https://doi.org/10.1111/1365-2664.12883>.
- Litt, A.R., Pearson, D.E., 2013. Non-native plants and wildlife in the intermountain west. *Wildl. Soc. Bull.* 37 (3), 517–526. <https://doi.org/10.1002/wsb.306>.
- Ludlow, S.M., Brigham, R.M., Davis, S.K., 2015. Oil and natural gas development has mixed effects on the density and reproductive success of grassland songbirds. *Condor* 117 (1), 64–75. <https://doi.org/10.1650/CONDOR-14-79.1>.
- MacLaren, P.A., Anderson, S.H., Runde, D.E., 1988. Food habits and nest characteristics of breeding raptors in southwestern Wyoming. *Gt. Basin Nat.* 48 (4), 548–553. <https://ojs.lib.byu.edu/ojs/index.php/wnan/article/viewArticle/1765>.
- Maguire, K., Tanner, S.J., Winikoff, J.B., Williams, R., 2024. Utility-scale solar and wind development in rural areas: Land cover change (2009–20). In: *Economic Research Report*, 330. U.S. Department of Agriculture. www.ers.usda.gov.
- McMahon, T., Larson, T., Zhang, T., Shuster, M., 2024. Geologic characteristics, exploration and production progress of shale oil and gas in the United States: an overview. *Pet. Explor. Dev.* 51 (4), 925–948. [https://doi.org/10.1016/S1876-3804\(24\)60516-1](https://doi.org/10.1016/S1876-3804(24)60516-1).
- Menkens, G.E., Biggins, D.E., Anderson, S.H., 1990. Visual counts as an index of white-tailed prairie dog density. *Wildl. Soc. Bull.* 18 (3), 290–296.
- Miller, S., Knight, R., Miller, C., 2001. Wildlife responses to pedestrians and dogs. *Wildl. Soc. Bull.* 29 (1), 124–132. <http://www.cabdirect.org/abstracts/20013079688.html>.
- Millington, C.R., Rachlow, J.L., Johnson, T.R., Forbey, J.S., Shipley, L.A., 2017. Seasonal variation in behavioral thermoregulation and predator avoidance in a small mammal. *Behav. Ecol.* 28 (5), 1236–1247. <https://doi.org/10.1093/beheco/axx084>.
- Millsap, B.A., Zimmerman, G.S., Kendall, W.L., Barnes, J.G., Braham, M.A., Bedrosian, B.E., Bell, D.A., Bloom, P.H., Crandall, R.H., Domenech, R., Driscoll, D., Duerr, A.E., Gerhardt, R., Gibbs, S.E.J., Harmata, A.R., Jacobson, K., Katzner, T.E., Knight, R.N., Lockhart, J.M., Watson, J.W., 2022. Age-specific survival rates, causes of death, and allowable take of golden eagles in the western United States. *Ecol. Appl.* 32 (3), 1–22. <https://doi.org/10.1002/eap.2544>.
- Moeslund, J.E., Arge, L., Bøcher, P.K., Dalgaard, T., Odgaard, M.V., Nygaard, B., Svenning, J.C., 2013. Topographically controlled soil moisture is the primary driver of local vegetation patterns across a lowland region. *Ecosphere* 4 (7), 1–26. <https://doi.org/10.1890/ES13-00134.1>.
- Morin, D.J., Yackulic, C.B., Diffendorfer, J.E., Lesmeister, D.B., Nielsen, C.K., Reid, J., Schaubert, E.M., 2020. Is your ad hoc model selection strategy affecting your multimodel inference? *Ecosphere* 11 (1). <https://doi.org/10.1002/ecs2.2997>.
- Nakagawa, S., Cuthill, I.C., 2007. Effect size, confidence interval and statistical significance: a practical guide for biologists. *Biol. Rev.* 82 (4), 591–605. <https://doi.org/10.1111/j.1469-185X.2007.00027.x>.
- Nasen, L.C., Noble, B.F., Johnstone, J.F., 2011. Environmental effects of oil and gas lease sites in a grassland ecosystem. *J. Environ. Manag.* 92 (1), 195–204. <https://doi.org/10.1016/j.jenvman.2010.09.004>.
- Northrup, J.M., Wittemyer, G., 2013. Characterising the impacts of emerging energy development on wildlife, with an eye towards mitigation. *Ecol. Lett.* 16 (1), 112–125. <https://doi.org/10.1111/ele.12009>.
- O'donnell, M.S., Fancher, T.S., Freeman, A.T., Ziegler, A.E., Bowen, Z.H., Aldridge, C.L., 2014. Large scale Wyoming transportation data—A resource planning tool. <https://pubs.usgs.gov/publication/ds821>.
- O'Garra, T., Martin, R., Pynegar, E., Polo-Urrea, C., Eklund, J., 2025. Selecting among counterfactual methods to evaluate conservation interventions. *Conserv. Sci. Pract.* 7 (11), 1–18. <https://doi.org/10.1111/csp.270066>.
- Olson, L.E., Oakleaf, R.J., Squires, J.R., Wallace, Z.P., Kennedy, P.L., 2015. Nesting pair density and abundance of Ferruginous Hawks (*Buteo regalis*) and Golden Eagles (*Aquila chrysaetos*) from aerial surveys in Wyoming. *J. Raptor Res.* 49 (4), 400–412.
- Olson, L.E., Squires, J.R., Oakleaf, R.J., Wallace, Z.P., Kennedy, P.L., 2017. Predicting above-ground density and distribution of small mammal prey species at large spatial scales. *PLoS ONE* 12 (5), 1–21. <https://doi.org/10.1371/journal.pone.0177165>.
- Ostojia, S.M., Schupp, E.W., 2009. Conversion of sagebrush shrublands to exotic annual grasslands negatively impacts small mammal communities. *Divers. Distrib.* 15 (5), 863–870. <https://doi.org/10.1111/j.1472-4642.2009.00593.x>.
- Palomo-Munoz, G., Fidino, M., Werdell, T.J., Piper, C.W., Gallo, T., Peek, M.S., Ricketts, A.M., Ahlers, A.A., 2024. Mesopredators have differing influences on prey habitat use and diel activity in a multipredator landscape. *Ecosphere* 15 (9), 1–17. <https://doi.org/10.1002/ecs2.70006>.
- Pechacek, P.E., Lindzey, F.R.G., Anderson, S.T.H., 2000. Autumn and winter diet of the swift fox (*Vulpes velox*) in south-eastern Wyoming. *Hystrix* 11 (2), 83–87. <http://www.italian-journal-of-mammalogy.it/index.php/hystrix/article/viewArticle/4153>.
- R Core Team (2023). R: A language and environment for statistical computing. In: *R: A language and environment for statistical computing*. (4.3.2; p. URL <https://www.R-project.org/>). R Foundation for Statistical Computing, Vienna, Austria. <https://www.R-project.org/>.
- Remington, T.E., Deibert, P.A., Hanser, S.E., Davis, D.M., Robb, L.A., Welty, J.L., 2021. Sagebrush conservation strategy—challenges to sagebrush conservation. U.S. Geological Survey Open-File Report 1125, 327. <https://doi.org/10.3133/ofr2021125>.
- Samson, F., Knopf, F., 1994. Prairie conservation in North America. *BioScience* 44 (6), 418–421.
- Sanders, L.E., Chalfoun, A.D., 2018. Novel landscape elements within natural gasfields increase densities but not fitness of an important songbird nest predator. *Biol. Conserv.* 228 (October), 132–141. <https://doi.org/10.1016/j.biocon.2018.10.020>.
- Sandoval Lambert, M., Sawyer, H., Merkle, J.A., 2022. Responses to natural gas development differ by season for two migratory ungulates. *Ecol. Appl.* 32 (7), 1–13. <https://doi.org/10.1002/eap.2652>.
- Sawyer, H., Kauffman, M.J., Nielson, R.M., Horne, J.S., 2009. Identifying and prioritizing ungulate migration routes for landscape-level conservation. *Ecol. Appl.* 19 (8), 2016–2025. <https://doi.org/10.1890/08-2034.1>.
- Scholtz, R., Twidwell, D., 2022. The last continuous grasslands on Earth: Identification and conservation importance. *Conserv. Sci. Pract.* 4 (3), 1–20. <https://doi.org/10.1111/csp2.626>.
- Simes, M.T., Longshore, K.M., Nussear, K.E., Beatty, G.L., Brown, D.E., Esque, T.C., 2015. Black-tailed and white-tailed jackrabbits in the American west: history, ecology, ecological significance, and survey methods. *West. North. Am. Nat.* 75 (4), 491–519. <https://doi.org/10.3398/064.075.0406>.
- Sirimarco, X., Barral, M.P., Villarino, S.H., Laterra, P., 2018. Water regulation by grasslands: a global meta-analysis. *Ecohydrology* 11 (4), 1–9. <https://doi.org/10.1002/eco.1934>.
- Squires, J.R., Olson, L.E., Wallace, Z.P., Oakleaf, R.J., Kennedy, P.L., 2020. Resource selection of apex raptors: implications for siting energy development in sagebrush and prairie ecosystems. *Ecosphere* 11 (8). <https://doi.org/10.1002/ecs2.3204>.
- Stewart, K.M., Bowyer, R.T., Kie, J.G., Hurley, M.A., 2010. Spatial distributions of mule deer and North American Elk: resource partitioning in a sage-steppe environment. *Am. Midl. Nat.* 163 (2), 400–412.
- Tablado, Z., Jenni, L., 2017. Determinants of uncertainty in wildlife responses to human disturbance. *Biol. Rev.* 92 (1), 216–233. <https://doi.org/10.1111/brv.12224>.
- Thomason, E.C., Turley, N.J.S., Belthoff, J.R., Conkling, T.J., Katzner, T.E., 2023. Illegal shooting is now a leading cause of death of birds along power lines in the western USA. *IScience* 26 (8), 107274. <https://doi.org/10.1016/j.isci.2023.107274>.
- Thompson, C.M., Gese, E.M., 2013. Influence of vegetation structure on the small mammal community in a shortgrass prairie ecosystem. *Acta Theriol.* 58 (1), 55–61. <https://doi.org/10.1007/s13364-012-0098-5>.
- Trainor, A.M., McDonald, R.I., Fargione, J., 2016. Energy sprawl is the largest driver of land use change in United States. *PLoS ONE* 11 (9), 1–16. <https://doi.org/10.1371/journal.pone.0162269>.

- U.S. Geological Survey, 2019. The National Map—New data delivery homepage, advanced viewer, lidar visualization. US Geological Survey Fact Sheet (Fact Sheet). <https://doi.org/10.3133/fs20193032>.
- van der Marel, A., Waterman, J.M., López-Darias, M., 2021. Barbary ground squirrels do not have a sentinel system but instead synchronize vigilance. *Behav. Ecol. Sociobiol.* 75 (11), 1–10. <https://doi.org/10.1007/s00265-021-03094-1>.
- Wallace, Z.P., Kennedy, P.L., Squires, J.R., Oakleaf, R.J., Olson, L.E., Dugger, K.M., 2016. Re-occupancy of breeding territories by Ferruginous Hawks in Wyoming: relationships to environmental and anthropogenic factors. *Plos. One* 11 (4), e0152977. <https://doi.org/10.1371/journal.pone.0152977>.
- Wallace, Z.P., Kennedy, P.L., Squires, J.R., Olson, L.E., Oakleaf, R.J., 2016. Human-made structures, vegetation, and weather influence ferruginous hawk breeding performance. *J. Wildl. Manag.* 80 (1), 78–90. <https://doi.org/10.1002/jwmg.1000>.
- Walsh, K.B., Rose, J., 2022. A review of restoration techniques and outcomes for rangelands affected by oil and gas production in North America. *Ecol. Restor.* 40 (4), 259–269. <https://doi.org/10.3368/er.40.4.259>.
- Wang, T., Hamann, A., Spittlehouse, D., Carroll, C., 2016. Locally downscaled and spatially customizable climate data for historical and future periods for North America. *PLoS. ONE* 11 (6), e0156720. <https://doi.org/10.1371/journal.pone.0156720>.
- Wiggins, D.A., Grzybowski, J.A., Schnell, G.D., 2017. Ferruginous hawk demography in areas differing in energy extraction activity. *J. Wildl. Manag.* 81 (2), 337–341. <https://doi.org/10.1002/jwmg.21194>.
- WOGCC, 2015. Wyoming Oil and Gas Conservation Commission Homepage. <http://wogcc.state.wy.us/>.
- WYNDD, 2024. Wyoming Natural Diversity Database. Data Explor. Map View. https://wyndd.org/portal/apps/data_explorer/map.
- Zegers, D.A., 1984. *Spermophilus elegans*. *Mamm. Species* (214), 1–7. <https://doi.org/10.2307/3503955>.