

Landscape composition and proximity to water structure American badger (*Taxidea taxus*)
distributions in shortgrass prairies

by

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Abstract

Widespread land-use changes in the Great Plains have resulted in a patchy mosaic of prairie embedded within human-modified landscapes. The distribution of prairie-obligate species in this region may be constrained by these alternate land-use types, though many carnivore-specific examples are unknown. We used three years (2018-2020) of data collected from camera-trap sites ($n = 381$) in western Kansas, USA to assess multiscale effects of landscape change on the distribution of American badgers (badger, *Taxidea taxus*), an important predator and ecosystem engineer. We predicted initial site occupancy probabilities and colonization rates would be positively associated with the amount of prairie and Conservation Reserve Program (CRP) hectares surrounding sites. We also predicted site occupancy and colonization would be negatively associated with the amount of row crop agriculture and density of wind towers surrounding sites along with distance to permanent water sources. Habitat use by badgers was insensitive to the amount of prairie and CRP surrounding sites at both local and landscape scales. Contrary to our hypothesis, badgers were more likely to occupy and colonize sites with greater amounts of row-crop agriculture at both landscape and local scales. Additionally, badgers were less likely to occupy sites farther from permanent water sources. Our study suggests that badgers, although considered prairie-obligate carnivores, may be exploiting row crop agricultural areas because of increased prey densities or suitable burrowing/digging substrates. Moreover, our research highlights the importance of permanent water resources to badgers in arid regions within the Great Plains.

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Dedication

To my parents and grandmother, Rick and Donna Piper and Marcia Whitney, who raised me and supported me throughout my journey to pursue my dreams; to my amazing fiancé Ruben Amedee Jr. for his continued love and reassurance.

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Chapter 1 - Introduction

Grassland habitats occur on every continent across the globe and once made up approximately 40% of the world's landcover (Robertson et al. 1997; Burton et al. 1988; Singh et al. 1983). In North America, prairie is the most abundant form of natural grassland (Kuchler 1964). Prairie lands in North America once extended approximately 360 million hectares along the foothill of the Rocky Mountains from Canada south to New Mexico and Texas and as far eastward as Kentucky (Risser et al. 1981; Weaver 1954; Transeau 1935). In the 1830's, westward expansion of European settlers essentially marked the beginning of the transformation of native prairie lands into developed or working lands used to support the growing human population and livestock (Rollings 1995; Smith 1992; Mlot 1990). Approximately 556 million hectares of North America is now considered working land (i.e. roads, cities, agriculture, etc.) (USDA 2018). Of these working lands, roughly 27% was once native prairie habitat and 60% of the remaining prairie is at moderate or high risk of becoming working land (Olimb and Robinson 2019). The conversion of much of North America's native prairie lands has created a patchy mosaic and has drastically altered native habitat (Samson and Knopf 1996).

The Great Plains of North America provide key prairie habitat to a wide variety of vertebrate species, including some rare or endangered species (Perkins et al. 2019; Samson et al. 2004). Of these vertebrates, ~121 native mammalian species can be found throughout the extent of the Great Plains (Jones Jr. et al. 1983). The fragmentation of native prairie within the Great Plains gravely threatens the species richness and persistence of these mammals (Wilcove et al. 1998; Crooks 2002). Mammalian carnivores are particularly sensitive to habitat fragmentation caused by increased human activity because of their relatively large home range sizes, lower reproductive rates, and general victimization by humans (Woodruffe and Ginsburg 1998; Noss et

al. 1996). This sensitivity makes mammalian carnivores important topics of research to monitor intensity of habitat fragmentation across the landscape.

The American badger (*Taxidea taxus*) is an important mammalian carnivore and furbearer with an extensive geographic range (Long 1973). In the Great Plains, the American badger has been described as a ‘prairie-obligate’ species, as they require large tracts of treeless habitat to support their fossorial nature (Messick and Hornocker 1981; Sargeant and Warner 1972). Burrows created by badgers in prairie landscapes can be utilized by a variety of vertebrate species, and some invertebrate species, as refuge (Symes et al. 2019). The fragmentation of prairie habitat in the Great Plains can therefore be affecting the movement patterns of badgers and the population stability of badgers’ prey species or species that rely on badger burrows for shelter within this ecoregion. The role of the American badger as an apex predator and ecosystem engineer in the Great Plains make it an important research topic in terms of wildlife conservation. Information regarding the status and trends of badger populations in various ecoregions within its range can provide insight to the intensity of habitat fragmentation and the extent of its effects on natural ecosystems.

My thesis research focuses on the distribution of American badgers in a shortgrass prairie ecosystem in Western Kansas in relation to various characteristics of the landscape including both natural and human-altered habitats. I used trail cameras to determine presence/absence of badgers in varying habitat types throughout the western third of Kansas and a multiseason occupancy modeling approach to expose potential relationships between badger occupancy probabilities/ colonization rates and landcover types. To my knowledge, this is the most extensive study of dynamic badger occupancy in shortgrass prairie ecosystems to date.

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Chapter 2 - Landscape composition and proximity to water structure American badger (*Taxidea taxus*) distributions in shortgrass prairies

Introduction

The Great Plains were historically dominated by prairie landscapes that extended contiguously across 363 million ha in North America. Currently, prairie covers only a small portion of their historic range (~162 million ha) and continues to diminish (Samson and Knopf 1996; Lark et al. 2019). Increasing human activity post-colonialism in the Great Plains resulted in conversion of prairie to working lands (e.g., cities, roads, agriculture, energy production) with ~97.2 million ha of native prairie now devoted to crop land use alone (Olimb and Robinson 2019). Conversion of native landscapes to developed (e.g., urbanization, roads) or working lands has also resulted in habitat loss and fragmentation which is the leading threat to mammal conservation on a global scale (Schipper et al. 2008).

Carnivores, because of their large home-range requirements and potential for negative human interactions, are especially sensitive to human-modified landscapes (Bruskotter et al. 2017; Sálek et al. 2015). There is species-specific variation, however, in response to various landuse changes. Previous studies suggest human development negatively influences distributions of some native carnivore species while positively affecting others (Moreira-Arce et al. 2015; Hilty et al. 2006; Crooks 2002). Additionally, carnivore species occurring in areas of increased human activity may have an increased risk of mortality (e.g., collisions with vehicles, conflict with ranchers over livestock; Van Eeden et al. 2018 Červinka et al. 2015). Although energy development (e.g., wind farms) is widespread on the Great Plains, the effects of these

activities remain uncertain for many carnivore species. However, Kuvlesky et al. (2007) suggest the “footprint” of wind farms (area of habitat disturbed by construction) could negatively affect species’ survival and distributions because of habitat alteration and fragmentation.

American badger (*Taxidea taxus*, hereafter ‘badger’) is an important predator and ecosystem engineer throughout its range and requires large, treeless landscapes for home-range establishment and maintenance (Messick and Hornocker 1981; Apps et al. 2002; Hoodicoff et al. 2009). Badgers are widely distributed throughout North America and occur in regions associated with short, mixed, and tallgrass prairies (Messick 1987). Badgers are medium-sized carnivores (6 -15 kg and 60-73 cm) and adapted as fossorial predators (Long 1973). Reported home-range sizes of badgers vary (2 - 30 km²; Lindzey 1978; Doyle et al. 2019) and space use is likely related to patches of grassland with friable soils and increased prey densities (Feldhamer et al. 2003). The limited number of studies evaluating resource selection of badgers suggest positive associations between home-range sizes and amount of available prairie, pasture, and cropland landcover and prey occurrence (e.g., Kinley and Newhouse 2008; Duquette and Gehrt 2014; Kinley et al. 2014). Common prey items include rabbits (Leporidae), gophers (Geomysidae), and ground squirrels (Sciuridae; Tumlison et al. 2012) along with available arthropods and amphibians (Tumlsion and Surf 2016). Duquette and Gehrt (2014) suggested badgers select for row crop agricultural landscapes because of availability of loose soils for burrows. This effect, however, may be due, in part, to the widespread coverage of row crop agriculture present within the study area (Illinois and Ohio, USA; Duquette and Gehrt [2014]).

Badgers are widely distributed throughout remaining semiarid prairies, though effects of landscape changes on badger distributions are largely unknown. In western Kansas, USA, a region historically characterized by shortgrass prairie landscapes, landuse has largely shifted to

agricultural production (e.g., row-crop agriculture, livestock) and energy development (e.g., wind farms). Approximately 1.06 million ha of native grasslands in western Kansas have been converted to agriculture (Egbert et al. 2001). Current efforts at prairie restoration in this region include the Conservation Reserve Program (CRP) which has converted ~122,206 ha of agriculture into prairie (Peterson et al. 2004). The conservation benefit of CRP is well documented with bird populations (e.g., Sullins et al. 2018; Drum et al. 2015; Blank 2013). It remains unclear, however, how the distribution or abundance of CRP affects badger distributions. Western Kansas is a semi-arid region and annual precipitation is lower than the amount of water that is evaporated or transpired by vegetation annually (Araya et al. 2017). It is plausible that badgers, along with other mammals, occurring in this region are likely sensitive to the distribution of non-ephemeral water resources (Krausman et al. 2006).

We used three years of presence/absence data and multiseason occupancy models to assess how landscape changes affect multiscale habitat use by badgers in western Kansas. Because badgers are considered prairie-obligate carnivores, we predicted initial site occupancy probabilities and colonization rates would be positively related to the amount of prairie and Conservation Reserve Program (CRP) landcover surrounding sites. We also predicted initial site occupancy probabilities and colonization rates would be negatively related to the amount of row crop agriculture surrounding sites and also to the number of wind turbines surrounding sites. Because our study area occurs in a semiarid region, we also predicted greater initial occupancy probabilities and colonization rates at sites closer to permanent water sources.

Methods and Materials

Study Area

Our study area included the western third of Kansas, USA (~ 7,180,273 ha, centered on 38.539557N, 100.676007W; Fig. 1) and expanded across four ecoregions (High Plains, Smoky Hills, Arkansas River Lowlands, and the Red Hills; EPA Ecoregions). Historically, this area was dominated by native prairie but is now largely devoted to agriculture and energy use (e.g., wind farms, oil extraction). The majority of landcover (41%) included row crop agriculture (corn [16%, *Zea maize*], sorghum [8%, *Sorghum bicolor*], and wheat [17%, *Triticum aestivum*]), followed by mixed-grass (15%) and shortgrass prairie (13%). Native prairie grass communities were characterized by big bluestem (*Andropogon gerardii*), broomsedge (*Andropogon virginicus*), side-oats grama (*Bouteloua curtipendula*), buffalo grass (*Bouteloua dactyloides*), witchgrass (*Panicum capillare*), tumblegrass (*Schedonnardus paniculatus*), Indiangrass (*Sorghastrum nutans*). Annually, our study area received an average of 56cm of precipitation (range = 46cm – 66cm) and had an average temperature of 20°C (range = -9° - 49°C).

Site Selection

We randomly selected the spatial location of sites (location of a single camera trap) using the random-point generator in ArcMap 10.6 (Theobald et al. 2007). We adjusted the exact camera-trap location at sites to occur on properties accessible with permission from the landowner or organization. Additionally, we ensured that camera-trap locations occurred on areas with grass landcover. For instance, if our random point was located in an accessible agricultural field, we adjusted it to the nearest patch of grass adjacent to the field (e.g., fence row, field border). Once site locations were established in 2018 (n = 381), we attempted to

sample that site for the duration of the study (2019, 2020). We were not granted access to sample all sites each year (2019, n = 361; 2020, n = 360). All sites were separated by \geq 2km and we assumed each site was spatially independent.

Assessing Presence/ Absence

We documented presence/absence of badgers at each site during 2018 (05 May – 02 November), 2019 (20 May – 02 October), and 2020 (22 April – 28 September). At each site, we attached a single motion-sensor camera trap (Bushnell Trophy Cam®; Model 119436C) to a metal post \sim 40cm above the ground. We positioned a wooden stake 3m in front of the camera with the top of the stake flush with the camera lens. We smeared a mixture of Vaseline® and commercial skunk essence (F&T Fur Harvester's Trading Post, Alepna, MI, USA) onto the top of the wooden stake to lure badgers in front of the camera (Iannarilli et al. 2021). To minimize the potential negative influence of vegetation on view obstruction, we removed all vegetation in the field of view using a hand-held weed cutter. We set cameras to medium exposure taking photos at 30 second intervals once triggered. Cameras were active for 28 days at each site and we revisited each site at approximately two weeks (\sim 14 days) to replace batteries, memory cards (16 GB), and reapply scent.

Once memory cards were retrieved from each site, we transferred photo data to the Colorado Parks and Wildlife Photo Warehouse 4.0 (Colorado Parks and Wildlife, Fort Collins, CO, USA). We manually viewed all photos and considered a site occupied by badgers if we observed \geq 1 photo of a badger during the 28-day sampling period for a given year. From these data, we constructed 28-day detection histories (1= badger observed, 0 = no badgers observed) for each site each year (*sensu* Murray et al. 2021).

Characterizing Landscape Context and Composition

Each year we measured local-scale (0.25 ha) habitat characteristics at each site immediately prior to establishing the camera trap. This sampling scale was much smaller than the average home range of badgers (3-30km²; Doyle et al. 2019) and represents finer-scale resource use. We measured the proportion of cover of grasses (Grass), forbs (Forbs), woody vegetation (Shrub), and agriculture (Crops), and height of vegetation (cm, VegHeight), along with the proportion of bare ground (Bare), using a meter stick and Daubenmire square. All measurements were taken from 15 locations within a 50 x 50m grid square centered on the camera (Fig. 2B). For each site each year, we averaged the proportion of each variable across all measuring locations (n = 15) to include as a single site-level covariate.

We created 2-km radius circular buffers (ArcMap 10.6) centered on each camera trap site from which to extract relevant landscape-scale habitat metrics. We considered alternate buffer-radius sizes to approximate other scales (500m, 1km, and 5km). All derived landcover covariates, however, were correlated across all measured buffer sizes ($r > 0.80$) so we proceeded with a 2km buffer to be similar to other carnivore studies in the region (e.g., Wait et al. 2018). We created a custom raster layer by merging raster datasets from which to pull landcover information from each buffer (30-m x 30-m cell size). These datasets included Conservation Reserve Program (CRP; FSA 2018), CropScape (CropScape 2018), and PRAIRIEMAP (PRAIRIEMAP 2003). We used Fragstats (McGarigal 1995) to estimate the proportion of row crops (RowCrop), grasslands (Prairie), land enrolled in the Conservation Reserve Program (CRP), and total habitat edge (m, Edge) surrounding each site. We measured the Euclidian distance from each site (centered on the camera trap) to permanent water sources (DistWater; streams, ponds, rivers, and lakes). Number of wind turbines within the 2km buffer area of each

sites (Wind) were calculated using the USGS 2010 wind turbine map layer (United States Department of Energy 2014 Kansas Wind Energy Infrastructure Layer).

Multi-season Occupancy Analyses

We used multi-season occupancy models (unmarked package in R, Fiske and Chandler 2011) to assess both local and landscape characteristics important for site occupancy and turnover by badgers. Because diminishing scent could influence detection probabilities (Holinda et al. 2020) we considered the number of days since scent was applied at each site (Scent; 0–13 days). We also controlled for the day of year the camera was active (DOY; range = 112 –308) and for the potential of unmeasured variation between years (Year). We assessed support for seven detection models including single (DOY, Scent, Year) and additive effects (DOY + Scent, DOY + Year, Scent + Year, DOY + Scent + Year) of covariates along with a constant model of detection (Constant).

We created two candidate model sets to assess the influence of our measured covariates on both local- and landscape-scale site occupancy and turnover patterns by badgers. For our local-scale candidate set, we included models with both single (VegHeight, Grass, Shrub, Forbs, Crop, and Bare) and additive effects (Bare + Grass, Bare + Crop, Grass + Crop, Grass + VegHeight, Crop + VegHeight) along with a constant model (Constant). For our landscape-scale candidate set, we assessed 31 models including both single (CRP, RowCrop, Prairie, Edge, DistWater, Wind, and Precip) and additive effects (Prairie + Edge, Prairie + DistWater, Prairie + Wind, Prairie + RowCrop, Prairie + CRP, RowCrop + CRP, Edge + DistWater, Edge + Wind, Edge + RowCrop, Edge + CRP, DistWater + CRP, DistWater + RowCrop, DistWater + Wind, Wind + CRP, Wind + RowCrop, Wind + Edge + RowCrop, Wind + Prairie + RowCrop, Wind + DistWater + RowCrop, DistWater + RowCrop + Edge, DistWater + RowCrop + Prairie,

RowCrop + Prairie + Edge, DistWater + Prairie + RowCrop + Wind, DistWater + Prairie + RowCrop + Edge) along with a constant model (Constant). Each candidate set of models was identical for initial occupancy probability and colonization rate parameters. Because of model convergence issues, we did not model local extinction rates. All covariates were standardized prior to analyses.

For each rate parameter, we ranked models using Akaike's Information Criterion corrected for small sample sizes (AICc) and considered all models with $\Delta\text{AICc} \leq 2.0$ competitive (Burnham and Anderson 2002). We fit models for both local and landscape-scale effects by first identifying the most-supported detection model while holding all other parameters constant. We used this model to subsequently fit models of initial occupancy probability and colonization rates using the most-supported model from each parameter to fit subsequent parameters (Wait et al. 2018). We report model-averaged estimates from our most-supported models derived from a subset of models in the candidate set ($\Sigma w \leq 0.95$).

Results

From 2018-2020, we collected 5,217,641 photographs over 30,688 trap days. Naïve site occupancy by badgers was 0.35 ($n = 133$) in 2018, 0.42 ($n = 151$) in 2019, and 0.33 ($n = 119$) in 2020. Over the duration of the study, badgers colonized 139 sites (80 in 2019; 59 in 2020) and went locally extinct from 153 sites (62 in 2019; 91 in 2020). Our only competitive detection model indicated we were less likely to detect badgers earlier in the year (DOY: $\beta = 0.002$, SE= 0.001) and with greater days since sent application (Scent: $\beta = -0.078$, SE= 0.009; Table 2.1). Adjusted site occupancy probabilities in 2018, 2019, and 2020 were 0.458 (SE= 0.029), 0.401 (SE= 0.029), and 0.437 (SE= 0.035), respectively.

Local Scale

Our top and most-supported initial occupancy model included VegHeight and Crop (Table 2.1). Badgers were more likely to occupy sites with more crop cover (Crop: $\beta= 2.215$, SE= 0.646) and shorter vegetation heights (VegHeight: $\beta= -0.403$, SE= 0.157;Figure 2.3). Models including both covariates had the majority of model weights ($\Sigma w= 0.80$). There were two other competitive models (Table 2.1), however, inclusion of ‘Grass’ and ‘Bare’ did not improve model fit relative to our top model. Our most supported local-scale colonization model suggested badgers were more likely to colonize sites with greater proportions of bare ground (Bare: $\beta= 0.327$, SE= 0.160;Figure 2.4). There were three other competitive models (Table 2.1), however, the inclusion of ‘Crop’, and ‘Grass’ did not improve model fit relative to our top model.

Landscape Scale

Our top model for initial occupancy included RowCrop and DistWater (Table 2.2). Badgers were more likely to occupy sites with greater proportions of row-crop agriculture (RowCrop: $\beta= 0.689$, SE= 0.172;Figure 2.5) and sites closer to permanent water sources (DistWater: $\beta= -0.303$, SE= 0.142; Figure 2.5). Both covariates held the majority of model weights ($\Sigma w= 0.75$). There were three other competitive models (Table 2.2), however, the inclusion of ‘Prairie’, ‘Edge’, and ‘Wind’ did not improve model fit. Badgers are more likely to colonize sites with greater proportions of row-crop agriculture (RowCrop: $\beta= 0.573$, SE= 0.180; Figure 2.6). Models including RowCrop held the majority of model weights ($\Sigma w= 0.59$). There were two other competitive colonization models (Table 2.2), but the inclusion of ‘Prairie’ and ‘CRP’ did not improve the model fit relative to our top model.

Discussion

Contrary to our hypothesis, our results suggest site occupancy by badgers is positively associated with the amount of row-crop agriculture at both local and landscape scales. Additionally, badgers were more likely to colonize sites surrounded by agriculture at the landscape scale and were less likely to occur at sites farther from permanent water resources. At the local scale, badgers were more likely to occur at sites with shorter vegetation heights and colonize sites with more bare ground. Site occupancy and colonization by badgers was insensitive to the amount of prairie, CRP, and number of wind towers surrounding sites.

Row crop agriculture was an important predictor of site occupancy by badgers at both local and landscape scales and site colonization at the landscape scale. Our results concur with Duquette and Gehrt (2014), suggesting agricultural areas may provide suitable resources for badgers across their range. A mechanistic explanation for why badgers use agricultural areas remains uncertain. However, the fossorial life history of badgers, in combination with soils present in row crop agriculture fields, likely provide opportunity for badgers to burrow for shelter and forage for fossorial prey species. An alternate hypothesis would suggest badgers use agricultural fields because of increased availability of ephemeral water resources in irrigated fields. Unfortunately, we could not determine landscape-level distributions of ephemeral water resources provided by irrigated agricultural fields (compared to dryland agricultural fields) in our analysis. Small mammal species including deer mice (*Peromyscus maniculatus*), northern grasshopper mouse (*Onychomys leucogaster*), and house mouse (*Mus musculus*) in similar regions (northern Great Plains and north-central Kansas) occur at greater densities in agricultural fields (Heisler et al. 2013; Kaufman et al. 2000). It is unclear how densities of prey species are distributed across agricultural landscapes or field margins between adjacent field or road edges.

It is likely that the abundance of grain seeds in these areas provide increased resources for small mammals that may be unavailable in shortgrass prairies. It is plausible that artificially irrigated agricultural areas may supplant traditional bottom-up effects (Flowerdew and Sutherland 2017; Milstead et al. 2007; Báez et al. 2006) in historically arid prairie regions and these areas provide more prey resources for predators especially during drought. Future research should focus on understanding how row-crop agriculture modifies the distribution and abundance of prey species for carnivores occurring in the Great Plains.

At the local scale, badgers were more likely to use sites with shorter vegetation heights and colonize sites with greater amounts of bare ground (though this was a modest effect). These results may be explained by the way in which badgers navigate movements through vegetation. Movements in shorter, less dense vegetation may increase the badgers' perceptual range (e.g., Zollner and Lima 1997) and aid in foraging efficiency. Reduced vegetation densities at the local scale likely provide visual access to more areas and enhance line-of-site distances. An alternate explanation for these results would suggest that sites with lower vegetation heights and more bare ground could have resulted in greater detection probabilities. We do not believe that these results are reflective of enhanced detection probabilities at sites with shorter vegetation heights and more bare ground since we cut vegetation in front of cameras. However, we are unable to confirm this.

The presence of permanent water resources is important for mammals occurring in arid or semi-arid regions (Clemente et al. 1995; Leeuw et al. 2001; Webb et al. 2007). Our results concur with past studies underscoring the importance of water availability in our study region to mammal populations (Ochoa et al. 2021). There were ephemeral sources of water throughout our study area (e.g., roadside ditches, crop irrigation) that we could not identify and account for in

our analyses and these resources are likely as important to badgers as permanent water resources. The relative importance of permanent water resources is likely climate dependent where in times of drought, ephemeral sources of water disappear (Dalu and Dalu 2017). Dependable sources of water are key for mammals, especially during important life-history events when species experience greater energy expenditures (e.g., lactation and young-rearing; Robbins 2012). In our study, sampling occurred during badger kit-rearing season (Messick and Hornocker 1981; Symes et al. 2019), reflecting a need for dependable sources of water near dens. However, more research is needed to evaluate the importance of ephemeral water resources, specifically as it relates to crop irrigation and future climate-change scenarios, to mammal populations in arid or semiarid regions.

We hypothesized that greater amounts of prairie and CRP landcover would be associated positively with badger distributions. Our results, however, suggest that badger habitat use in our study area was relatively insensitive to these landcover types. Additionally, we did not detect an effect of wind farms (i.e., the number of wind turbines present at the landscape scale) on badger distributions. An undetected effect of prairie landcover on badger habitat use could possibly reflect lower energetic costs of obtaining prey within row crop agriculture versus prairie grasslands. Although badgers were not avoiding areas with prairie landcover, the locomotive costs of navigating through vegetation in prairie grasslands may outweigh the foraging gains in crop fields (Karasov 1992). Thus, badgers may be concentrating their foraging and kit-rearing efforts along agricultural field edges. This hypothesis assumes that prey densities, or access to prey, is greater along agricultural field edges than in prairies and warrants future research. The effects of CRP on mammal distributions, particularly carnivores, are largely undocumented though Kamler et al. (2005) reported transient coyotes select for CRP whereas more established

residents selected native prairie. The insignificance of number of wind turbines on badger occupancy and colonization could be explained by the small average number of wind turbines ($x= 0.22$; range= 0-22) at our measured landscape-scale.

Our study underscores the ability of badgers to use alternate habitats in human-dominated landscapes. These results enhance our understanding of the broadly defined effects of human development on carnivores occurring in semiarid regions, particularly badgers. Understanding the effects of specific landuse changes on carnivore distributions can inform effective species-specific management plans, enhance conservation efforts, and predict future range-shift scenarios. Future research should use finer-scale population demographic information (e.g., fecundity, known-fate survival) to enhance our understanding of how human-dominated landscapes affect population growth and persistence of badgers in the Great Plains.

Table 2.1 Local-scale model-selection results describing detection, initial occupancy, and colonization rates of sites (range = 0.22- 0.16) by American badgers (*Taxidea taxus*) in western Kansas, USA during summers of 2018-2020. Models are ranked by descending ΔAICc values. w = Model weight, -2ll = -2(log likelihood), and K = number of parameters. Models annotated with an asterisk are used to fit models in subsequent initial occupancy and colonization parameters.

Model	ΔAICc	w	-2ll	K
<i>Detection</i>				
Scent + DOY*	0.00	1.00	-3730.03	6
Scent	30.25	0.00	-3746.19	5
DOY	76.12	0.00	-3769.12	5
Null	394.52	0.00	-3929.35	4
<i>Initial Occupancy</i>				
VegHeight + Crop*	0.00	0.49	-3706.80	8
VegHeight + Crop + Grass	1.37	0.25	-3706.44	9
VegHeight + Crop + Bare	1.87	0.20	-3706.67	9
Null	42.29	0.00	-3730.03	6
<i>Colonization</i>				
Bare	0.00	0.17	-3698.23	10
Bare + Crop	0.04	0.16	-3697.31	10
Bare + Grass	1.80	0.08	-3698.13	10
Crop + Grass	1.98	0.07	-3698.22	10
Null	2.77	0.05	-3706.80	8

Table 2.2 Landscape-scale model-selection results describing detection, initial occupancy, and colonization rates of sites (range = 0.22- 0.16) by American badgers (*Taxidea taxus*) in western Kansas, USA during summers of 2018-2020. Models are ranked by descending ΔAICc values. w = Model weight, -2ll = -2(log likelihood), and K = number of parameters. Models annotated with an asterisk are used to fit models in subsequent initial occupancy and colonization parameters.

Model	ΔAICc	w	-2ll	K
<i>Detection</i>				
Scent + DOY*	0.00	1.00	-3730.03	6
Scent	30.20	0.00	-3746.19	5
DOY	76.12	0.00	-3769.12	5
Null	394.52	0.00	-3929.35	4
<i>Initial Occupancy</i>				
RowCrop + DistWater*	0.00	0.31	-3706.05	8
RowCrop + DistWater + Prairie	1.51	0.14	-3705.76	9
RowCrop + DistWater + Edge	1.79	0.13	-3705.90	9
RowCrop + DistWater + Wind	1.93	0.12	-3705.97	9
Null	43.80	0.00	-3730.04	6
<i>Colonization</i>				
RowCrop	0.00	0.23	-3697.62	9
RowCrop + Prairie	1.77	0.09	-3697.45	10
RowCrop + CRP	1.97	0.08	-3697.55	10
Null	14.80	0.00	-3706.05	8

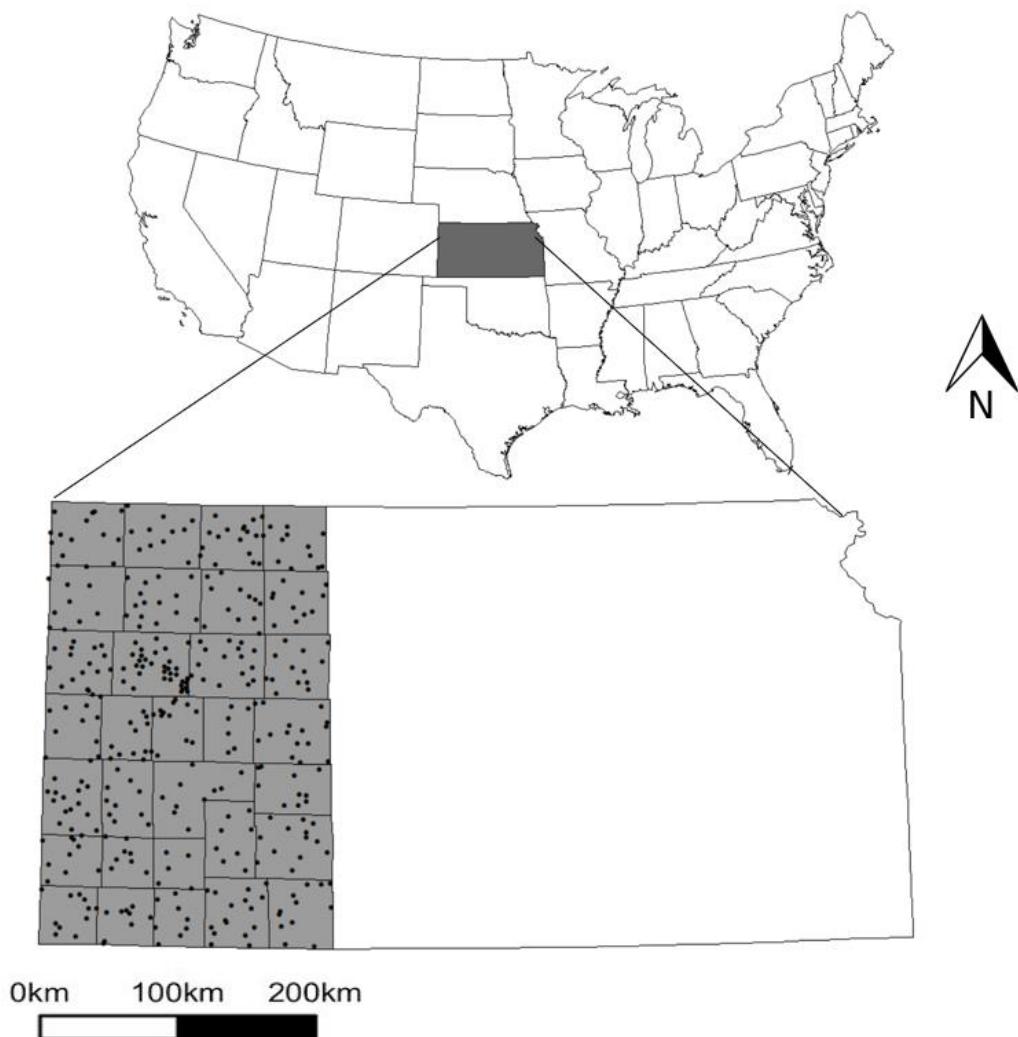


Figure 2.1 Study area (gray) and camera-trap sites (dark dots, n=381) used to assess American badger (*Taxidea taxus*) distributions in western Kansas, USA during summers of 2018-2020.

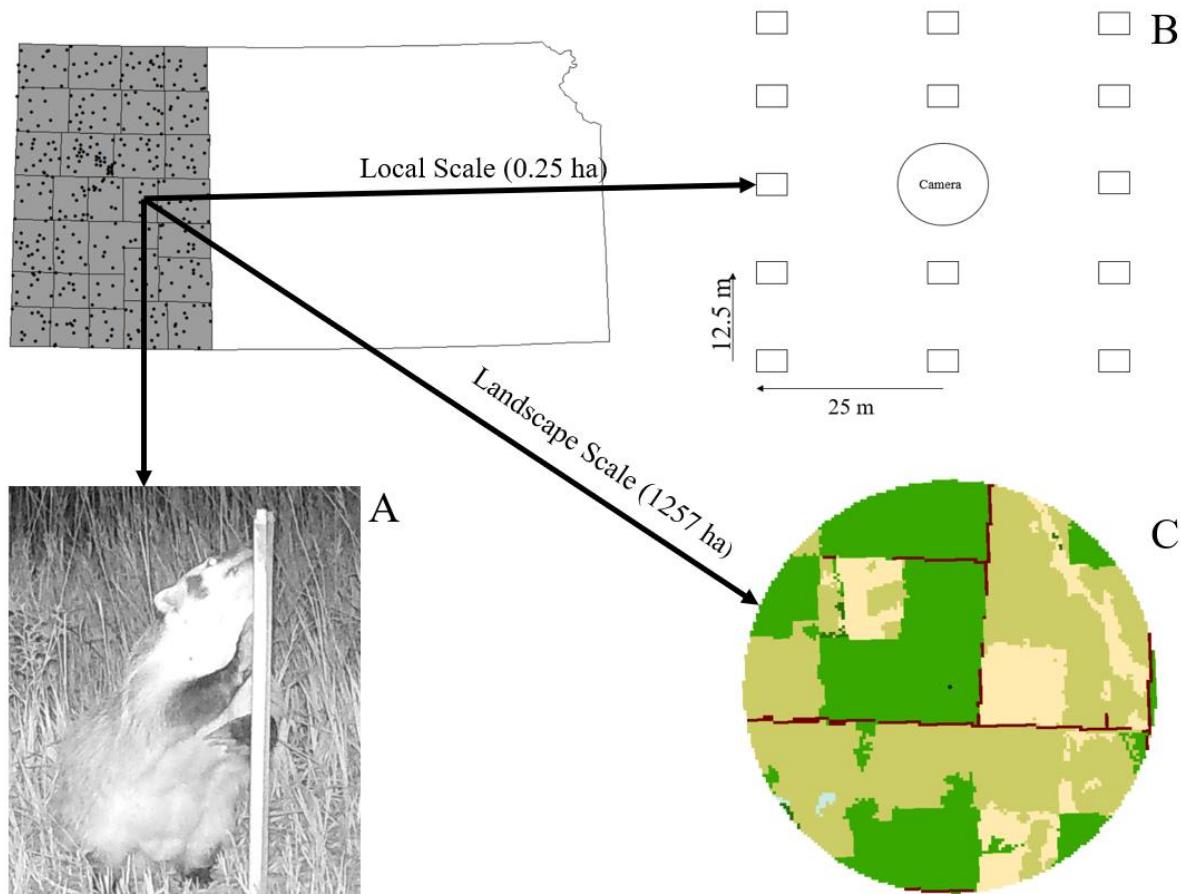


Figure 2.2 Description of local and landscape scales used to predict occurrence of American badger (badger, *Taxidea taxus*; A) at camera-trap sites in western Kansas, USA during summers of 2018-2020. We measured and characterized local scale vegetation structure and composition at 15 stations (boxes) surrounding the camera trap (B). We characterized landscape-scale landcover composition and configuration within a 2-km radius buffer surrounding each camera-trap site (C).

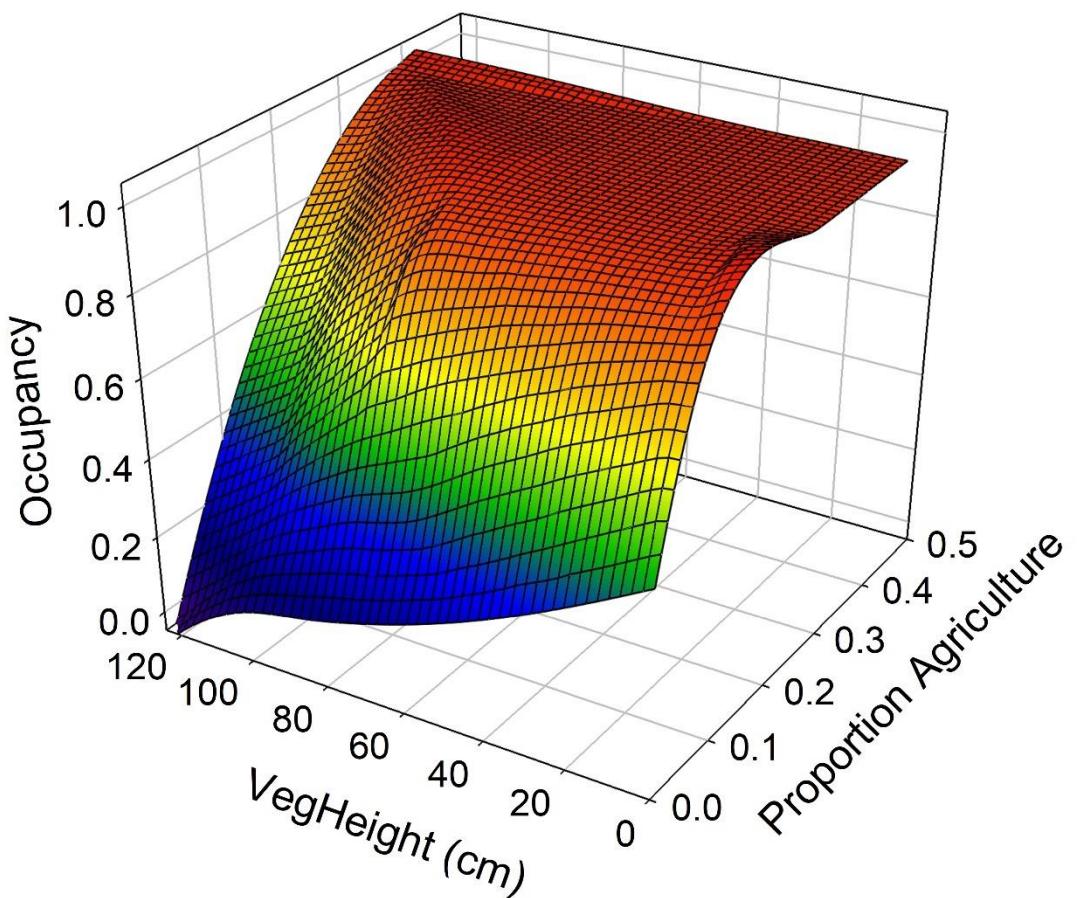


Figure 2.3 Predicted initial site occupancy probabilities from our most-supported local-scale (0.25 ha) occupancy model. American badgers (*Taxidea taxus*) were more likely to occur at sites with shorter vegetation heights and more row-crop agriculture.

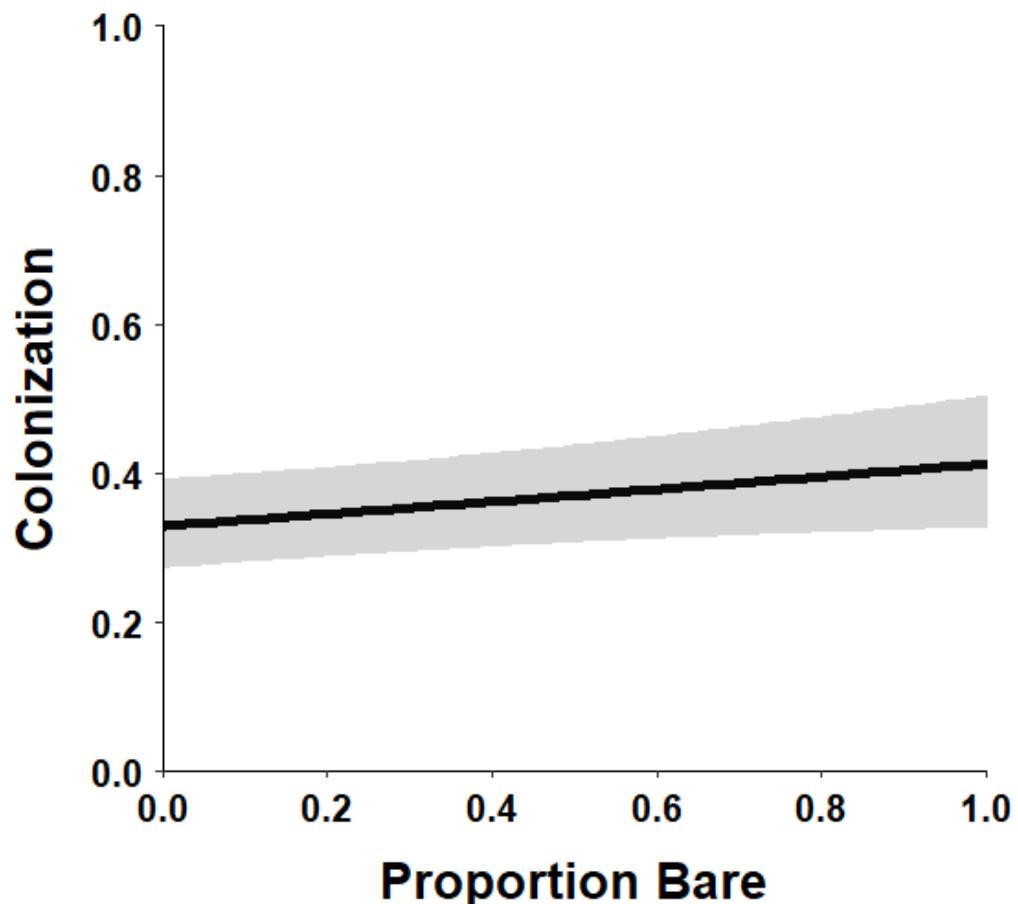


Figure 2.4 Predicted site colonization rates from our most-supported local-scale (0.25 ha) colonization model. American badgers (*Taxidea taxus*) were more likely to colonize sites with more bare ground.

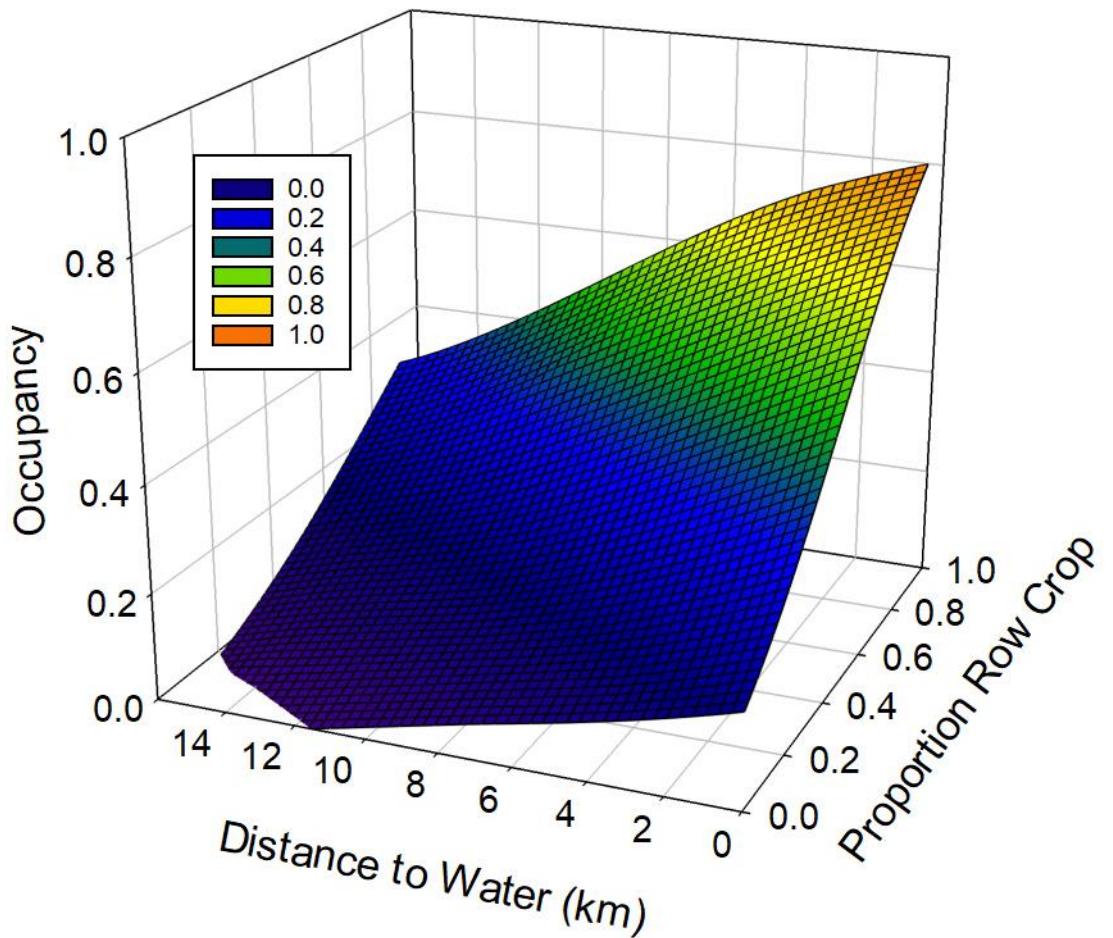


Figure 2.5 Predicted initial site occupancy probabilities from our most-supported landscape-scale initial occupancy model. American badgers (*Taxidea taxus*) were more likely to occur at sites closer to permanent water resources and sites surrounded by greater proportions of row-crop agriculture.

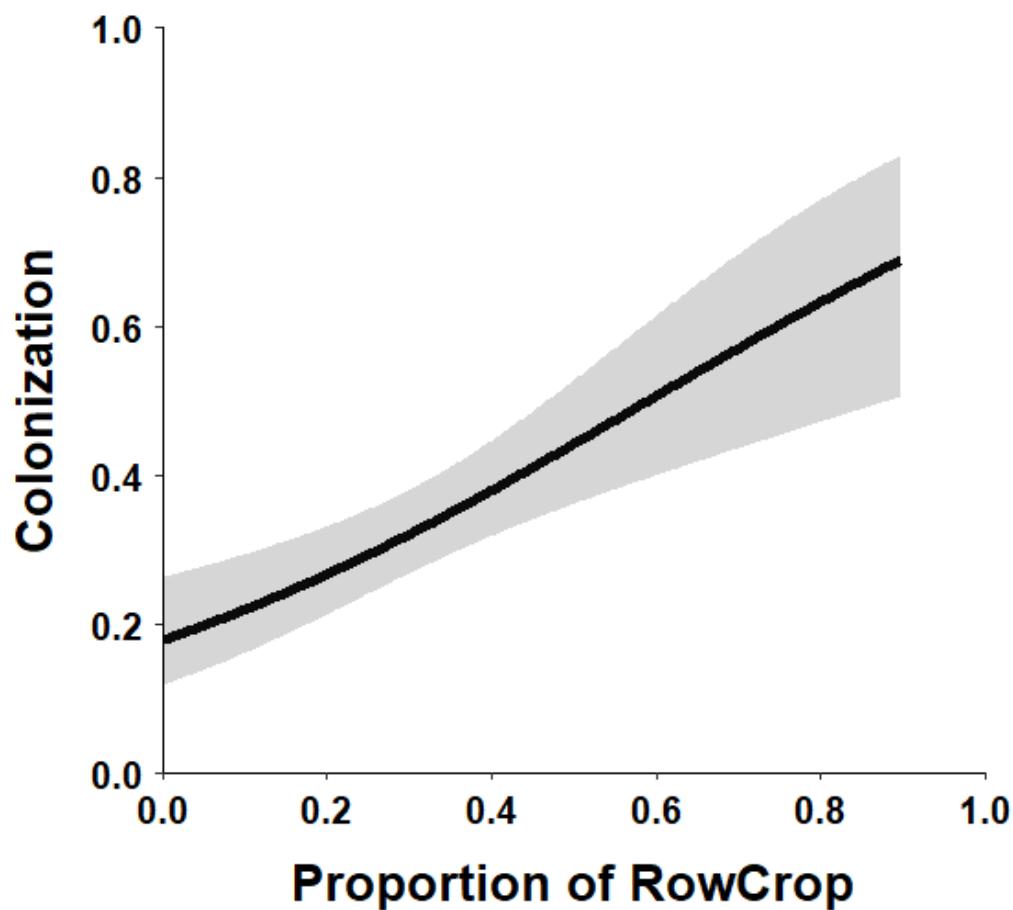


Figure 2.6 Predicted initial site occupancy probabilities from our most-supported landscape-scale initial occupancy model. American badgers (*Taxidea taxus*) were more likely to occur at sites closer to permanent water resources and sites surrounded by greater proportions of row-crop agriculture.

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Chapter 3 - Conclusion

American badgers are valuable as furbearers, top predators throughout their range, and possible ecosystem engineers because of their frequent burrowing (Symes et al. 2019; Obbard et al. 1988; Messick and Hornocker 1981). Being able to clearly define distributions of key predators in specific ecosystems can provide insight to the overall health of the ecosystem, including information regarding prey abundances and nutrient cycling (Schmitz et al. 2010). Literature defining distributions of American badgers within shortgrass prairie ecosystems is currently lacking. Within the Great Plains, badgers were once considered ‘prairie-obligate’. This definition, however, may prove to be outdated.

As human populations continue to increase and expand, so too will the development of natural lands to compensate for living space, food production, and energy production (Ehrlich and Holdron 1971). With this development comes the loss/ fragmentation of critical habitat and loss of biodiversity (Fahrig 1997). Prairie lands, including shortgrass prairies, are in danger of being transformed into working lands (Samson and Knopff 1994). To species defined as prairie-obligate, such as the American badger, this could lead to population decreases and shifts in historically documented geographic ranges (Wiegand et al. 2005). We can get a deeper understanding of how prairie habitat loss will continue to effect badgers if we can pinpoint their specific distributions and resource-use relative to landcover types within prairie ecosystems.

I used three years of presence-absence data from 381 camera trap sites to determine the occupancy dynamics of American badgers in western Kansas. Using landcover data on both the local and landscape scale, I was able to quantify effects of different landcover classes on badger occupancy and colonization. I had an average naïve occupancy of badgers of 0.37 at the 381 sites over the three-year study period and experienced 139 colonization events in the last two years.

My results suggest that American badgers have a higher probability of occupying areas with larger proportions of row crop agriculture that occur within closer proximity to permanent water sources on the landscape scale. On the local, or 2km scale, badgers are more likely to occupy areas with lower vegetation heights and higher proportions of crop species (i.e. wheat, corn, and milo). My results also suggest that colonization rates of badgers on the landscape scale increase with higher proportions of row crop agriculture. On the local scale, colonization rates increase with higher proportions of bare ground.

My study reveals that American badgers within shortgrass prairie ecosystems may not actually be prairie-obligate, and populations are not as sensitive to specific forms of human activity as previously considered. Our result that badgers are occurring in agricultural settings are similar to those defined in Doyle et al. (2019) and Duquette and Gehrt (2014). As of 2007, 37% North America is classified as agricultural land, and is forecasted to increase (USDA 2007). Looking at our results and supporting literature, this may not pose a substantial threat to American badger populations. This could also allude to occurrence of key badger prey species in suitable abundances within row crop agriculture habitat (Heisler et al. 2013; Kaufman et al. 2000). Taking these results into account when creating management plans in shortgrass prairie ecosystems can help to consider relative importance of some human modified landcover to specific native species populations. Future research focused on specific American badger population dynamics (e.g. fecundity, survival, mortality) in varying seasons within agricultural habitat may provide useful information into exactly how some human activity impacts badger (and similar carnivores) populations.

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