The effects of human development, environmental factors, and a major highway on mammalian community composition in the Wasatch Mountains of northern Utah, USA

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Abstract
Human development and roads threaten wildlife through distinct mechanisms and understanding the influence of these elements can better inform mitigation and conservation strategies. We used camera traps to quantify the effects of major roads, environmental factors, and human development on the mammalian community composition between sites north and south of a major interstate highway in northern Utah, USA. We found no significant differences in species richness nor community similarity across the north–south divide of the highway. Through Bayesian hierarchical modeling, we compared the effects of the distance to the highway, housing and human population density, normalized difference vegetation index (NDVI), and the human footprint index to changes in mammalian community composition and species-specific habitat usage. Community occupancy response, similarity, and species richness were negatively affected by increased housing and human population densities and positively affected by increased NDVI and decreased human footprint, whereas their response to the highway was more inconclusive. We conclude that mammalian community composition in our study area is influenced by both environmental conditions and human development while the effect of the highway was more nuanced, possibly due to the presence of a newly constructed wildlife overpass. Taken together, the lack of differences in species richness or community composition across the highway suggests that it may not currently exacerbate the effects of other anthropogenic sources of habitat fragmentation and highlights the need for additional research into human–wildlife conflict mitigation strategies.

Kelsey A. Barnick and Austin M. Green contributed equally to this work and would like to be considered as co-first authors.

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1 INTRODUCTION

Both the expansion of roads and increased human development threaten wildlife occupancy and mobility (Benitez-López et al., 2010; Crooks et al., 2017; DeMars & Boutin, 2018; Gleeson & Gleeson, 2012; Kiesecker et al., 2010; Krauss et al., 2010). However, roads and development differ in the processes by which they affect both individual species and community composition, as well as in the conservation strategies needed to mitigate their impacts (Benitez-López et al., 2010). Therefore, understanding the role of these elements in influencing wildlife distribution and community composition can better inform conservation strategy and practice.

While species-level effects of roads may vary, evidence suggests that their impacts on ecosystems, in general, are detrimental (Crooks et al., 2017; Krauss et al., 2010). The damaging effects of roads on wildlife populations and community diversity occur over short- and long-term spatiotemporal scales (Clements et al., 2014; Crooks et al., 2017; Krauss et al., 2010). Avoidance of roads due to the lack of vegetation cover, noises, and increased exposure to vehicles prevent species from moving between habitats, causing minimal dispersal and the creation of habitat “islands” (Benitez-López et al., 2010; Fahrig & Rytwinski, 2009). Limited connectivity between habitats combined with reductions in habitat area can contribute to population declines and increased local extinction risk for many species (Crooks et al., 2017; Krauss et al., 2010). However, there is evidence that suggests that some small mammals and bird occupancy may experience no impact or even be positively associated with roads (Rytwinski & Fahrig, 2013). The predator release hypothesis suggests that prey find success in habitats near roads by evading road-avoidant apex predator populations (Cove et al., 2012; Fahrig & Rytwinski, 2009; Goad et al., 2014; Rodriguez et al., 2021). Furthermore, simulation studies and modeling results have found that species with high reproductive rates and small home ranges are most likely to have positive associations with roads because populations are likely to be able to persist within the areas confined by roads (Fahrig & Rytwinski, 2009). Therefore, there is rationale to suggest that the presence of roads may alter community composition within any given ecosystem.

Similar to roads, human development such as housing and other built structures negatively impacts wildlife through multiple mechanisms. In the case of development, these mechanisms include habitat loss, degradation, and fragmentation (Scanes & Toukhsati, 2017); disruption of ecosystem function and processes (Vukomanovic et al., 2013); and change in animal behavior (Gaynor et al., 2018; Hansen et al., 2005). Not only does development directly affect and often degrade wildlife habitat, but it also increases wildlife exposure to humans (Hansen et al., 2005). Previous studies have shown large carnivores, mesocarnivores, and ungulates to be negatively associated with increasing housing densities (Fahrig & Rytwinski, 2009; Goad et al., 2014; Rodriguez et al., 2021; Wang et al., 2015). However, similar to the phenomena seen with roads, these same studies have shown mid-sized and small mammals to demonstrate positive associations with developing areas, with resulting changes to community composition (Cove et al., 2012; Fahrig & Rytwinski, 2009; Goad et al., 2014; Rodriguez et al., 2021).

Both roads and development increase wildlife interactions with and exposure to people (Karanth, 2016; Naidoo & Burton, 2020). Additionally, development and roads both often result in fragmented landscapes, which degrade the quality and quantity of habitat in a way that greatly threatens species (Benitez-Lopez et al., 2010; Clements et al., 2014). Although there are many similarities, roads and human development also have distinctive mechanisms in which they affect wildlife movement and occupancy. For example, while roads often serve as a barrier between habitat, development may only affect a specific area. This results in a need for differing mitigation measures that target specific characteristics of either infrastructure. Therefore, in order to develop effective conservation strategies, understanding the role of these factors in influencing wildlife is vital.

This study focused on understanding the role of a major highway, environmental factors, and human development in shaping mammalian community composition and species-specific habitat use between sites north and south of a major highway—interstate highway 80 (I-80)—just outside of Salt Lake City in the surrounding Wasatch Mountain Range (WMR) of northern Utah, USA. The WMR, located directly east of Salt Lake City, hosts an intermix of diverse habitat and housing and is divided into a northern and southern section by a major highway (I-80) that runs from east to west along the WMR. Understanding the factors that cause potential dissimilarities in community composition across the WMR’s I-80 divide is imperative to determining how to best mitigate impact and protect the wildlife in the WMR,
especially when considering that human influence in the area is growing at a rapid rate. We compared areas north and south of I-80 to uncover and compare the effects of major roads and human development on community composition and habitat usage. We assessed whether the combination of housing and human population density differences, the presence of I-80, and increases in human footprint index paired with decreases in normalized vegetation difference index (NDVI) have led to changes in the mammalian community composition across the I-80 divide of the WMR. We predicted that at both the community and species-specific level, we would find a negative response to increasing human development and human population densities, and a positive response to increasing NDVI and decreasing human footprint. We also anticipated a positive response to increasing distance from I-80, with the hypothesis that species would avoid the highway and habitat edge created by it.

2 | METHODS

2.1 | Study area

Salt Lake City, Utah (SLC), is located in the valley between the Wasatch and Oquirrh mountain ranges, where the juxtaposition of nature and development is evident. On the eastern side of the valley, this juxtaposition creates a buffer known as the wildland–urban interface (Martinuzzi et al., 2015), which includes both interface and intermix land classifications. The intermix classification defined by Stewart et al. (2007) includes an area with greater than 50% wildland vegetation (e.g., vegetated protected areas, undeveloped land, slopes, mountaintops, etc.) and a minimum of 6.18 housing units per square kilometer. The interface classification includes the same criteria as intermix, though it also requires the area to be located “within [~2.4 km] of [a] large area with >75% wildland vegetation” (Stewart et al., 2007). As of 2015, approximately 1.3 million Utahns (~46% of the state’s human population) live along the WMR wildland–urban interface, which accounts for approximately 2% of the state’s total land area (Martinuzzi et al., 2015). The WMR east of SLC receives more than 9 million visitors per year (U.S. Forest Service, Personal Communication). These visitors engage in various types of recreation including hiking, camping, fishing, skiing, snowboarding, and wildlife viewing. Our study area comprised a total of 453 km² within the WMR, where a combination of wildland urban interface and intermix areas is located throughout (Figure 1).

I-80 is a major transportation route that runs across the United States from San Francisco, California to New Jersey. The highway was constructed in 1956, with the segments found within the study area being completed in 1986. In northern Utah, I-80 bisects the WMR and runs through Salt Lake City. The study area includes sites north and south of I-80 to assess whether or not measures of community similarity and composition, as well as species habitat usage, are influenced by proximity to the highway, human development, and vegetation differences (Figure 1). As reported by the Utah Department of Transportation (UDOT), the Annual Average Daily Traffic (AADT) rate is meant to represent traffic on a typical day of the year, calculated by dividing the total volume of vehicle traffic of a highway or road by 365 days. The portion of Interstate 80 included in the study area had an estimated AADT of 59,000 vehicles during the study period (Utah Department of Transportation, 2020). As of 2018, 8 ft tall wildlife fencing stretches along approximately 15 km of the portion of Interstate-80 involved in the study area (approximately 35 km in total), with more being constructed in the years since this study was conducted (Egan, 2018; Utah Department of Transportation, 2013). The wire fencing has heavy posts every 10 ft as well as galvanized steel at the top and bottom to prevent wildlife from going under or over it. A wildlife overpass established in December 2018 is located centrally to the study area (Figure 1). The overpass measures 97.5 m × 15.2 m with 5.6 km of wildlife fencing on each side designed to funnel wildlife to and traffic across all six lanes of the highway.

Red Butte Canyon Research Natural Area (RBC) is a protected area that borders SLC on the north side of I-80. Since 1969, RBC has been a designated protected area where only researchers are permitted access (Ehleringer et al., 1992). Before that, from 1862 to 1968, RBC was a protected watershed for the US Army. The protected area includes approximately 25 km² and an elevation range of 1500–2500 m. Recent research has shown relatively high species richness and mammalian diversity in RBC, suggesting its role as a protective habitat for the WMR (Green et al. unpublished data). We consider this site to be the best available example of an intact mammalian community in the WMR (Ehleringer et al., 1992).

2.2 | Covariate selection

To estimate the effects of environmental conditions, human development, and the presence of Interstate Highway 80 on both species-specific and community wide habitat use, we elected to construct a Bayesian multispecies, single-season occupancy model to describe the patterns of species detection/non-detection data. For our analysis, each site was defined by the actual GPS location
of the camera surrounded by a 100-m buffer. When appropriate, covariate data within the buffer were averaged to record a single value for each site. In an effort to both limit the number of covariates used in our model and address the collinearity in multiple covariate pairs (Pearson’s $r > .5$), we conducted principal component analysis (PCA) on our collinear predictors and used the first two principal components (PC1 and PC2), which collectively accounted for 70% of the total variation across predictors (Supplementary Table 1), as covariates in our occupancy model. Each predictor included in the PCA was scaled to a mean $= 0$ and standard deviation $= 1$ before analysis. The predictors included in the PCA were human population density, housing density, NDVI, Human Footprint Index (a measure from 0 to 50 that includes multiple aspects of human influence, including development, agriculture, and transportation infrastructure [Venter et al., 2016]), and human detection rate from camera trap photo captures. Human detection rate was included in case there was significant variation in site composition not accounted for in the human population density, housing density, and human footprint index. However, loading factors for human detection rate were low (mean $= 0.28$, Supplementary Table 1) and were not included in the interpretation of either principal component moving forward. For all further analysis, we use the first two principal components, which accounted for 70% of the variation across sites. Principal components described axes of increasing population and housing density (PC1) and increasing NDVI and decreasing human footprint (PC2; Supplementary Table 1). Human population density and housing density values were obtained from the spatial dataset created by Radeloff et al. (2017), which included analysis of demographic data from the US Census using the Bureau’s spatial dataset, TIGER/Line geodatabases. Human population and housing densities were calculated using these data US by dividing housing units and persons by square kilometer of area, respectively, resulting in a resolution of 30 m (Radeloff et al., 2017). NDVI was calculated in ArcGIS using Landsat 8 OLI/TIRS data at a resolution of 30 m. Human footprint index values were obtained from the dataset FIGURE 1 Study area. Map of the study area located in northern Utah. Wildland urban interface and intermix areas as delineated by Radeloff et al. (2017) are indicated. Intermix areas are defined as locations with >50% vegetation and at least 6.177635 housing units per square kilometer. Interface areas have the same criteria as intermix areas, but they also must be located within 1.5 mile of large area with >75% wildland vegetation.
produced by Venter et al. (2016) at a resolution of 1-km. We included variables of human development at different resolutions (e.g., human population density and human footprint index) in our PCA to account for the discrepancies between large-scale and fine-scale human influence within our study area (Figure 1). As we wanted to assess the effects of habitat modification and the presence of the interstate highway separately, we also included distance to the interstate as a covariate in our occupancy model.

### 2.3 Data collection

We set up 26 Bushnell Aggressor HD camera traps (Bushnell Corporation, Overland Park, Kansas, USA) at various sites between Salt Lake City and Park City (Figure 1). Along I-80, we placed 18 cameras at nine sites, one on the interstate’s northern side, and one on the southern side. Each site was 5 km apart, and each camera was <1500 m from the highway on both the north and south sides of the study area. Additionally, we placed eight cameras >1500 m from I-80 to assess whether any of the measured effects were distant dependent. These cameras were each approximately 8 km away from the interstate and located on both sides of I-80 (Figure 1). Camera distance from the interstate was bimodally distributed, with 11 cameras within 250 m of the interstate, 14 within 500 m, and 16 within 1 km. The other 10 cameras were greater than 1 km away, with 8 greater than 5 km from the interstate (Figure 1). Sites had a mean elevation of 1926.76 m (min = 1529.81 m, max = 2716.43 m) and a mean slope of 17.70° (min = 1.14°, max = 35.96°). Human footprint index, housing density, population density, vegetation, and development levels varied slightly between sites north and south of the highway (values listed below represent the difference between the sites located in the south compared to the sites located in the north). Although not significantly, sites south of the highway, on average, had a larger human footprint index (mean difference = 2.12; 95% confidence interval of difference from north [CI] = −6.07–10.31), greater housing density (units/km²) (mean difference = 55.95; CI = −16.91–128.82), greater human population density (persons/km²) (mean difference = 150.88; CI = −46.85–348.61), lower PC2 values (mean difference = 0.03; CI = 0.19–0.14), and larger PC1 values (mean difference = 0.10; CI = −0.05–0.26) than sites north of I-80 (the latter two factors do not contain units, as they are calculated as a result of the PCA described above). Although these trends highlight differences among northern and southern sites, they also show that, in general, the areas on either side of the highway differ only slightly in our measured covariates.

Data collection occurred between August 31, 2019 and February 22, 2020. Pilot studies had shown complete random placement within the WMR to result in trail cameras placed in areas of extreme slope and difficult terrain, resulting in extremely low capture rates for all target species (Green, unpublished data), so cameras were placed facing areas of high wildlife activity such as game trails and canyon drainages and were set up as close as possible to predetermined GPS coordinates (<500 m), with a new GPS location taken at the actual deployment site. Although this type of camera placement has been found to bias detections based on species’ preferences for specific habitat features (Cusack et al. 2015), these features also represent natural funnels where most of the individual animals are likely to traverse throughout the otherwise steep and difficult-to-navigate terrain of the WMR. Original GPS locations were assigned using ArcGIS, starting from the point of I-80 at the mouth of Parley’s Canyon within the WMR and progressing eastward along the interstate in 5-km intervals (Figure 1). Vegetation was cleared to prevent obstruction of the lens, and all sites were active for a period of 4 weeks. The cameras were not baited, operated for 24 h a day, and were set to take a series of three photographs when triggered with an interval of 15 s between subsequent triggers.

### 2.4 Data analysis

#### 2.4.1 Multispecies, single-season occupancy model

We adopted a hierarchical, Bayesian multispecies, single-season occupancy approach (Dorazio & Royle, 2005; Rich et al., 2016) to estimate the probability that each species used the habitat surveyed by a camera station while correcting for imperfect detection (MacKenzie et al., 2002). Using recommendations from MacKenzie et al. (2017), we defined the parameter of interest (occupancy) as the proportion of camera sites used by a species (hereafter referred to as site use) instead of estimating the proportion of total study area occupied by each species (Kery & Royle, 2015; Mackenzie et al., 2017). This approach defined site use as a partially observed latent, binary variable equal to 1 if a species uses the area surveyed by the camera during the study period and 0 otherwise. Occupancy analysis requires repeat surveys in either space or time to garner information on each species’ site-specific detection probability (MacKenzie et al., 2002). For this analysis, each site’s sampling period was broken into multiple repeats of 7-day sampling periods. We chose this period as an offset between (1) increasing the detection probability of the large-bodied, wide-ranging species in
our study and (2) providing enough spatial replicates for accurate estimation of detection probability across species (Blount et al., 2021; Mackenzie et al., 2017; Whittington et al., 2018).

We investigated the effects of site-level covariates on both species-specific and community site use while allowing detection probability to vary across species (Rich et al., 2016). This was done by modeling the linear effect of each covariate on the logit-transformed site use parameter (Dorazio & Royle, 2005; Zipkin et al., 2009; Zipkin et al., 2010). For this analysis, we used the three continuous covariates outlined above: PC1, PC2, and distance from Interstate Highway 80. As described earlier, the first two principal components described axes of increasing population and housing density (PC1) and increasing NDVI and decreasing Human Footprint (PC2; Supplementary Table 1). As specified above, all covariates were scaled to a mean = 0 and standard deviation = 1 before running the occupancy model. We linked species-specific parameters to a larger community hyperparameter using the hierarchical approach of Rich et al. (2016). Under this framework, a community-wide hyperparameter specifies the mean response of the entire community to a given variable. Species-specific responses were modeled as random effects derived from the aforementioned hyperparameter (Kery & Royle, 2015; Rich et al., 2016; Zipkin et al., 2010). We estimated posterior distributions for coefficients using Markov Chain Monte Carlo (MCMC) implemented in JAGS through program R using the “rjags” and “jagsUI” packages (Kellner, 2019; Su & Yajima, 2020). We used three chains of 20,000 iterations each. The initial 10,000 iterations per chain were discarded as burn-in, and the resultant distributions were thinned by five, leaving a total of 6000 samples to craft the posterior distribution of each parameter. We used uninformative, uniform priors for all parameters and assessed convergence using the Gelman–Rubin statistic, where values <1.1 indicate convergence (Gelman et al., 2004) and by visual inspection of parameter traceplots. “Significant” differences were assessed using posterior probability, where “strong significant” effects were expressed as parameters with >95% posterior probability that did not overlap with 0 (i.e., our model estimates that the probability the parameter effect is more extreme than 0 is >95%), and “moderate significant” effects were expressed as parameters with >85% posterior probability that did not overlap with 0. We included the 85% threshold to capture species and community effects that exhibited a strong trend toward positive or negative association but were limited by sample size and high variation in the community hyperparameters, which can lead to “shrinkage” of species-specific effects toward the community mean (Kery & Royle, 2015; Mata et al., 2017; Suraci et al., 2021).

2.4.2 | Species richness and community similarity indices

Each iteration of the MCMC chain presents a new estimate for all parameters, including site-specific estimates of site use for each species. Summing the total number of species using a particular site during each iteration of the model gives a derived estimate of species richness. This generates a posterior distribution of derived site-specific species richness estimates (Rich et al., 2016; Zipkin et al., 2010). Furthermore, these estimates can also be used to estimate community similarity among sites (Kery & Royle, 2015). Specifically, we used the estimated occupancy matrix from each species during each iteration of the three MCMC chains to calculate site-level Jaccard similarity indices (hereafter referred to as community similarity), which specify the similarity of two sites in terms of the species they share in common (Dorazio et al., 2011). The index ranges from 0, where no species are shared between sites, to 1, where all species co-occur between sites. As the Jaccard index requires a reference site, and although we do not consider RBC to be a source population for our study area, we randomly selected a site within RBC as reference because it represents the most pristine, untouched land in the Wasatch Front (Ehleringer et al., 1992), where we consider the community within to represent the best available reflection of what an intact, undisturbed mammalian community would resemble.

Using these derived measures of community similarity and species richness, we adopted a Bayesian generalized linear modeling framework for assessing the effects of our occupancy covariates on these parameters. However, since the species richness and community similarity indices were themselves estimates from a previous model, we had to make sure to account for their estimated error in our model. To do this, we propagated the error from our occupancy model through our generalized linear model by modeling the response (i.e., species richness and community similarity) as a random variable with associated “observation” error,

$$N_i \sim \text{Normal} (\mu_i, 1/\sigma_i^2)$$

where $N_i$ denotes each site’s predicted response, $\mu_i$ is the site-specific estimated mean from the occupancy model, and $\sigma_i$ is the site-specific derived posterior standard
deviation from the occupancy model (Kery & Royle, 2015; Suraci et al., 2021). We then modeled \( N_0 \) as a function of PC1, PC2, and distance from I-80 using a Gaussian linear model. Our species richness model was run across all sites, while our community composition model excluded the reference site from analysis. We again used MCMC through JAGS in program R to estimate posterior distributions for all parameters. For both models, we ran three chains of 10,000 iterations each. The initial 5000 iterations per chain were discarded as burn-in, and the resultant distributions were thinned by two, leaving a total of 7500 samples to craft the posterior distribution of each parameter. We used uninformative, normal priors for all coefficients and a vague, uniform prior for standard deviation. We assessed convergence for both models, as well as “significance” of predictors, as described above.

Finally, to compare differences in community similarity and species richness across the north–south dividing Interstate, we used the same error-propagation methods described above and ran a Bayesian one-way ANOVA, which included the categorical factor of whether or not a site was on the north or south of I-80. Model parameters were the same as the above linear model, with both “significance” of predictors and model convergence assessed in the same manner as above. We elected to run separate models using our categorical and linear predictors to both improve parameter convergence and avoid over-parameterizing a model with low sample size.

### 3 | RESULTS

#### 3.1 | Survey effort

Across all sites, during 864 survey days, we recorded 868 detections of 15 species. Mule deer (*Odocoileus hemionus*) were observed most frequently (577), followed by coyote (*Canis latrans*; 36) and elk (*Cervus canadensis*; 29). Species were excluded from analysis if they were unreliable detected by cameras (e.g., smaller body size) or if they were only detected once by all camera traps. These species included American red squirrel (*Tamiasciurus hudsonicus*), rock squirrel (*Otospermophilus variegatus*), Uinta ground squirrel (*Urocitellus armatus*), fox squirrel (*Sciurus niger*), and mountain cottontail (*Sylvilagus nuttallii*). This resulted in a final dataset of 689 detections of 10 target species (Table 1). Specifically, sites north of I-80 resulted in 400 detections of wildlife, compared to 289 detections in sites south of I-80. Species will subsequently be referred to by their common name listed in Table 1. The data used in this study were directly downloaded from and are publicly available in the eMammal Database: https://emammal.si.edu/.

#### 3.2 | Community occupancy model

Community site use was relatively high (posterior mean = 0.87, Bayesian 95% credible interval [CI] = 0.27–0.98). Species-specific estimates ranged from 0.40 for cougar (CI = 0.19–0.77) to 0.95 for mule deer (CI = 0.85–1.00). Community detection probability was relatively low (posterior mean = 0.09, CI = 0.03–0.21). Species-specific estimates ranged from 0.04 for moose (CI = 0.01–0.10) to 0.59 for mule deer (CI = 0.50–0.69; Supplementary Table 2).

Community response to PC1, describing axes of increasing population and housing densities, was strongly negative (median \( \beta = -3.10, 95\% \) posterior density [PD] < 0). Median species-specific responses ranged from -4.51 for coyote (97% PD < 0) to -0.74 for bobcat (59% PD < 0). Coyote, mule deer (\( \beta = -2.74, 99\% \) PD < 0), and cougar (\( \beta = -3.88, 96\% \) PD < 0) showed strong negative responses to PC1, while elk (\( \beta = -3.37, 86\% \) PD < 0), moose (\( \beta = -3.67, 90\% \) PD < 0), North American porcupine (\( \beta = -3.28, 87\% \) PD < 0), northern raccoon (\( \beta = -3.27, 88\% \) PD < 0), and striped skunk (\( \beta = -3.73, 92\% \) PD < 0) showed moderate negative responses (Figure 2).

Community response to PC2, describing axes of increasing NDVI and decreasing human footprint, was moderately positive (median \( \beta = 2.74, 89\% \) PD > 0).

![Table 1: Number of individual detections for each of the 10 species included in analysis](https://example.com/table1.png)

| Species detections | Common name | Number of detections |
|--------------------|-------------|----------------------|
| *Odocoileus hemionus* | Mule Deer | 577 (North = 232, South = 345) |
| *Canis latrans* | Coyote | 36 (North = 17, South = 19) |
| *Cervus canadensis* | Elk | 29 (North = 20, South = 9) |
| *Alces alces* | Moose | 10 (North = 5, South = 5) |
| *Procyon lotor* | Northern Raccoon | 8 (North = 4, South = 4) |
| *Puma concolor* | Cougar | 6 (North = 3, South = 3) |
| *Vulpes vulpes* | Red Fox | 6 (North = 1, South = 5) |
| *Erzthon dorsatum* | North American Porcupine | 5 (North = 0, South = 5) |
| *Mephitis mephitis* | Striped Skunk | 5 (North = 3, South = 2) |
| *Lynx rufus* | Bobcat | 4 (North = 2, South = 2) |
Median species-specific responses ranged from \(-1.68\) for red fox (68% PD < 0) to 6.41 for cougar (99% PD > 0). Cougar showed the only strong positive response to PC2, while bobcat (\(\beta = 5.63\), 91% PD > 0), coyote (\(\beta = 3.03\), 86% PD > 0), elk (\(\beta = 4.20\), 87% PD > 0), and northern raccoon (\(\beta = 3.54\), 91% PD > 0) showed moderate positive responses (Figure 3).

Community response to distance from I-80 was moderately negative (i.e., site use increased closer to the interstate; median \(\beta = -2.54\), 86% < 0). Median species-specific responses ranged from \(-3.63\) for striped skunk (88% PD < 0) to \(-1.44\) for bobcat (67% PD < 0). Striped skunk, mule deer (\(\beta = -2.67\), 89% PD < 0), and red fox (\(\beta = -3.56\), 89% PD < 0) showed moderate negative responses to distance from I-80 (Figure 4).

### 3.3 Community similarity and species richness

Community similarity strongly decreased in response to PC1 (median \(\beta = -0.12\), \(-100\%\) PD < 0), with mean predicted indices ranging from 0.72 to 0.22 across the range of PC1 values (Figure 5). Community similarity strongly increased in response to PC2 (median \(\beta = 0.15\), \(-100\%\) PD > 0), with mean predicted indices ranging from 0.28 to 0.97 across the range of PC2 values (Figure 5). Community similarity showed no change in response to distance from I-80 (median \(\beta = -0.03\), 77% PD < 0), with mean predicted indices ranging from 0.66 to 0.59 across the range of distance values (Figure 5). Mean community similarity did not differ across the north–south divide (mean difference in community similarity between south and north = \(-0.02\), Bayesian 95% credible interval [CI] = \(-0.09\) to 0.13; Figure 5).

Species richness strongly decreased in response to PC1 (median \(\beta = -1.37\), \(-100\%\) PD < 0), with mean predicted species richness ranging from 7.96 to 2.33 across the range of PC1 values (Figure 6). Species richness strongly increased in response to PC2 (median \(\beta = 1.46\), \(-100\%\) PD > 0), with mean predicted species richness ranging from 3.51 to 10.00 across the range of PC2 values (Figure 6). Species richness strongly decreased with distance from I-80 (i.e., sites closer to I-80 detected more species; median \(\beta = -1.41\), \(-100\%\) PD < 0), with mean predicted species richness ranging from 8.13 to 4.60 across distance values (Figure 6). Mean species richness did not differ across the north–south divide (mean difference in species richness between south and north = \(-0.11\), CI = \(-1.09\) to 0.87; Figure 6).
FIGURE 3  PC2 community occupancy model results. Species-specific posterior densities (a), community logit-linear response (b), and species-specific logit-linear response (c) to PC2, describing axes of increasing NDVI and decreasing human footprint. Y-axes in (a) represents posterior probability density, with listed proportions equal to the posterior probability of a positive or negative response, respectively. Vertical gray line in (a) represents an effect size of 0, and species are identified by symbols. White line in (b) represents the median predicted response of the modeled community to PC2, with the 95% Bayesian credible interval signified by the yellow band. Lines in (c) represent species-specific median responses to PC2. Only species with >85% posterior density greater than or less than 0 are plotted.

FIGURE 4  Distance from I-80 community occupancy model results. Species-specific posterior densities (a), community logit-linear response (b), and species-specific logit-linear response (c) to distance (m) from interstate highway 80. Y-axes in (a) represents posterior probability density, with listed proportions equal to the posterior probability of a positive or negative response, respectively. Vertical gray line in (a) represents an effect size of 0, and species are identified by symbols. White line in (b) represents the median predicted response of the modeled community to distance from Interstate Highway 80, with the 95% Bayesian credible interval signified by the purple band. Lines in (c) represent species-specific responses to distance from interstate highway 80. Only species with >85% posterior density greater than or less than 0 are plotted.
Figure 5  Jaccard community similarity index results. Differences in Jaccard community similarity indices between north–south divide (a) and linear response to PC1, describing axes of increasing human population and housing densities (b); linear response to PC2, describing axes of increasing NDVI and decreasing human footprint (c); and linear response to distance (m) from Interstate Highway 80 (d). Triangles and error bars in (a) represent mean estimates across sites in the north and south and standard errors, respectively. Yellow points in (a) represent site-specific mean estimates. White lines in (b–d) represent median predicted response to PC1, PC2, and distance from Interstate Highway 80, respectively, with 95% Bayesian credible intervals are represented by the colored bands in each. Points in (b–d) represent site-specific estimates, with error bars corresponding with site-specific 95% Bayesian credible intervals.

Figure 6  Species richness results. Differences in species richness between north–south divide (a) and linear response to PC1, describing axes of increasing human population and housing densities (b); linear response to PC2, describing axes of increasing NDVI and decreasing human footprint (c); and linear response to distance (m) from Interstate Highway 80 (d). Triangles and error bars in (a) represent mean estimates across sites in the north and south and standard errors, respectively. Yellow points in (a) represent site-specific mean estimates. White lines in (b), (c), and (d) represent median predicted response to PC1, PC2, and distance from Interstate Highway 80, respectively, with 95% Bayesian credible intervals are represented by the colored bands in each. Points in (b–d) represent site-specific estimates, with error bars corresponding with site-specific 95% Bayesian credible intervals.
4 | DISCUSSION

In this study, we investigated the effects of environmental factors and the presence of an interstate highway on mammalian community composition and space use. We found that community occupancy response, similarity, and species richness were negatively affected by increased housing and population densities and positively affected by increased NDVI and decreased human footprint. Our results were more inconclusive in respect to I-80, as we found only moderate evidence that community composition was affected by distance to the highway.

Our results indicate that increases in human population and housing densities result in decreased mammalian habitat usage in the WMR. Community similarity to the RBC site and community richness had strong negative associations with increased housing and human population densities. These results corroborate our hypothesis that increasing housing and human population densities in the WMR negatively influence mammal site use. Specifically, cougar, coyote, and mule deer had strongly negative associations, and elk, moose, North American porcupine, northern raccoon, and striped skunk had moderately negative associations (Figure 2). These results are in accordance with studies in other ecoregions that illustrate the impact of housing and human population densities on mammalian occupancy (Goad et al., 2014; Hansen et al., 2005; Ordeñana et al., 2010; Polfus & Krausman, 2012).

Community similarity to the RBC reference site, mammalian community site use, and species richness were strongly positively associated with increasing NDVI and decreasing human footprint index (Figure 3). These results suggest that relatively small-scale forest cover and areas with limited human impact may support wildlife occupancy throughout the WMR and are vital to conservation efforts as human influence increases across the landscape (Gray et al., 2016; Watson et al., 2014). Specifically, cougar had a strong positive response, and bobcat, coyote, elk, and northern raccoon had moderately positive responses (Figure 3). Red fox had a weakly negative association with increasing NDVI and decreasing human footprint. This finding is in consensus with other studies that found red foxes to be associated with increased human influence levels and reduced habitat quality (Cove et al., 2012; Fahrig & Rytwinski, 2009; Goad et al., 2014; Rodriguez et al., 2021). The adaptability of red foxes as generalists and their release from larger predators like coyotes may draw red foxes to anthropogenic areas or areas of reduced forest cover (Newsome & Ripple, 2014).

Mammalian response to the highway varied and conflicted with our initial hypothesis that species would avoid I-80. Community response was moderately negative, indicating that species were slightly more likely to use sites closer to the highway (Figure 4). Mule deer, red fox, and striped skunk had a moderate response, with no species having a strong association. This finding is in accordance with previous studies that have found these same species to be human commensals, meaning they are able to co-exist in anthropogenic settings with minimal adaptations or change in habitat use (Goad et al., 2014; Lewis et al., 2021). Another potential explanation for why these species may have increased site use in these areas is the meso-predator release hypothesis, which supports the results of small- to medium-sized mammals showing negative associations with increasing distance from the highway to avoid conflict with larger predators/competitors (Fahrig & Rytwinski, 2009). Furthermore, results from the community similarity analysis indicate no change in response to distance from I-80, and species richness was strongly negatively associated with distance to the highway. The highest levels of species richness were found at sites closest to I-80, with community similarity to the RBC site experiencing no effect as distance to I-80 increased. Another possible explanation for this finding is that mammals may have been drawn to the wildlife overpass located centrally in the study area (Figure 1). Established in December 2018 with dimensions of 97.5 m by 15.2 m, the wildlife overpass is a land bridge designed to encourage animal movement across the six lanes of highway. This study began less than a year after this overpass was finalized, and the length of time before habituation ranges greatly for mammals, taking some species years for utilization (Bond & Jones, 2008; Clevenger & Waltho, 2003). Looking into this specific question further should be a focus of future research.

While sites south of I-80 showed slight differences in average human footprint index, housing density, population density, vegetation, and development levels, community similarity to the northernly located RBC site did not differ across the north–south divide, nor did species richness. As with the community analysis described above, these findings may too be explained, in part, by the presences of the wildlife overpass and other sections of permeability across this stretch of the highway. Although this study is not an evaluation measure nor does it test the effect of the wildlife overpass, the role of the overpass and wildlife fencing mitigation strategies is important in understanding the results of this study, as they have proven successful in reducing animal vehicle collisions and providing safe crossing opportunities for wildlife (Huysjer et al., 2016). Video cameras located at this overpass (Division of Wildlife Resources, personal communication) indicate its use by 7 of the 10 species analyzed in
this study, which may explain the small differences in community similarity and species richness we found across the two study sites, even though there were small-scale differences in environmental factors across the north–south divide (factors we found to be important predictors of community composition and habitat use). In this light, further research and evaluation is needed to determine the effectiveness of this mitigation strategy in increasing mammalian movement and occupancy. Additionally, repeating this study across multiple years may be able to elucidate the overpasses’ gradual effect on community composition in the WMR.

Although camera traps have been used to answer a number of important research questions and their use as wildlife monitoring tools continues to increase (Blount et al., 2021; Burton et al., 2015), it is important to note the potential limitations that can come from using this method to make inferences on multispecies habitat use (Devarajan et al., 2020). Although animal movement near a camera trap is thought to be indicative of species presence across a larger area, it is important to note that fine-scale differences in species-specific space use behavior and relative abundance could bias estimates of occupancy and detection probability, potentially leading to erroneous conclusions about habitat preference across species (Kays et al., 2021). In our study, we paid close attention to topography and terrain when selecting sites for sampling, capitalizing on natural funneling landscape features that would concentrate wildlife activity within a particular area. Furthermore, this methodology was corroborated by pilot studies showing that each of our target species strongly avoided areas of high slope and rugged terrain, concentrating their activity within these landscape features (Green, unpublished data). However, even with prior information about species-specific habitat use, and inclusion of covariates previously shown to affect site use, we acknowledge that unmeasured habitat use heterogeneity across species could be present, and this may affect our ability to generalize these results beyond the unique landscape of the WMR.

Growing anthropogenic development in the WMR characterized by increasing human population and housing densities poses a significant threat to mammal communities, as we found these measures to be detrimental to both community and species-specific habitat use. Due to these strong community and species-specific effects, protecting sites, even at small scales, with increased forest cover and decreasing Human Footprint Index values represents an opportunity for sustainable conservation efforts as human influence increases. The role of the highway in influencing mammalian community composition throughout the WMR remains unclear, as current practices and mitigation strategies may mask the typical effects we would expect to see as a result of fragmentation. Specifically, the presence of a new wildlife overpass may have already begun to alleviate the small-scale effect of habitat fragmentation we initially hypothesized, warranting further evaluation of its effect on community composition, especially across a larger scale than the scope of this study.

AUTHOR CONTRIBUTIONS
All authors in this publication made substantial contributions to developing the ideas, designing the methods, collecting the data, and editing the manuscript.

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CONFLICT OF INTEREST
The authors declare no conflict of interest.

DATA AVAILABILITY STATEMENT
The data utilized in this analysis is available for download at https://emammal.si.edu/analysis/data-download under the project Wasatch Wildlife Watch.

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