

Estimating winter habitat use and population abundances of large ungulates at Fort Riley
Military Reservation

by

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Abstract

Rocky Mountain elk (*Cervus canadensis*) populations have been reintroduced across North America including the Fort Riley Military Reservation (FRMR) in the tallgrass prairie region of Kansas, USA. Although habitat use by elk has been assessed in many areas of North America, we have limited understanding of how elk use space in prairie landscapes and under multiple habitat-management regimes. I used detection/non-detection data from 149 camera-trap sites on FRMR to assess winter habitat use by elk in relation to landscape composition and management efforts. Specifically, I used single-season occupancy models to investigate the effects of woody encroachment, prairie landcover, food plots, distance to refugia and water, and burn history on site occupancy by elk in January - February 2023. The most supported-model indicated that burn timing was the best predictor of elk habitat use. Elk were more likely to occur at sites recently burned than sites that were not burned > 24 months. Interestingly, I did not detect an effect of amount of food plots or permanent water sources near sites. The results demonstrate that managed fire regimes may benefit elk during winter in the lower Great Plains where winter temperatures can be moderate. Precise estimation of population abundances is a vital consideration for biologist who develop harvest regulations and manage populations. Traditionally, methods such as aerial census surveys and spotlight surveys are used to estimate abundances of Rocky Mountain elk and white-tailed deer (*Odocoileus virginianus*). These approaches can be logistically challenging and may not reveal precise population estimates. Thus, there is a need to evaluate the effectiveness of complimentary abundance estimation procedures. I used presence/absence data from 146 camera-trap sites to evaluate the usefulness of space-to-event (STE) models for estimating deer and elk abundance on Fort Riley Military Reservation in February 2023. I compared The STE estimates to complete census surveys (elk)

and distance sampling (deer) currently used by FRMR biologists. Abundance estimates derived from STE models for elk (197.8, 95% CI = 163.7 - 238.9) and deer (2,290.0, 95% CI = 2,140.9 - 2,449.5) were slightly greater than those derived from census sampling for elk (102) and distance sampling for deer (1,526.9, 95% CI = 1,078.8 - 2,160.9), though precision was much better with STE estimators. The results demonstrate that STE models can be a useful and cost-effective tool and the primary impact of using these methods is that they allow for concurrent estimation of elk and deer abundances at large spatial scales. The primary contribution of this work lies in the science-based information provided to FRMR biologists. Specifically, it enhances their understanding of habitat management techniques they use through prescribed burns and offers an efficient method to estimate ungulate abundance within the FRMR area.

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Dedication

I would like to dedicate this to my family who have always cheered me on to follow my dreams. To my veteran brothers and sisters, “Til Valhalla” and I love you. Also, to all of my teammates and coaches/professors at Combative Sports Center, thank you for teaching me a new philosophy in this growth stage of life.

Chapter 1 - Introduction

Rocky Mountain elk (*Cervus canadensis*, hereafter ‘elk’) have been pursued by European and indigenous hunters for the last 10,000 to 20,000 years in North America. Since the 1800’s elk and white-tailed deer (*Odocoileus virginianus*, hereafter ‘deer’) are both prevalent big game species in North America (Miles 1895; Rosatte et al. 2007; Lee Lyman 2011). Elk which are considered “old world deer” and deer, “new world deer”, ancestors likely originated in central Asia in the Pliocene era (Lorenzini and Garofalo 2015) and all are in the family *Cervidae*. Elk are mammalian cervids whose weight can range from 226 – 454kg and have large antlers that can weigh nearly 20kg (Hobbs et al. 1981; O’Gara and Dundas 2002). Elk, also known as wapiti, are a generalist herbivore species that feed on grasses, shrubs and other woody plant species (Poole and Mowat 2005). Unlike elk, deer are much smaller in body size and weight (range = 70 – 181kg) and are considered browsers that feed on woody plants and both elk and deer will feed on food plots if they are available (Månsson et al. 2015). The great plains provide great habitat for elk that ranged widely through this area in abundance until their extirpation in 1800s, and deer that were similarly extirpated from much of the region, but subsequently recovered through multiple mechanisms. Due to overharvesting from early settler’s elk populations in the great plains have relied on reintroduction efforts (Rosatte et al. 2007). Deer populations have been relatively stable in recent time with some populations broadly expanding west and increasing in the Midwest (Kansas) portion of the US and high densities in the eastern (North Carolina, Alabama, and Louisiana) based on historical trends (Hanberry and Hanberry 2020). Estimating big game population densities can be vital for biologists and wildlife managers to understand the best ways to implement harvest regulations and maintain habitat (Fuller et al. 2016).

There are many ways to manage habitat so that ungulates forage is available and hopefully optimal. Fire can play a major role in maintaining ecosystems by optimizing habitat to provide forage for ungulates (Knapp et al. 1998; Allred et al. 2011; Proffitt et al. 2019). Fire can remove detritus and leaf litter which can provide habitat for new growth as well as control for woody encroachment to maintain grasslands (Johnson and Matchett 2001). When fire occurs in shrubby habitats it can create an environment where new shoots on browse can grow that are dense in vitamin and minerals which provide beneficial forage for ungulates (Allred et al. 2011). Prescribed burning is a commonly used tool to maintain grasslands in the tallgrass prairie including parts of Fort Riley Military Reservation.

Rocky Mountain elk have been reintroduced in northeast Kansas at the Maxwell Wildlife Refuge in the early 1950's and in 1986 biologists and wildlife managers contributed elk to Fort Riley Military Reservations (hereafter 'FRMR'; Pitts et al. 1987). The elk population had seemingly peaked in the early 2000's to a point where they began causing damage in private agriculture fields adjacent to FRMR. State biologist and FRMR staff decided to increase lottery harvest tags for elk to mitigate crop damage (Conard et al. 2010; Peek 2021). Habitat management on FRMR includes maintaining and restoring grasslands, timber stand improvement, controlling for both noxious weeds and woody encroachment with prescribed burning, chemical and mechanical controls as well as implementing diversionary food plots to attract and keep elk and deer on the installation. Military training operations occur regularly on FRMR. There are two impact areas that restrict access to hunters and recreationists. However, elk and deer can maneuver these areas relatively unrestricted. Historically, biologists have relied on distance spotlight surveys to estimate deer abundances and aerial census surveys using helicopters to estimate elk populations at FRMR (Roberts et al. 2006; Collier et al. 2013).

My thesis research focuses on investigating elk winter habitat use within Fort Riley Military Reservation located in the tallgrass prairie region. This area is undergoing multiple habitat management techniques and has unique military operations with refuge areas. I used occupancy models to determine if site occupancy probabilities would be positively related to proportion of food plots, woody encroachment, or most recently burned areas. I also predicted that distance to refuge areas (impact areas) and permanent water sources would negatively impact elk occupancy. I also evaluated novel methods of estimating the abundance of multiple unmarked species and compared results to previous methods used by wildlife biologists. To estimate abundances for elk and deer I used novel Space-to-Event models and compared them to aerial census surveys for elk and distance spotlight surveys for deer. I used motion-triggered and time-lapsed photos compared to previous studies as well as developed a tool to use in the field to estimate the cameras viewshed.

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Chapter 2 - Winter habitat use by a restored Rocky Mountain elk (*Cervus canadensis*) population in the tallgrass prairie

Introduction

Rocky Mountain elk (*Cervus canadensis*, hereafter ‘elk’) were once abundant across the North American Great Plains before near extirpation by European colonizers in the nineteenth century (Rosatte et al. 2007; Lee Lyman 2011). Reintroduction efforts in North America have largely been successful with localized populations east of the Rocky Mountains being reestablished (Popp et al. 2014). For instance, the state of Kentucky reintroduced 1,046 elk in the Appalachian region (O’Gara and Dundas 2002; Larkin et al. 2003) with populations now exceeding 10,000 individuals (Popp et al. 2014). Post-translocation efforts must consider habitat quality assessments to evaluate long-term success of restored wildlife populations (Cheyne 2006), including restored elk populations (Quinlan et al. 2022).

Habitat use and selection by elk is well documented (Hobbs et al. 1981; Pearson et al. 1995; Sawyer et al. 2007; Smith et al. 2018; Quinlan et al. 2022) though our understanding of these behaviors in expanding and reintroduced elk populations in the Great Plains is relatively unclear. Elk are generalist herbivores that will browse and graze (Poole and Mowat 2005) and select habitat with high-quality forage and new growth vegetation (Merkle et al. 2016). Elk will often select refuge from humans and predators in areas with hiding cover, physical barriers, or low road densities (Thomas et al. 1979; Lyon 1992; Proffitt et al. 2010; Prokopenko et al. 2017; Pero et al. 2023). Woody vegetation can be used as both refuge and thermal cover (Jost et al. 1999; Baasch et al. 2010; DeVore et al. 2016), though the relative quality of these areas may vary by season (Quinlan et al. 2022). Availability of water is important for elk and individuals

may travel 1.21 – 1.61 km for water sources during winter (Mackie 1970; Thomas et al. 1979). In non-forested regions elk select landscapes with south-facing slopes, lower elevations, managed food plots, and recently burned areas (Unsworth et al. 1998; Boyce et al. 2003; Sawyer et al. 2007; Pero et al. 2023; Sittler et al. 2019). Winter diet selection for elk is closely related to quality and concentration of nutrients (Hobbs et al. 1982).

Elk were first reintroduced to Kansas, USA in 1951. Subsequent efforts to reintroduce elk into Kansas included translocating a small population (n = 12) to Fort Riley Military Reservation (FRMR) in 1986 (O’Gara and Dundas 2002; Pitts et al. 1987). By 1999, the FRMR herd increased to ~325 individuals necessitating managers to reduce the population through regulated harvest to mitigate crop damage on neighboring agricultural fields (O’Gara and Dundas 2002; Conard et al. 2010; Peek 2021). This region was historically prairie landcover, and fire played a role in maintaining these native ecosystems (Fuhlendorf et al. 2011). Currently, grasslands on FRMR are managed with prescribed burning to set back woody encroachment and sustain natural ecosystem function. Additionally, FRMR manages various food plots to bolster local wildlife populations and to reduce agricultural damage by elk on neighboring properties. Winter temperatures in Kansas are relatively mild and elk consume greater diversity of grasses during this time (Conard and Gipson 2012). Common elk predators (e.g., gray wolves [*Canis lupus*], mountain lions [*Puma concolor*], black bear [*Ursus americanus*]) were extirpated from Kansas during European colonization. Thus, elk in this region are likely making winter habitat-use decisions without regard to predation risks common in western elk populations. Human activity on FRMR is highly restricted and used mostly for military activities. Hunting is regulated on FRMR (including very limited elk hunting) and only allowed in designated areas away from ongoing military activities.

My objective was to investigate winter habitat use by a restored elk population in the tallgrass prairie region. I hypothesized food plots would provide supplemental food resources for elk during winter and predicted site occupancy probabilities would be positively related to the proportion of food plot landcover surrounding sites. I predicted site occupancy probabilities would be positively related to the proportion of woody encroachment surrounding sites as these areas may provide refuge. If military impact areas on FRMR provide refuge from recreationists (e.g., upland gamebird and white-tailed deer [*Odocoileus virginianus*] hunters; shed antler hunters) I predicted site occupancy probabilities would be higher in these areas and therefore negatively related to distance from these areas. Because water can be a limiting resource for large ungulates, I predicted site occupancy probabilities would similarly be negatively related to distance to permanent water sources. Finally, I hypothesized that recently burned areas would have forage with increased nutritional benefits to elk and predicted site occupancy probabilities would be negatively related to time (in days) since sites were last burned.

Study Area

The study occurred on Fort Riley Military Reservation (FRMR; 39.1930°N, 96.8129°W; ~ 40,900 ha; Fig. 2.1) located in northeast Kansas, USA. Due to safety concerns and FRMR regulations, two large areas were inaccessible to researchers including an impact area (~5,800 ha) and a range complex (~2,545 ha) (hereafter ‘Impact areas’). These areas are subject to military training operations and inaccessible by hunters and other recreationists though elk commonly move through them unrestricted. I also excluded the southernmost portion of FRMR which included high-density housing and administrative infrastructure rarely used by elk. Landcover on FRMR was ~74% grassland dominated by big bluestem (*Andropogon gerardii*), indiangrass (*Sorghastrum nutans*), little bluestem (*Schizachyrium scoparium*), and switchgrass

(*Panicum virgatum*; Busby et al. 1996; Conard et al. 2010). Woody vegetation made up ~21% of landcover including ash (*Fraxinus* sp.), American elm (*Ulmus americana*), eastern redcedar (*Juniperus virginiana*), hackberry (*Celtis occidentalis*), and a mixed oak assemblage of burr oak (*Quercus macrocarpa*), and chinquapin oak (*Quercus muehlenbergii*; Freeman and Delisle 2004). Food plots and cultivated fields contributed to ~1% of FRMR landcover and consisted of alfalfa (*Medicago sativa*), corn (*Zea mays*), sorghum (*Sorghum bicolor*), soybean (*Glycine max*) and wheat (*Triticum aestivum*). Remaining landcover was characterized as developed, quarry, firebreaks, or other. Managers used prescribed fire with spatial and temporal variation in burn frequency throughout FRMR. White-tailed deer were the only other ungulate on FRMR, and all elk predators were likely absent. Regulated elk hunting occurs on FRMR though not during the sampling timeframe. Our data collection took place during the winter season between 03 January 2023 – 07 February 2023. During this time, the study area received 2.50 cm of precipitation and had an average temperature of -0.01°C, (range = -15.50°C - 19.44° C; National Weather Service 2023).

Methods

Site Selection and Camera Deployment

I identified 150 sites across FRMR using a randomized ‘fishnet design’ tool (ArcGIS Pro v. 3.1.1). This approach allowed me to identify random sampling points across the study extent excluding both impact areas and garrison (Fig. 1). Average Euclidian distance between sites was 989 m (range = 427 – 3586 m). Each site included a single, unbaited camera (Bushnell Trophy Cam®; Model 119717CW) on the nearest structure (e.g., tree, fence post) at ~1 m height and oriented north or south to avoid solar glare and would be towards the most unobstructed terrain. I adjusted exact placement of cameras at sites to nearest suitable attachment structure and removed

tall vegetation in the field of view to a max distance of 50m from the camera at a width of ~1m with a hand-held electric hedge trimmer. Cameras were equipped with memory cards (16 GB) and set to trigger using both motion-sensor (three second interval between triggers) and time-lapse (one photo every 30 min) functions. I used both of these trigger functions to accommodate a concurrent study assessing elk densities on FRMR. Cameras were in place and active an average of 33 days (range = 23 – 36 days) Once cameras were retrieved, I transferred photo data to an online photo processing software (www.wildlifeinsights.org) and I manually assessed detection/non-detection of elk at each site. I established a 36-day detection history for each site (1 = elk observed during a particular day, 0 = no elk observed during a particular day). I removed one site from the analysis because of a camera malfunction resulting in a total of 149 sites for analyses.

Characterizing Landcover

I used detailed digital landcover data collected and cataloged by the Kansas Biological Survey (KBS; www.biosurvey.ku.edu). Landcover types throughout FRMR were characterized by vector polygon layers with a 1-m spatial resolution. I created 250-meter radius circular buffers centered on each site (ArcGIS Pro v. 3.1.1) from which to extract landcover information scale-specific to elk habitat use. I chose this scale to be consistent with past elk resource selection studies (e.g., Sawyer et al. 2007; Rumble and Gamo 2011). Within each buffer, I calculated the proportion grassland, agriculture, and woody cover. I combined all grassland types into a single layer (hereafter ‘Grassland’; Table 2.1) because of similar functional characteristics. I combined food plots and cultivated fields into a single variable (hereafter ‘Foodplots’) because both serve as a non-native, supplemental food sources planted and maintained by managers. I could not ascertain food-plot type from this spatial data, and I did not have enough sites to

partition food-plot type into multiple categories for comparison in analyses. I also combined shrubland, forest, and tree plantation categories as a single variable representing woody encroachment (hereafter ‘Woody’; Table 2.1). Because hunters and other recreationists cannot access impact areas on FRMR, these impact areas may serve as refugia from human contact. Additionally, supplemental food and water resources may also be important to elk habitat use. To assess spatial proximity to these areas, I measured the Euclidean distance (m) of each site to the nearest impact area (Distance to Impact) and permeant water sources (Water Distance; Table 2.1). I normalized all variables prior to analyses (Miguet et al. 2017; Yeiser et al. 2018; Werdel et al. 2022).

Burn History

Using detailed spatiotemporal burn history on FRMR since 2000 I created polygon layers using ArcGIS Pro, categorizing the burn date and burn frequency within the past 10 years. For each site I established the most recent day of a prescribed burn (hereafter ‘Days Since Burned’; average = 751, range = 45 – 3,653; Table 2.1). I used the feature ‘spatial joined’ of the burn history polygon layer to the 250-meter radius buffer for every year for the past 10 years. If the proportion of the site had <20% burned within the 250-meter buffer on a burn date, it was considered below the cutoff threshold and was not counted as burned during that particular year. If the site was not burned within the past 10 years I established the site’s burn day as 3,653. I generated a variable representing annual burn frequency at each site (hereafter ‘Burn Frequency’; average = 4, range = 0 – 8; Table 2.1) by summing number of years a burn occurred within each site buffer ($\geq 20\%$ of the buffer) since 2013.

Single-season Occupancy Analysis

I used single-species, single-season occupancy models (unmarked package v.1.2.5; MacKenzie et al. 2002; Fiske and Chandler 2011; Sollmann 2018) to evaluate the relative importance of the measured covariates on winter habitat use by elk. To account for imperfect detection, I assessed two detection covariates that could potentially affect the ability to detect elk at sites if present. I quantified ‘Day of Year’ (DOY; days since 01 January 2023; range = 3-36) to determine if within-season effects would affect the ability to detect elk if present (Weir et al. 2005; Werdel et al. 2022). Because of site-level variation in the number of days cameras were active (average = 33, range = 23-35 days), I created a variable to account for sampling duration (DA). I created four detection models using the single and additive effects of both covariates (DA, DOY, DA + DOY) along with an intercept-only model (*Intercept Only*). My candidate set of occupancy models (n = 27; Table 2.3) included single (Days Since Burned, Burn Frequency, Water Distance, Food Plot, Grassland, Distance to Impact, and Woody) and additive effects of my measured covariates along with a constant model (*Intercept Only*). To limit the number of models, and to avoid overparameterization, I only considered additive models with 2 covariates. I did not include ‘Woody’ or ‘Grassland’ covariates in the same model because of multicollinearity issues ($|r| > 0.59$). I evaluated support for models using Akaike’s Information Criterion corrected for small sample sizes (ΔAIC_c , Burnham and Anderson 2004). I first identified the most-supported detection model, then used this model in subsequent occupancy models. I evaluated all of the competitive models ($\Delta AIC_c \leq 2.00$) using an information-theoretic approach (Burnham and Anderson 2002; Arnold 2010; Sutherland et al. 2023).

Results

I documented 1,173 photos of elk out of a total of 316,823 photos in 4,908 camera-trap nights. I detected elk at 38 sites out of 149 sites (naïve occupancy = 0.26). The constant model

was the most-supported detection model (Table 2.3) and I used this model to fit subsequent occupancy models. Pre-survey detection probability estimate was 0.05 (SE = 0.01) and the probability of missing an elk was 0.157.

Five occupancy models were competitive (Table 2.1). The second-ranked, single-effect model (Days Since Burned) was the most parsimonious, and this covariate was also present in all competitive models (Table 2.3). Models including ‘Days Since Burned’ contributed to 0.89 of all model weights (Table 2.1). Elk were more likely to use areas recently burned ($\beta = -0.68$, SE = 0.30; Fig. 2.2). Because of relatively little change in model fit between competitive models (Table 2.3), and 95% CIs overlapping 0 for other covariate estimates in remaining models (Table 2.2), additional covariates in competitive models were likely ‘piggybacking’ on the explanatory power of ‘Days Since Burned’.

Discussion

Maintaining habitat quality for restored elk populations should be a priority for wildlife managers to ensure successful population-management towards restoration goals. My results suggest winter habitat use by elk in tallgrass prairies is influenced by temporal variation in burn timing, suggesting elk will use fresh growth from burned areas as a high-quality food resource. These results support findings from previous studies (Long et al. 2008; Sittler et al. 2019). Elk were more likely to use areas recently burned, and my models suggest that these areas may be more important than established and partially harvested food plots (most corn fields were harvested with spillage available) or woody habitat. Providing good-quality habitat may fulfill nutritional needs for elk and increase survival or enhance population persistence (Christianson and Creel 2007). High quality wintering habitat could benefit elk populations when milk production is reduced in lactating females from December to February (Hudson et al. 2002;

Cook et al. 2013). Additionally, diminishing fat reserves in juveniles may occur during winter as individuals are compensating from a transition from milk to vegetation. Weight loss occurs rapidly in winter and more dramatically for larger elk (>250kg, Hudson et al. 2002) because of larger energy demands. An increase in grasses in elk diets may help reduce weight loss in winter (Christianson and Creel 2009). Previous stable-isotope research on FRMR found elk primarily consumed cool-season grasses during summer and winter (Conrad and Gipson 2012). Winters in Kansas are relatively mild compared to other western areas that elk occupy, and I noted fresh, sprouting grasses and sedges at the recently burned sites even though I sampled in January and February.

Grazed grasslands and shrubby vegetation undergoing postfire succession can provide new sprout growth that is nutritionally beneficial and desirable to elk in winter (Johnson and Matchett 2001; Lyon and Christensen 2002; Nichols et al. 2021). When woody cover becomes closed canopy, the taller and more dominant woody species can outcompete shade-intolerant browse. These areas can have lower forage quality for elk and exhibit lower burn success rates as elevated moisture contents reduce fire ignition (Nanney et al. 2018). Areas burned more frequently generally have reduced cover and amounts of woody vegetation. However, the variable 'Burn Frequency' was not well-supported in my models. This could suggest that elk prioritize recent burns because of immediate physiological demands and that burn frequency is not as important as the area being recently burned (Van Dyke and Darragh 2007). It is likely that elk are making spatial foraging decisions based on finer-scale spatial information than I had available for my study. Future studies assessing fine-scale movement and foraging patterns may help reveal how elk use areas that have temporal variation in burn frequency and/or burn intensity.

When the distribution of water resources is limiting, elk will use habitats near permeant water (Beck et al. 2006). I did not detect an effect of water availability in my models suggesting that water is not a limiting resource for elk on FRMR. It is likely that my coarse measurement of water availability from remote sensed data did not capture other sources of water that were present on the landscape. Contrary to my hypothesis, elk did not use woody cover as much as expected. This may be because elk do not need to seek woody cover during our mild winters and solar reflection could be more effective for thermoregulation in these areas (Skovlin et al. 2002; Pool and Mowat 2005). I also did not detect an effect for ‘Impact Areas’ as being spatial refuges for elk. I could not access these areas to establish camera sites, so it is plausible (and likely) elk may have been using these areas extensively. My attempt to model this effect likely did not fully reflect the refuge potential of these areas. Again, future studies that incorporate GPS tracking technology would uncover finer spatial resolution in habitat use by elk on FRMR.

I observed elk using food plots extensively during the growing season on FRMR. Many food plots were either harvested (e.g., corn, soybean) or completely foraged by white-tailed deer, turkey (*Meleagris gallopavo*), and elk prior to my winter sampling season. Additionally, I observed elk using corn and soybean agricultural fields adjacent to FRMR during the duration of my study. Although I did not detect that food plots on FRMR were important for winter habitat use by elk, this may be because they had reduced forage capacity at this time. The random placement of cameras may also impact detection of use in some of these areas that make up such a small percentage of FRMR. Managers on FRMR use food plots, in part, to reduce human-elk conflicts on neighboring agricultural fields by providing agricultural resources for elk on FRMR during the growing season (May – September). It is currently unclear if food plots on FRMR

provide similar or better nutritional quality for elk during winter. However, my models suggest that elk were using recently burned grasslands over established food plots on FRMR.

Management Implications

Implementing an intentional 2-3 year prescribed fire rotation may support winter nutritional demands for elk by maintaining native grassland habitat, promote sprouting of desirable shrubby forage, as well as to safely manage fuel-control. Creating a spring and fall burn schedule may provide recently burned areas across a wide temporal extent for elk and potentially increase elk fat reserves for winter. It is important to incorporate spatiotemporal patterning in burn regimes to maintain natural variation in habitat heterogeneity across landscape scales. Additionally, continuing to target areas encroached by eastern redcedar via mechanical and burn management may be beneficial for creating good-quality elk habitat in tallgrass prairie ecosystems.

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Tables

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Table 2.1.

Descriptions of covariates and associated summary statistics used to model initial site use probability of elk (*Cervus canadensis*) in Fort Riley, Kansas, USA. Days Since Burn = number of days since site was last burned; Burn Frequency = number of burns occurring within the past 10 years surrounding site; Grassland and Woody = proportions of respective landcover types within a 250-m buffer surrounding sites; Water Distance = Euclidean distance (m) to nearest water source. Mean = average; Min = minimum; Max = maximum; SD = standard deviation.

Covariate	Description	Mean	Min	Max	SD
<i>Days Since Burn</i>	Number of days since site was burned within a 250-m buffer surrounding site.	750.78	45.00	3653.00	755.80
<i>Burn Frequency</i>	Number of times site was burned in the past 10 years (2013-2023) within a 250-m buffer surrounding site.	3.80	0.00	8.00	1.70
<i>Grassland</i>	Proportion of grassland landcover (vector polygon at 1-m spatial resolution) within a 250-m buffer surrounding site.	0.83	0.02	1.00	0.22
<i>Water Distance</i>	Euclidean distance (m) from camera site to permeant water source.	719.16	0.00	2368.78	547.70
<i>Woody</i>	Proportion of woody forest and shrubland (vector polygon at 1-m spatial resolution) within a 250-m buffer surrounding site.	0.13	0.00	0.97	0.20

Table 2.2.

Model-selection statistics depicting the beta coefficients on the logit scale and the 95% confidence intervals (lower and upper in parentheses) associated for competitive models of single-season occupancy by elk (*Cervus canadensis*) within Fort Riley Military Reservation, KS, USA. Days Since Burn = number of days since site was last burned; Burn Frequency = number of burns occurring within the past 10 years surrounding site; Grassland and Woody = proportions of respective landcover types within a 250-m buffer surrounding sites; Water Distance = Euclidean distance (m) to nearest water source.

Occupancy Model (logit-scale)	Days Since Burn	Burn Frequency	Grassland	Water Distance	Woody
Days Since Burn + Burn Frequency	-0.93(-1.60, -0.26)	-0.42(-0.95, 0.11)	-	-	-
Days Since Burn	-0.68(-1.27, -0.09)	-	-	-	-
Days Since Burn + Grassland	-0.72(-1.31, -0.13)	-	-0.32(-0.81, 0.17)	-	-
Days Since Burn + Water Distance	-0.69(-1.28, -0.10)	-	-	-0.28(-0.73, 0.17)	-
Days Since Burn + Woody	-0.70(-1.29, -0.11)	-	-	-	0.23(-0.24, 0.70)

Table 2.3.

Single-season occupancy models explaining detection and occupancy of elk (*Cervus canadensis*) across sites ($n = 149$) distributed within Fort Riley Military Reservation in the Flint Hills ecoregion of Kansas, USA. Model set includes models within $\sum w_i = 0.95$, used for model-averaged estimates. Intercept model is also provided. I used camera-trap data at each site to document the presence of elk in wintering season (03 January 2023 to 07 February 2023). Detection covariate is held constant in occupancy models. Days Active = number of days camera was active (range = 23-36); Day of Year = day of the year (1 – 365); The Grassland, Woody, and Food Plot covariates represent proportions of landcover types within a 250-m buffer surrounding camera sites. Days Since Burn represents amount of days since the site was last burned (range = 45 – 3653) by a proportion of $\geq 20\%$ within the same buffer. Burn Frequency represents amount of burns the site was burned within the past 10 years (range = 0- 8) by a proportion of $\geq 20\%$ within the same buffer. Covariates Water Distance and Distance to Impact are measurements of the Euclidean distance (m) from closest camera site. Models are ranked by descending ΔAIC_c values (Akaike’s Information Criteria, adjusted for small sample size). w_i = model weight. LL = (Log likelihood); K = number of parameters in each model.

Model	K	ΔAIC_c	w_i	LL
<u>Detection</u>				
Intercept only	2	0.00	0.59	-374.95
Days active	3	2.07	0.21	-374.95
Day of Year	3	2.08	0.21	-374.95
<u>Occupancy</u>				
Days Since Burn + Burn Frequency	4	0.00	0.22	-370.17
Days Since Burn	3	0.56	0.17	-371.51
Days Since Burn + Grassland	4	0.73	0.15	-370.54
Days Since Burn + Water Distance	4	1.05	0.13	-370.70
Days Since Burn + Woody	4	1.66	0.10	-371.01

Days Since Burn + Distance to Impact	4	2.47	0.06	-371.41
Days Since Burn + Food Plot	4	2.62	0.06	-371.48
Intercept Only	2	5.36	0.02	-374.95
Water Distance	3	6.13	0.01	-374.30
Grassland	3	6.45	0.01	-374.45
Distance to Impact	3	6.61	0.01	-374.53
Distance to Impact + Water Distance	4	7.03	0.01	-373.69
Woody	3	7.04	0.01	-374.75
Grassland + Water Distance	4	7.12	0.01	-373.73
Food Plot	3	7.42	0.01	-374.94
Burn Frequency	3	7.44	0.01	-374.95
Grassland + Distance to Impact	4	7.70	0.01	-374.02
Food Plot + Water Distance	4	8.24	0.00	-374.29
Burn Frequency + Water Distance	4	8.24	0.00	-374.29
Distance to Impact + Woody	4	8.27	0.00	-374.31
Grassland + Burn Frequency	4	8.54	0.00	-374.44
Food Plot + Grassland	4	8.55	0.00	-374.45
Distance to Impact + Burn Frequency	4	8.68	0.00	-374.51
Food Plot + Distance to Impact	4	8.69	0.00	-374.52
Burn Frequency + Woody	4	9.14	0.00	-374.74
Food Plot + Woody	4	9.14	0.00	-374.75
Food Plot + Burn Frequency	4	9.53	0.00	-374.94

Figures

Figure 2.1.

Study area of Fort Riley Military Reservation within the Flint Hills ecoregion (outlined in bolded black polygon) of Kansas, USA. I placed a single, unbaited camera trap at each study site ($n = 149$) and assessed elk (*Cervus canadensis*) habitat use from 03 January – 07 February, 2023. The northwest and eastern off-limit areas are subject to military ordinance and small arms fire and recreationist cannot access them. Southern off-limits area is the garrison area comprised of military housing and administrative buildings. Habitat within the surveyed area consist of continuous potential elk habitat consisting of a mosaic of natural and anthropogenic land cover types.

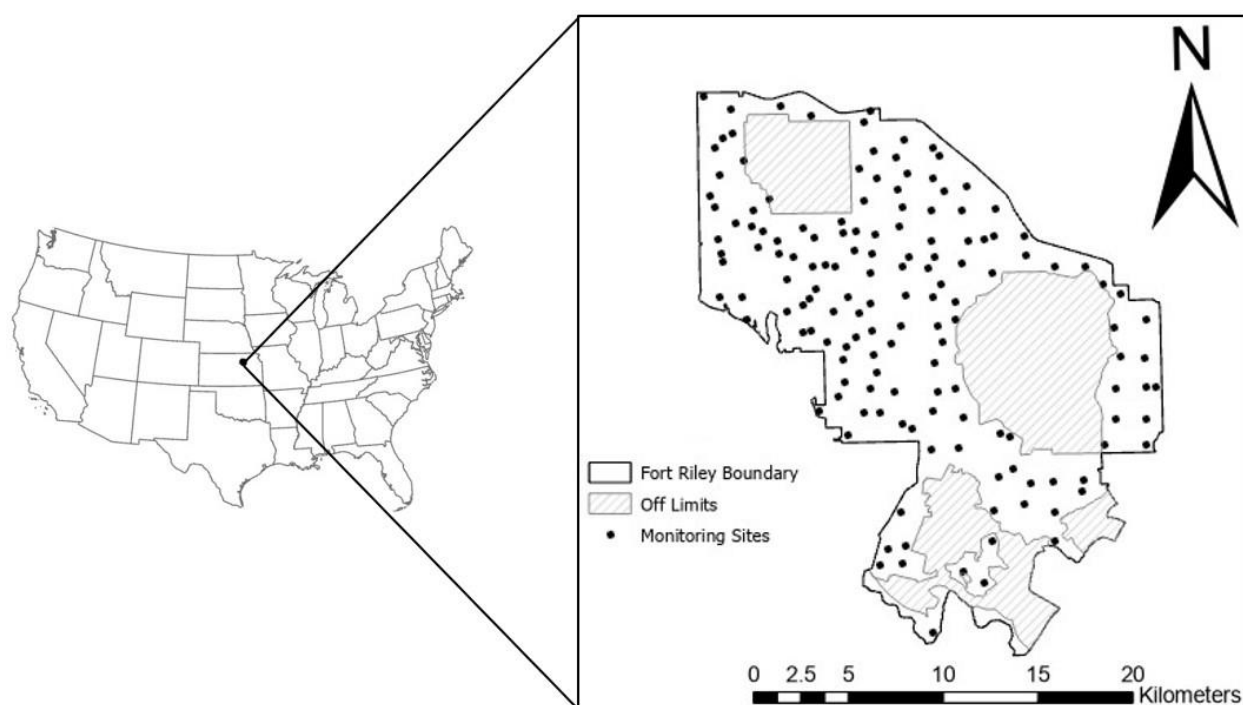
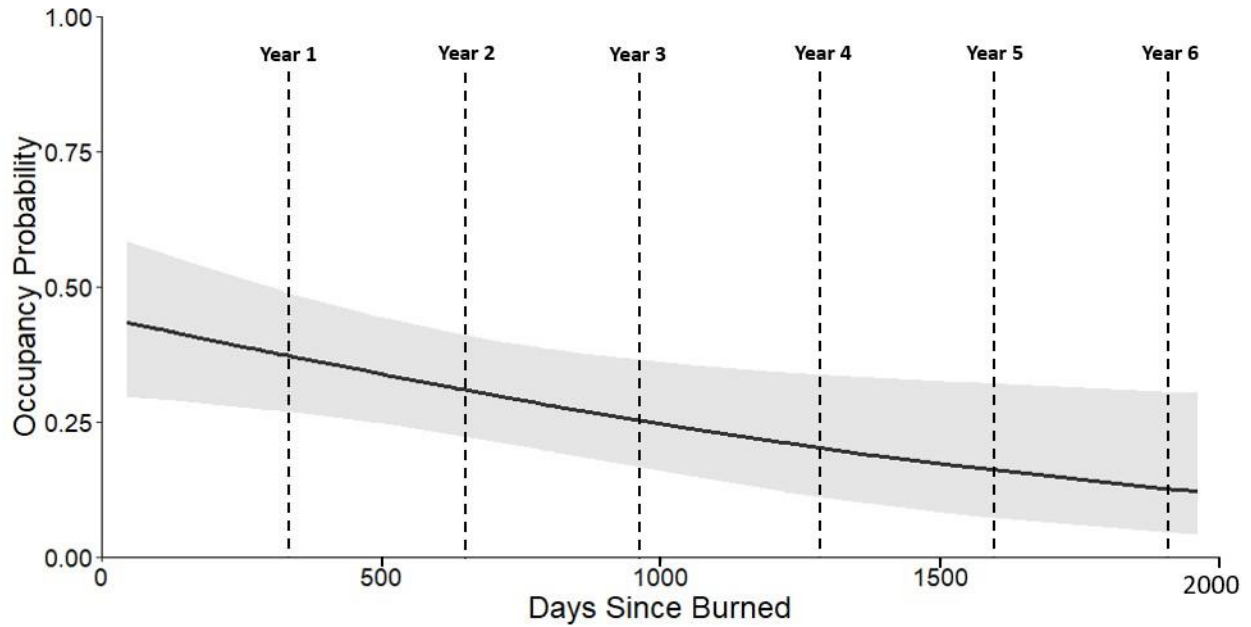


Figure 2.2.

The effect of burn timing on site occupancy probabilities for elk (*Cervus canadensis*) on Fort Riley Military Reservation in Kansas, USA. I assessed site ($n = 149$) occupancy by elk using camera traps distributed across Fort Riley Military Reservation during winter 2023. Occupancy probabilities (solid lines) and 95% confidence intervals (grey fill) were estimated from the most-supported, single-season occupancy model.



Chapter 3 - Comparing multiple techniques to estimate white-tailed deer and Rocky Mountain elk abundances

Introduction

Precise estimates of population densities are important for wildlife managers and biologists when considering habitat management efforts or to ensure that species' harvest quotas are sustainable (Fuller et al. 2016). There are many ways to estimate population densities or species' abundances, though they vary in precision and costs to implement. For instance, complete aerial surveys can identify population abundances with precision but they are costly and may be biased by landscape features affecting detection (Cogan and Diefenbach, 1998). Mark-resight methods using camera trap data can be useful if individuals within the population can be identified by unique fur patterns, tagging, or painting/markings (Alonso et al. 2015). However, time and effort to implement this approach can be costly and time consuming especially if biologists need to conspicuously mark large numbers of individuals in the population. Novel methods for estimating abundance of unmarked species (e.g., space-to-event models) are being tested and implemented to help increase efficacy and improve efficiency.

Rocky Mountain elk (*Cervus canadensis*, hereafter 'elk') and white-tailed deer (*Odocoileus virginianus*, hereafter 'deer') are native to Kansas, USA and are managed via regulated harvest (Miles, 1895). Reintroduction of elk in Kansas began in 1951 at Maxwell Wildlife Refuge followed by subsequent reintroduction efforts at Fort Riley Military Reservation (hereafter, 'FRMR') in 1986 (Pitts et al. 1987). Because of damage to crop fields adjacent to FRMR the elk herd was reduced from ~325 to ~110 in 1999 by increasing harvest quotas in FRMR (Conard et al. 2010; 2020-21 Kansas Elk Harvest Report, Matt Peek, 2021). Fort Riley

Military Reservation Environmental Division (hereafter, 'FRMRED') attempts a yearly aerial census surveys conducted by UH-60 Black Hawk helicopters when logistics and weather permit. Although this effort can reliably provide complete counts of the elk population on FRMR because of large, open prairie landscapes, availability of helicopters for surveys is largely impossible to predict as military missions take priority over elk counts. Additionally, costs to implement these surveys can be extremely high. Deer abundances on FRMR are estimated annually using more traditional distance surveys by observers driving transects at night using spotlights to detect deer. This approach can also be logistically and financially challenging and biased by detection issues and survey effort (Beaver et al. 2014). Nonetheless, managers on FRMR need precise annual estimates of elk and deer populations to evaluate habitat management activities, implement harvest regulations, and to track annual changes in population size.

Camera traps can be a relatively cheap and valuable tool for estimating abundance, density, and space use of an unmarked species by incorporating detection probability, true presence or absence of an individual, and other variables in different models (Sollmann, 2018). Space-to-event (hereafter, 'STE') models are a novel method which uses detection/non-detection data from camera traps to estimate unmarked species abundance within a defined study area (Moeller et al. 2018). STE models are generally simpler to implement than time-to-event (TTE) models because STE models do not require movement rates from marked individuals within a population (Moeller et al. 2018). When implementing STE models, cameras may use synchronous time-lapsed (i.e. all cameras are taking picture at the same time of day) and/or motion-sensor features (Ausband et al. 2022). More precise estimates of abundance are realized if researchers can accurately estimate camera viewable angle of the camera lens and the unobstructed landscape area within the viewshed of individual cameras (Moeller et al. 2023). For

instance, researchers have used STE models to derive precise estimates of elk and wolf (*Canis lupus*) populations (Moeller et al. 2018; Ausband et al. 2022). Additionally, accounting for daily obstructed viewshed is crucial during times with inclement weather such as snow, frost, dew, fog, etc. (Moeller et al. 2023). Although STE modeling approaches can be relatively cheap and logistically easy to implement at large spatial scales, it is unclear how these models perform (i.e., precision of estimates) compared to traditional approaches to estimate abundance (e.g., complete aerial census, distance sampling).

I used game trail cameras randomly placed within FRMR during Jan-Feb 2023 as a passive way to sample presence or absence of elk and deer. My objective was to compare the precision of STE model estimates against estimates of abundance derived from concurrent distance surveys (deer) and aerial census surveys (elk). I hypothesized that STE model abundance estimates derived with motion-sensing and time-lapsed photos (Moeller et al. 2018; Ausband et al. 2022) along detailed estimates of functional camera viewsheds (Moeller et al. 2023), would result in similar or more precise abundance estimates for both deer and elk of FRMR.

Study Area

My study occurred on FRMR (39.1930°N, 96.8129°W; ~ 40,900 ha; Fig. 3.1;), Kansas, USA. FRMR is located in the flint hills north of the Kansas and east of the Republican rivers in western Geary and Riley counties and has a topographic relief of ~330 ft (USGS, Coiner, R.L., 2010). There are 2 large off-limits areas that are inaccessible to recreationists which include an impact area (~5,800 ha) located in the eastern portion, and range complex (~2,545 ha) located in the northwest portion, where explosive ordinance and small arms fire occur. Another off-limits area is the garrison cantonment area where soldier housing and garrison offices are, located in

the southern portion of FRMR (~3,940 ha). The total study area I surveyed was ~28,383 ha which excluded the 3 off-limits areas. FRMR is mostly dominated by open tallgrass prairie by ~74%, shrubland and woodland vegetation made up ~21%, and the remainder consists of water, food plots, firebreaks, quarries and developed areas.

Methods

Distance sampling for deer

FRMRED and I conducted spotlight distance surveys to quantify the density of deer and classify the age, sex and species structure of the population within FRMR. The survey was conducted along 15 road-based established transects that had a combined distance length of 104.6 km (Rdistance; range = ~2.01 – ~14.5 km) within the 28,383-ha sampling area for deer on 16, 17 and 20 November 2022. Surveys were conducted in accordance with Kansas Department of Wildlife and Parks Deer Distance Sampling 2020 instructions (Survey 123). There were ≥ 2 vehicle traveling at ~19.3 kmh (range = 12.9 – 24.1 kmh) with 2-4 personnel (1 driver/data recorder; 1-2 spotters) equipped with binoculars, compass, GPS, rangefinders, and spotlights scanning a 180-degree arc perpendicular to the transect line. Surveys were conducted 1 hour after sunset and ran for 3-4 hours. When a deer was detected the vehicle would stop, record the GPS location, range the distance and compass angle, and record the age, sex, and activity (Standing; Moving; Laying down; Unknown) of the initial location of the deer.

Aerial census for elk

On 16 January 2023 FRMRED and I conducted a 4-hour aerial complete elk census with two UH-60 Black Hawk helicopters that covered both surveyed area and off-limit areas within FRMR. FRMR was divided into two sections from northwest to southeast and each helicopter

surveyed open ground and engaged in hovering efforts on wooded habitat (draws, ridges, and creek beds) to induce elk to relocate into open habitat, facilitating an accurate population count. Each helicopter was fitted with eight personnel; 2 pilots, ≥ 3 wildlife biologist, and the remaining were volunteer spotters. Wildlife biologist and volunteers were equipped with binoculars recorded their counts and confirmed a consensus prior to moving on to the next terrain feature.

Sampling for STE models

Site Selection and Camera Deployment

I used 150 random points determined by a randomized ‘Fishnet design’ tool in ArcGIS to establish camera sites for elk and deer. All sites were separated by an average Euclidian distance of 989 m (range = 427 – 3586 m) and contained a single unbaited camera (Bushnell Trophy Cam®; Model 119717CW). I adjusted the specific location of each unbaited camera-trap to avoid solar glare and to attach to the closest structure (e.g., tree, post). I attached cameras to structures at 1m height and removed vegetation in front of the camera with a handheld electric trimmer. Cameras were set to trigger using both motion-sensor (three second interval between triggers) and ‘field scan’ time-lapse (one photo every 30-min) functions following Ausband et al. (2022). Cameras were in place from 03 January 2023 – 07 February 2023 and active an average of 33 days (range = 23 - 35 days). Cameras were active ranging from 23 and 35 days and two cameras were picked up before 28 days due to prescribed fires. Because of time-lapse camera malfunctions I removed four sites from my analysis resulting in 146 sites.

Estimating camera viewshed area

Obtaining precise information regarding the area sampled by the camera viewshed is critical for estimating species abundances using STE models (Rowcliffe et al., 2008; Moeller et

al., 2023). Not accounting for obstructions within each camera's viewshed can bias estimates of the total area sampled. I used alternative geometries for measuring a circular sector when camera viewsheds were partially blocked (Moeller et al. 2023; Fig. 3.2). I divided the area in front of the camera into multiple circular sectors, each with angle θ , and distance r or r' (Moeller et al. 2023; Fig. 3.3). I chose the maximum distance sampled (r) as 50 m to align with previous studies (Moeller et al. 2018). To visually mark the range extent in each camera viewshed, I used an orange wooden stake placed at ≤ 50 m in front of the camera. If an elk or deer was detected beyond the orange stake I did not include that detection in the STE model (Moeller et al. 2018). If terrain within the viewshed had a sudden drop (i.e. ridge or hillside) I then placed the orange stake at the maximum cutoff range for that particular camera. The camera manufacturer suggested a 40° viewshed angle, and I confirmed this with multiple photo trials in a controlled setting. I constructed a novel lightweight camera topper to be placed on top of the game trail camera. Six sections were evenly depicted on the topper which broke the landscape of the viewable angle into portions that captured the viewable angle into $\sim 6.67^\circ$ respectively (Moeller et al. 2023; Fig. 3.3). If these portions had an obstruction of >1 m in width (e.g., tree, brush, terrain) I determined the distance to the obstruction with a handheld rangefinder and noted where obstruction occurred within the lines depicted on the camera topper. I then measured each of the six circular sectors and added them to estimate total viewable area. If a camera viewshed was completely unobstructed, the viewable area was 872.66 m^2 . I documented the viewshed of the camera on a modified U.S. Army range card (DA Form 5517-R; Fig. 3.4). The camera topper was only used during camera installation, it did not remain on the camera throughout the study period.

Assessing Presence/Absence

Once cameras were retrieved, I transferred photo data to Wildlife Insights (www.wildlifeinsights.org) to assess detection/non-detection of elk or deer at each site. For the STE model data set I chose to use picture data beginning on 06 January 2023 14:30:00 as this was when all 150 cameras were functional, and ending at 06 February 2023 07:30:01 (range = 23-35 days) because this was 20-minutes prior to the first camera being picked up at the end of my study period. I categorized the obscured viewshed area of individual photos affected by inclement weather (fog, frost, snow, etc.) into proportions of obscurity (0.50, 0.75, 1.00) to characterize a more accurate functional viewshed area for the STE models (spaceNtime package v1.3-0). A benefit of using 30-minute time-lapse intervals as well as motion-sensor enabled picture settings is that you are able to utilize time-lapse photos while also being able to account for viewshed visibility from fog, snow or other obstructions. A posthoc analysis of all photos indicated that ≤ 0.25 obscurity from adverse weather did not hinder my ability to detect elk or deer.

STE Modeling

Originally, the STE method was designed for cameras to sample using time-lapse photos in synchronicity. The cameras were not equipped with this feature and once cameras were turned on, the time-lapse feature began. I used combined motion-sensor and time-lapse functions (similar to Ausband et al. 2022) and sampled data with frequency of every 30-minutes (Rstudio v.4.2.1; spaceNtime package v.1.1.2). This method differs from previous studies where using time-lapse with motion would sample photos in a much shorter sample frequency and length (Ausband et al. 2022). In essence, I was sampling throughout the study as I was also using motion-sensing detections for both species. Like with previous motion-sensed and time-lapse STE research I assumed that if a photo was not taken while motion-sensing capabilities were

enabled, it was assumed that there was not an animal present in the camera's viewshed (McMurry et al. 2022).

My photo data for both elk and deer were established as 'cam', 'datetime' (YYYY/M/D hh:mm:ss), and 'count'. My camera deployment data was used for both elk and deer and was 'cam', 'start' (YYYY/M/D hh:mm:ss), 'end'(YYYY/M/D hh:mm:ss), and 'area' (m²). To build the encounter history I established the sampling frequency as 1,800-seconds (30 minutes) because I used the 30-minute time-lapse feature and the sampling length as 1,800-seconds because I used the motion-sensor photo capabilities. Since motion-sensor capabilities were used I chose to use the sampling length as 1,800 seconds stating that I was constantly sampling and any non-detection was treated as a true zero. For instance, I began the sampling frequency time from 2023-01-06 14:30:00 and ended at 2023-01-06 15:00:00, then again beginning at 2023-01-06 15:00:00 for 1,800 seconds and so on until the end of the condensed study period, sampling for photos throughout the time.

To assess for potential bias associated with the chosen sampling timeframe, I bootstrapped abundance estimates derived from different sampling intervals. I modeled an abundance estimate with a sampling frequency of 1,800-seconds with a sampling length of 1-second and running the sampling occasions in 30-second intervals beginning from the condensed study period. From this I collected each of the abundance estimates, and 95% confidence intervals to determine if the sampling length was biased for oversampling (Fig. 3.7 & 3.8; Table 3.2). I was unable to attain some of the time sequence abundance estimates due to no photos falling within the sampling occasion during that encounter history. I then ran the same simulated bootstrap analysis of the original STE models condensed data set and used a sampling frequency of 1,800-seconds with a sampling length of 1,800-seconds and ran the sampling occasions in 30-

second intervals beginning from the condensed study period. for a total of 59 iterations. I chose 59 iterations after consulting with the STE “spaceNtime” model developers who suggested at least 50 iterations to be completed. From this I collected each of the abundance estimates and lower and upper confidence intervals to determine if the sampling length was biased for oversampling (Fig. 3.5 & 3.6; Table 3.2).

Results

From distance sampling, I estimated 1,527 (95% CI = 1,079 – 2,161; Fig 3.10) deer on FRMR suggesting there were 0.05 deer/ha (95% CI = 0.04 – 0.08). Aerial elk census survey results were 102 total elk (Bulls: 53, Spikes: 5, Cows: 43, Calves: 1; Fig 3.9) in 14 separate clusters. I also observed a large cluster of elk (~75-100) on a property adjacent to FRMR but did not include in the aerial census.

I documented 1,173 photos of elk (49 time-lapse photos and 1,124 motion-sensor photos; Table 3.1) out of a total 316,823 photos in 4,854 total trap nights. I documented 12,350 photos of deer (212 time-lapse photos and 12,138 motion-sensor photos; Table 3.1) out of a total 316,823 photos in 4,854 total trap nights. My STE model estimated 198 elk on FRMR (SE = 19; 95% CI = 164 – 239; Table 3.2). My STE model estimated 2,290 deer on FRMR (SE = 79; 95% CI = 2,141 – 2,450; Table 3.2).

Elk mean abundance estimates from my 1-second sampling length simulated bootstrap analysis (59 iterations) was 4 (95% CI = 2 – 13; Fig. 3.7; Table 3.2). Deer average abundance estimates from my 1-second sampling length simulated bootstrap analysis (59 iterations) was 29 (95% CI = 18 – 46; Fig. 3.8; Table 3.2). Elk average abundance estimates from my 1,800-second sampling length simulated bootstrap analysis (59 iterations) was 193 (95% CI = 159 – 233; Fig. 3.5; Table 3.2). Deer average abundance estimates from my 1,800-second sampling length

simulated bootstrap analysis (59 iterations) was 2,259 (95% CI = 2,110 – 2,418; Fig. 3.6; Table 3.2).

Discussion

STE models may be a good alternative to traditional methods of estimating abundances of elk and deer at large spatial scales. My results demonstrate that using motion-sensing and asynchronous time-lapse photos may be effective over large landscapes with moderate camera densities. Additionally, my STE estimates provided more precise estimates for deer and elk compared to distance surveys (as inferred from 95% CIs). Spotlight surveys were limited to designated roads as transects which may have contributed to reduced precision in abundance estimates because of unaccounted detection issues or biased sampling routes. My abundance estimates for elk were almost double total aerial census counts which did not include a measure for variability. It should be noted that the protocol for FRMR observers was not to count individual elk that were detected outside the physical boundary of FRMR. However, during the aerial census I observed ~100 elk in an agriculture field directly adjacent to FRMR (Byrd personnel observation). This herd likely originated from FRMR and functionally contributes to the reestablished elk herd centered on FRMR. The camera sampling likely reflected this herd's contribution to overall population abundances on FRMR.

Although my results are compelling, there were some limitations to the sampling and analyses. My cameras were taking asynchronous photos during time-lapse settings as my cameras did not have the ability to synchronously collect time-lapse information. I attempted to correct for this by implementing motion-triggered as well as sampling during a 24-hour period instead of crepuscular times (as suggested by Moeller et al. 2018). Registration area of motion-triggered photos would have a reduced viewshed area as triggering occurs at ~14m. However,

developing a viewshed deployment history of each cameras time-lapse and motion-triggered setting would be too labor intensive and these efforts would likely outweigh benefits of increased model precision. Additionally, I could not compare precision of my STE model estimates to aerial census counts as FRMR survey methodology was not developed to estimate variability in total counts. However, it was very likely that all elk on FRMR were counted as the landscape (majority open prairie) and timing of sampling (winter with no leaf coverage in forested draws) would likely result in very high elk detection estimates. Future aerial count efforts would likely benefit from developing survey methods that will also allow researchers to derive confidence intervals for abundance estimates while also accounting for factors that may affect detection of individual elk.

My efforts focused on estimating viewshed area by using a camera topper likely decreased measurement bias by accurately accounting for stationary (brush, ridges, cliffs) and intermittent weather obstructions (fog, frost on lens) observable area within the 30-minute timeframes. My approach at estimating the functional viewshed of individual cameras may have provided a more reliable estimate of the total area sampled and could be incorporated in future studies using STE models. Perhaps there is a threshold for STE models that when a species is in low abundance I need to use both settings. During a post hoc sensitivity analysis I assessed a STE model sampling data at 15-minute sampling occasions and frequency to assess potential oversampling biases which resulted in deer estimates of 1,366 (95% CI = 1,290 – 1,446) and for elk 105 (95% CI = 8 – 126). This could suggest using a 30-minute sampling occasion and frequency for asynchronous cameras for species occurring in relatively low densities is permissible and for high densities it is more accurate to use 15-minutes in order to avoid oversampling. However, more detailed research regarding these details are necessary to identify

sampling frequencies necessary to achieve greatest precision and accuracy of abundance estimates.

Since my survey extent was twice as long as other studies this could have contributed to greater confidence in my estimates. Previous studies used motion-triggered photos and time-lapse photos only during daylight hours where I used time-lapse photos for 24-hours (Moeller et al. 2018). Elk on FRMR are likely occurring at relatively low densities compared to subsequent years and recent work has recommended using both motion-sensor photos and time-lapsed photos to ensure cameras were operating properly (Ausband et al. 2022). My *a priori* analysis using only data collected from time-lapsed elk photos ($n = 49$) resulted in unrealistically low abundance estimates (< 10 individuals) supporting my efforts to include both time-lapsed and motion-triggered photo data in my analyses. Previous studies used STE estimates and have successfully compared them to abundance estimates generated from genetic data (Ausband et al. 2022; McMurphy et al. 2023) and aerial surveys (Moeller et al. 2018). My research supports my hypothesis that STE approaches can concurrently provide precise estimates of abundance for multiple species and may be a cheaper alternative to single-species population surveys. Another benefit to the STE sampling approach is that researchers can model other state variables such as occupancy and or intensity of habitat use from other species also captured in my photo data. This added benefit would likely be difficult to achieve using traditional distance sampling or aerial census approaches.

It remains unclear if STE approaches are sensitive to seasonal variation associated with biological traits (e.g., dispersal, kit rearing, nesting) and this should be assessed in future studies.

Future studies should test these methods across seasons to determine if estimates are consistent and precise. Additionally, STE approaches would benefit from identifying thresholds

for camera densities or sampling durations. My results, however, provide precise abundance estimates that managers can replicate annually to better understand how habitat management efforts or harvest quotas affect changes in population sizes over time. It is likely that this approach may provide a cheaper, and more precise way to concurrently estimate population abundances of deer and elk during a short sampling time frame. Additionally, because of this sampling design, researchers can also address other important questions related to wildlife management on FRMR including distribution and abundances of mesopredators and how distributions of other harvested species (e.g., wild turkey [*Meleagris* spp.]) are affected by management activities in the region.

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Tables

Table 3.1.

Total photos collected by cameras (03 January – 07 February 2023) and condensed photo data (06 January – 06 February 2023) that I used to run the STE analysis for elk (*Cervus canadensis*) and white-tailed deer (*Odocoileus virginianus*) within Fort Riley Military Reservation, KS, USA. Time-lapsed = Photos taken every 30 minutes; Motion = Photos captured due to motion-sensor features; Total = Total photos of time-lapsed and motion combined.

<i>Species</i>	<i>4,854 trap nights</i>			<i>4,650 trap nights (condensed)</i>		
	Time-lapsed	Motion	Total	Time-lapsed	Motion	Total
Elk	49	1,124	1,173	48	900	948
Deer	212	12,138	12,350	198	10,859	11,057

Table 3.2.

Table of the Space-to-event model results from the condensed study period. Bootstrap results for both sample lengths of 1,800-seconds and 1-second. SE was not recorded for bootstrap iterations because of NA values due to lack of photos occurring within encounter histories for elk. Area = Fort Riley Military Reservation surveyed area in meters squared; LCI = Lower Confidence Interval; UCI = Upper Confidence Interval.

<i>Species</i>	<i>Abundance / Area (m²)</i>	<i>Standard Error</i>	<i>LCI</i>	<i>UCI</i>
<i>STE Model</i>				
Elk	197.8 / 283,830,000	19.1	163.7	238.9
Deer	2290.0 / 283,830,000	78.7	2140.9	2449.5
<i>Bootstrap 1,800-secs</i>				
Elk	192.9 / 283,830,000	-	159.3	233.6
Deer	2258.9 / 283,830,000	-	2110.7	2417.5
<i>Bootstrap 1-second</i>				
Elk	4.4 / 283,830,000	-	1.75	13.1
Deer	28.4 / 283,830,000	-	17.9	45.7

Table 3.3.

Table of the viewable angle that contains the left and right limits for both compasses used to measure azimuths (in degrees) for the pilot and football field test. Each left and right limit azimuth was taken three times by each compass. The angle was calculated by subtracting the left limit from the right limit. The average is the result of the three angles of each compass respectively. The average total is the average of both compasses combined for that distance.

	<i>Left Limit Reading</i>	<i>Right Limit Reading</i>	<i>Coleman compass</i>	<i>Left Limit Reading</i>	<i>Right Limit Reading</i>	<i>K & E compass</i>	<i>Average total</i>
<i>Pilot</i>							
10 meters	179.0	210.0	31.0	170.0	201.0	31.0	-
	178.0	210.0	32.0	172.0	203.0	31.0	-
	178.0	209.0	31.0	171.0	203.0	32.0	-
Average	-	-	31.3	-	-	31.3	31.3
20 meters	179.0	210.0	31.0	170.0	205.0	35.0	-
	178.0	210.0	32.0	175.0	205.0	30.0	-
	178.0	210.0	32.0	173.0	205.0	32.0	-
Average	-	-	31.7	-	-	32.3	32.0
25 meters	170.0	203.0	33.0	175.0	200.0	25.0	-
	178.0	207.0	29.0	170.0	212.0	42.0	-
	175.0	207.0	32.0	172.0	204.0	32.0	-
Average	-	-	31.3	-	-	33.0	32.2
<i>Football Field</i>							
Warm-up @ 50 yards	164.0	194.0	30.0	174.0	207.0	33.0	-
	168.0	196.0	28.0	173.0	207.0	34.0	-
	164.0	196.0	32.0	173.0	208.0	35.0	-
Average	-	-	34.0	-	-	32.0	33.0
25 yards	164.0	198.0	34.0	165.0	200.0	35.0	-
	169.0	200.0	31.0	167.0	200.0	33.0	-
	169.0	198.0	29.0	168.0	200.0	32.0	-
Average	-	-	31.3	-	-	33.3	32.3

30 yards	170.0	198.0	28.0	170.0	205.0	35.0	-
	166.0	196.0	30.0	172.0	204.0	32.0	-
	164.0	201.0	40.0	168.0	206.0	38.0	-
Average	-	-	32.7	-	-	35.0	33.9
35 yards	162.0	198.0	36.0	172.0	208.0	36.0	-
	164.0	200.0	36.0	172.0	202.0	30.0	-
	164.0	198.0	34.0	172.0	206.0	34.0	-
Average	-	-	35.3	-	-	33.3	34.3
40 yards	168.0	208.0	40.0	170.0	205.0	35.0	-
	170.0	206.0	36.0	169.0	204.0	35.0	-
	170.0	204.0	34.0	170.0	203.0	33.0	-
Average	-	-	36.7	-	-	34.3	35.5
45 yards	165.0	200.0	35.0	176.0	214.0	38.0	-
	178.0	208.0	30.0	172.0	214.0	42.0	-
	162.0	202.0	40.0	174.0	212.0	38.0	-
Average	-	-	35.0	-	-	39.3	37.2
50 yards	165.0	197.0	32.0	173.0	216.0	43.0	-
	160.0	196.0	36.0	173.0	215.0	42.0	-
	160.0	196.0	36.0	173.0	213.0	40.0	-
Average	-	-	34.7	-	-	41.7	38.2

Figures

Figure 3.1.

Study area of Fort Riley Military Reservation (FRMR) within the Flint Hills ecoregion (outlined in bolded black polygon) of Kansas, USA. I placed a single unbaited camera trap at each study site ($n = 146$) at FRMR Environmental Division's random survey locations and estimated elk (*Cervus canadensis*) and white-tailed deer (*Odocoileus virginianus*) from 03 January – 07 February, 2023.

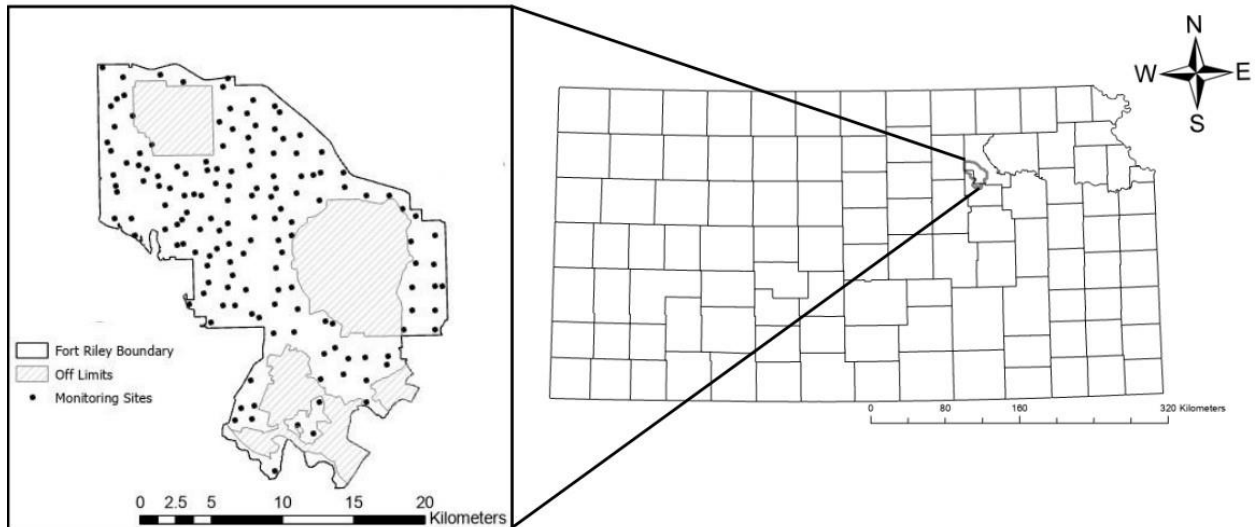


Figure 3.2.

Formula used to determine the camera's viewable area, a , when the camera is placed parallel to the ground at ~1-meter height. r is the viewshed radius (max range where orange stake was placed), and θ was the viewshed angle in degrees which for the study was 40° (Moeller et al. 2023).

$$a = \pi r^2 \left(\frac{\theta}{360} \right)$$

Figure 3.3.

Measurement of each viewshed section when an obstruction is present. The viewshed is divided into 6 sections that coincide with the camera topper. r indicates the max range for that site whereas r' indicated range to obstruction. When there is no obstruction at the site then $\theta^1 = 40^\circ$, max range is 50 meters unless terrain (woodland, ridge or hill) dictates otherwise. When calculating each section area then each section's respective would be $\theta^2 = 6.67^\circ$.

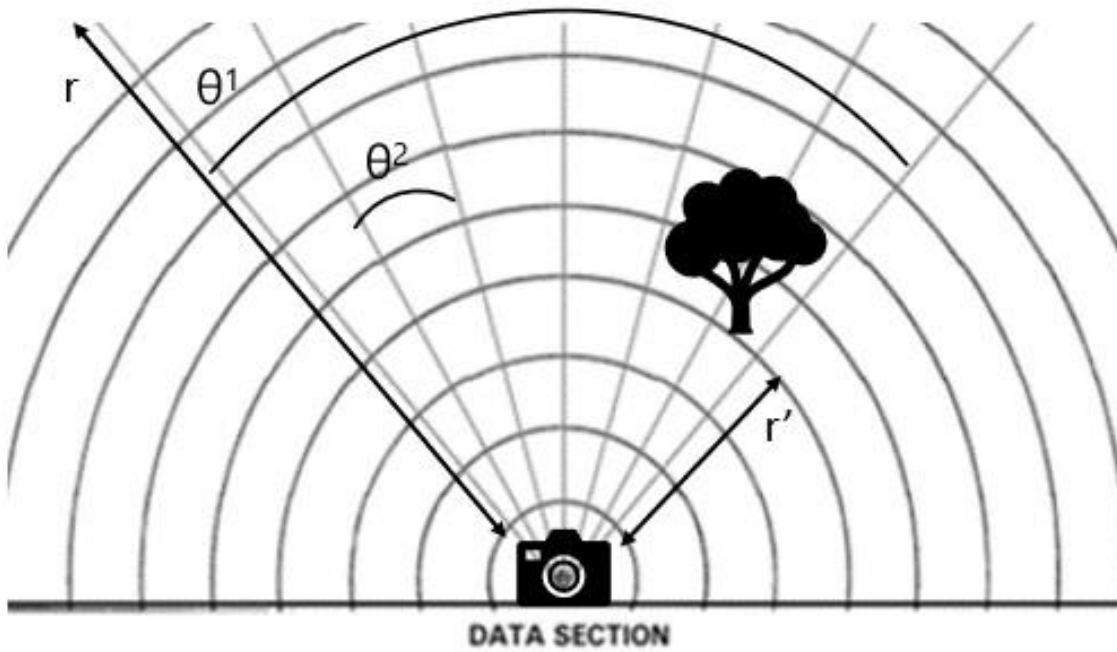


Figure 3.4.

Modified U.S. Army range card DA Form 5517-R, if an obstruction was present I measured distance, r , with a rangefinder by standing directly in front of camera lens. Each circle equated an equal distance determined by person placing camera, if max range for the camera at that site was 50 m then each circle was 5 meters.

PERSON _____		COMPASS DIRECTION OF CAMERA (IN DEGREES) _____		MAGNETIC NORTH	
TIME (24HR) CAMERA TURN ON, TAKEN FROM CAMERA ITSELF _____					
DATA SECTION					
GPS UTM 14 DIGIT GRID _____				DATE (DDMMYY) _____	
CAMERA ID # _____			EACH CIRCLE EQUALS METERS _____		
NO.	COMPASS AZIMUTH OF OBJECT	ELEVATION	RANGE (m)	WIDTH	DESCRIPTION
1					
2					
3					
4					
5					
6					
REMARKS:					

Figure 3.5.

Results from the simulated bootstrap analysis (59 iterations) for elk by running an abundance estimate with a sampling frequency of 1,800 seconds with a sampling length of 1,800 seconds and running the sampling occasions in 30-second intervals beginning from the study period (14:30:00, 06 January 2023 – 07:30:01, 06 February 2023), in Fort Riley Military Reservation. The average abundance estimate for elk was 193 (95% CI = 159 – 233).

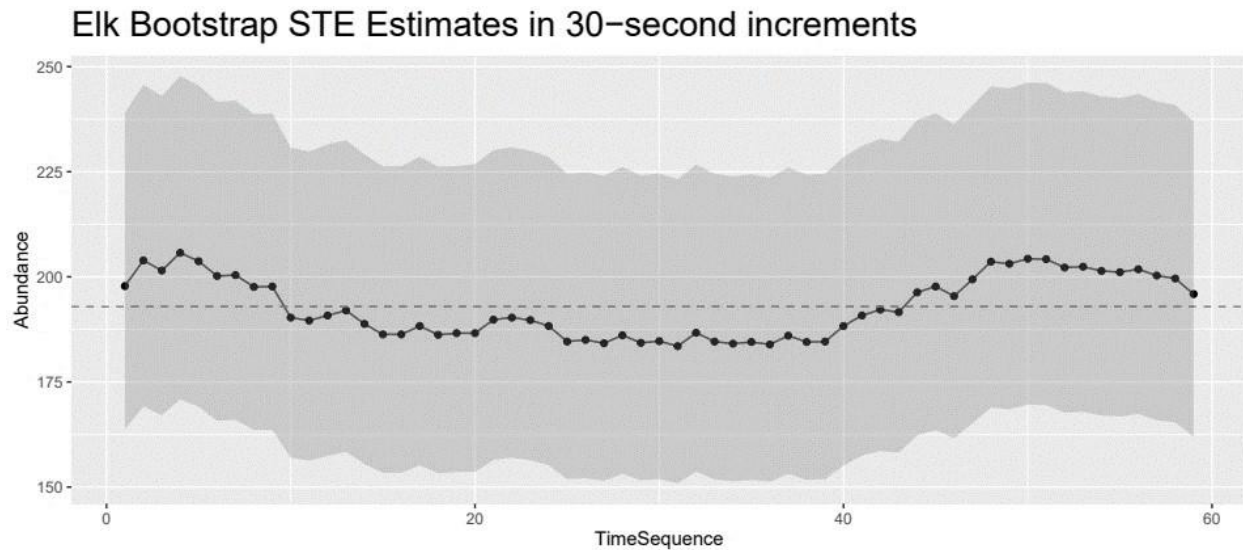


Figure 3.6.

Results from the simulated bootstrap analysis (59 iterations) for deer by running an abundance estimate with a sampling frequency of 1,800 seconds with a sampling length of 1,800 seconds and running the sampling occasions in 30-second intervals beginning from the study period (14:30:00, 06 January 2023 – 07:30:01, 06 February 2023), in Fort Riley Military Reservation. The average abundance estimate for deer was 2,259 (95% CI = 2,110 – 2,418).

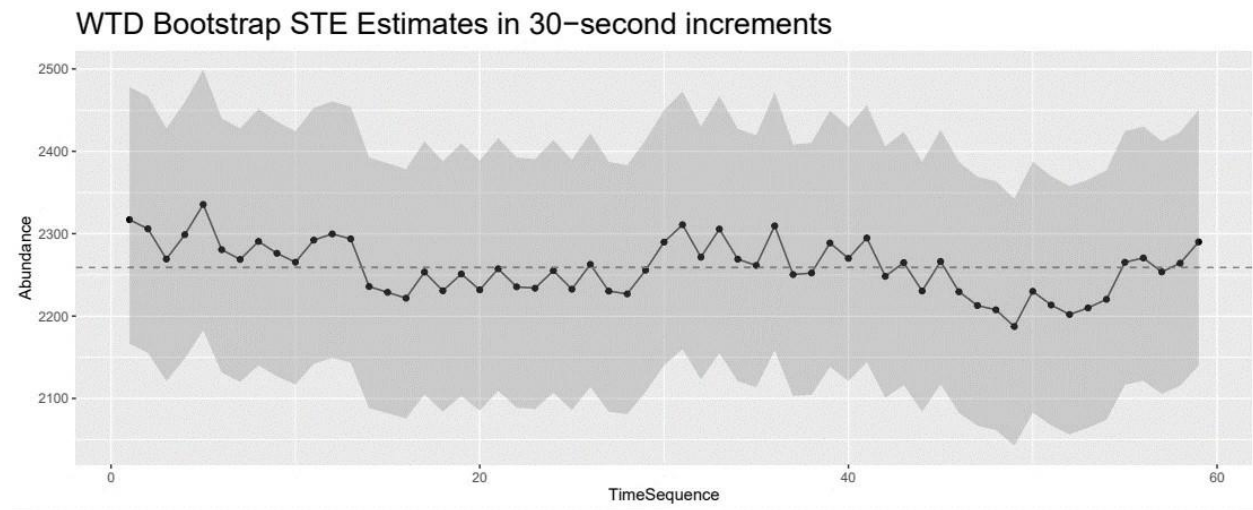


Figure 3.7.

Results from the simulated bootstrap analysis (59 iterations) for elk by running an abundance estimate with a sampling frequency of 1,800 seconds with a sampling length of 1 second and running the sampling occasions in 30-second intervals beginning from the study period, (14:30:00, 06 January 2023 – 07:30:01, 06 February 2023), in Fort Riley Military Reservation. The average abundance estimate for elk was 4 (95% CI = 2 – 13). Some of the time sequence abundance estimates were NA due to no photos falling within the sampling occasion during that encounter history.

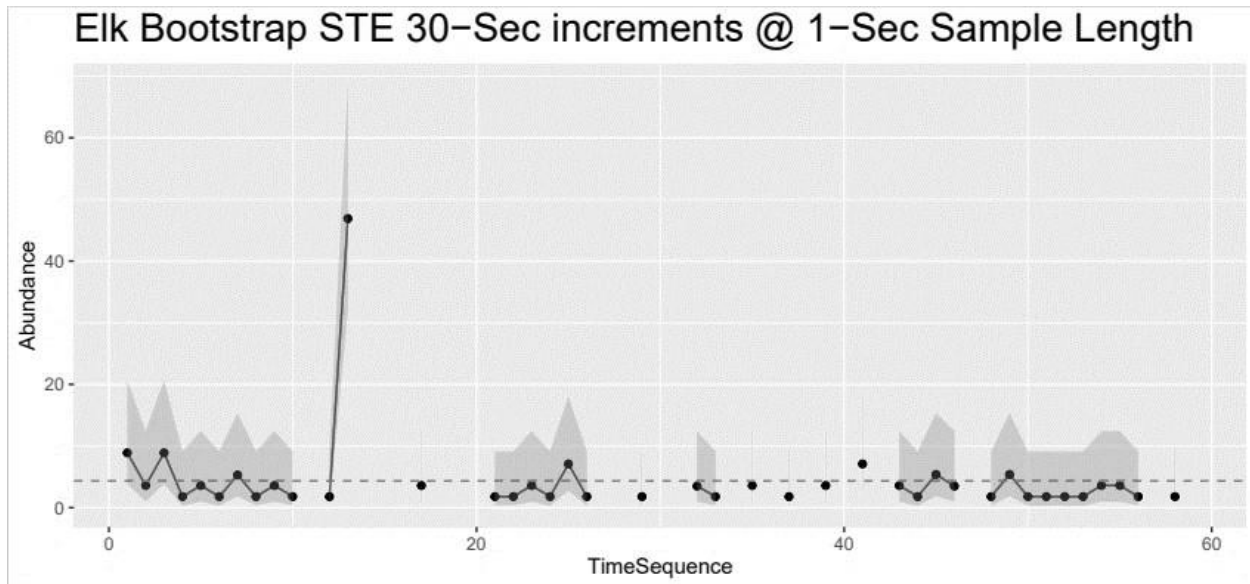


Figure 3.8.

Results from the simulated bootstrap analysis (59 iterations) for deer by running an abundance estimate with a sampling frequency of 1,800 seconds with a sampling length of 1 second and running the sampling occasions in 30-second intervals beginning from the study period (14:30:00, 06 January 2023 – 07:30:01, 06 February 2023), in Fort Riley Military Reservation. The average abundance estimate for deer was 29 (95% CI = 18 – 46).

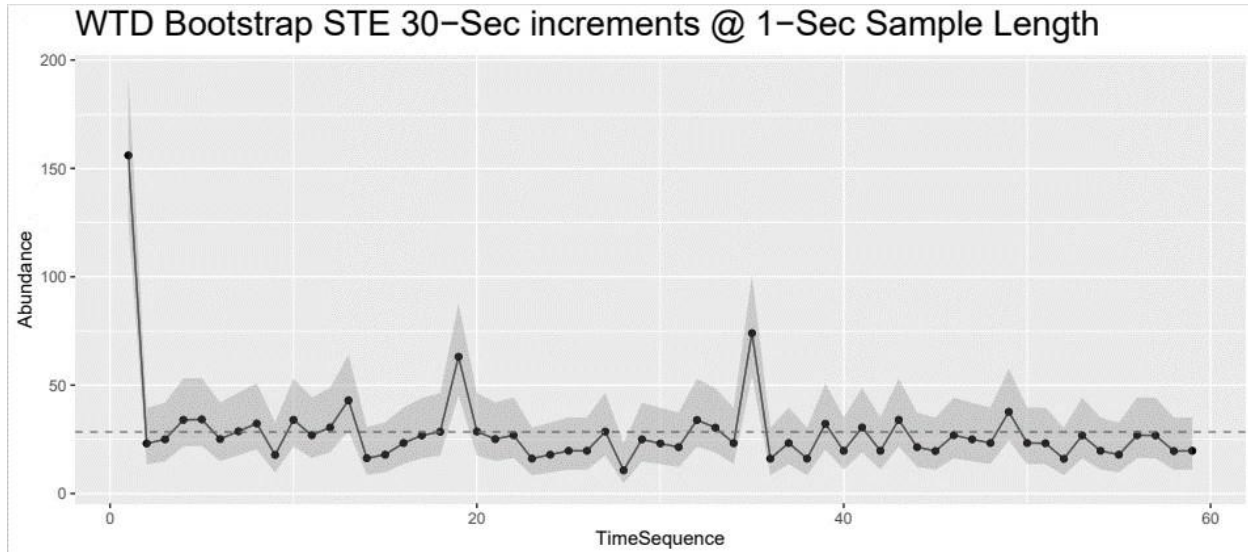


Figure 3.9.

The total 4-hour aerial census for elk directly occurring on Fort Riley Military Reservation (FRMR) conducted on 16 January 2023 was 102 (Bulls: 53, Spikes: 5, Cows: 43, Calves: 1, Separate clusters: 14, no error estimates). However, I observed ~75-100 elk on property directly adjacent to FRMR that were not included in this estimate. The space-to-event abundance estimate for elk was 198 (95% CI = 164 – 239).

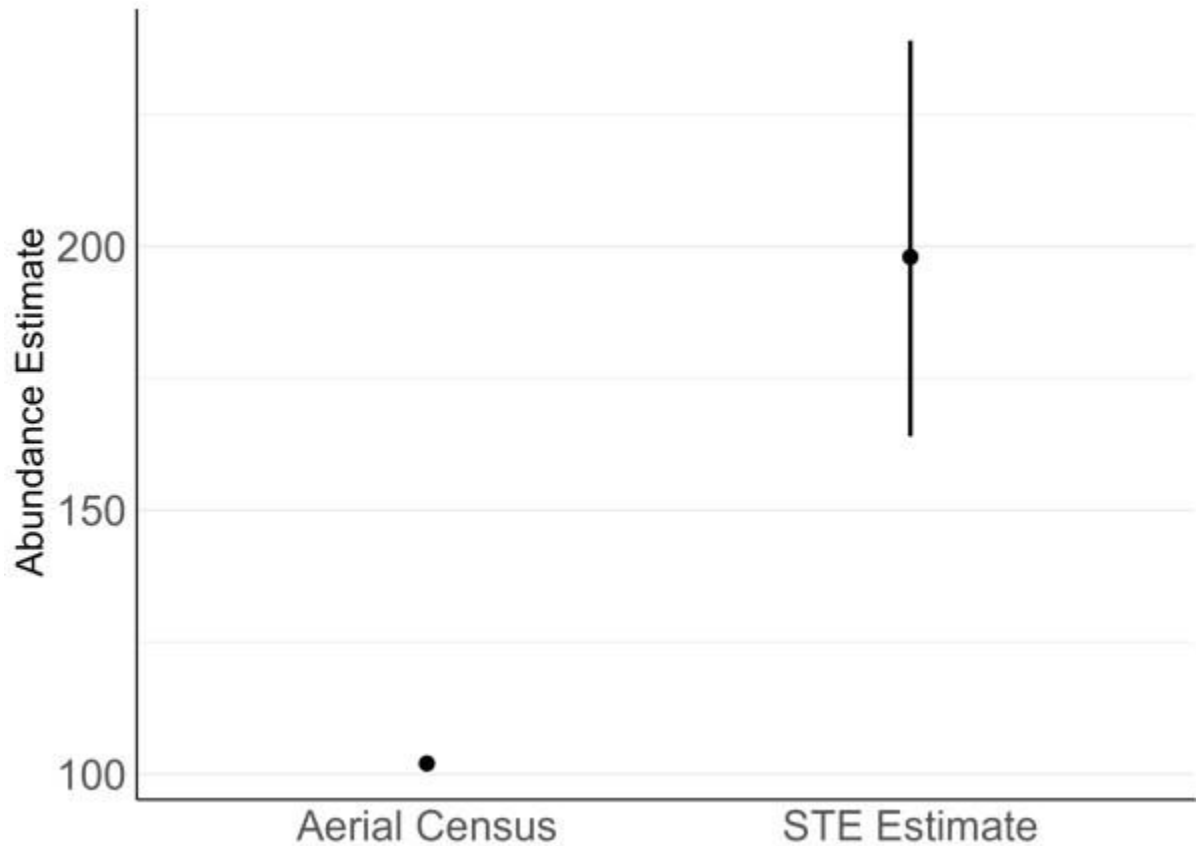
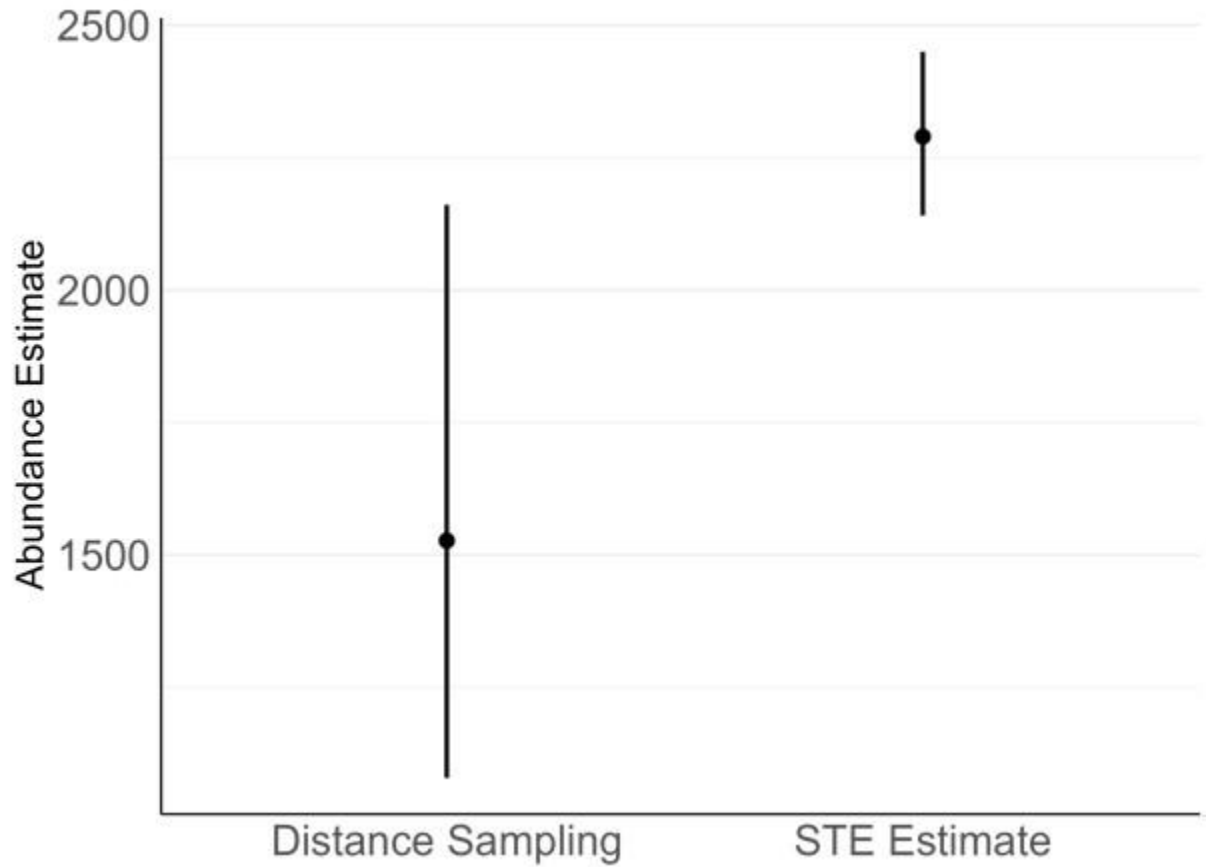


Figure 3.10.

The distance-sampling estimate for deer conducted within Fort Riley Military Reservation (16, 17, & 20 November 2022) was 1,527 (95% CI = 1,079 – 2,161). The STE abundance estimate for deer was 2,290 (95% CI = 2,141 – 2,450).



Chapter 4 - Conclusion

Rocky Mountain elk (*Cervus canadensis*, hereafter ‘elk’) are large ungulates that can be considered a keystone species and manipulate their environment through interactions with vegetation (Augustine and McNaughton 1998; Stewart et al. 2006). Using fire as habitat management can be beneficial to maintain grasslands and provide forage for a restored elk population and deer (Long et al. 2008; Sittler et al. 2019). When deer densities become high or above carrying capacity they can have a negative direct and indirect effects on habitat (Rooney and Waller 2003). As population densities increase, it becomes imperative to accurately estimate abundances. For example, factors like chronic wasting disease spread is increasing in ungulate populations and understanding spatial and population dynamics could be vital for controlling disease spread and harvest regulations (Joly et al. 2006; Wasserberg et al. 2009). Testing non-invasive novel abundance methods may be more logistically feasible for wildlife biologist than traditional methods and can also collect data on multiple species that could be used in different state variable models (Sollman 2018).

I used 36 days of presence-absence data from 149 camera traps and single-season occupancy models to evaluate the occupancy dynamics of Rocky Mountain elk in the tallgrass prairie in northeast Kansas. Using landscape cover data, I was able to look for patterns of habitat use while trying to control for detection. I had a naïve occupancy of elk of 0.26 at the 149 sites over the study period. My results suggest that elk have a higher occupancy probability in areas that were most recently burned which coincides with previous studies (Long et al. 2008; Sittler et al. 2019). I used 31 days of presence-absence data from 146 camera traps and novel Space-to-Event models to estimate abundance for elk and white-tailed deer (*Odocoileus virginianus*, hereafter ‘deer’) within the same study area. I had also developed a tool that could be useful for

biologist to estimate a cameras viewshed when implementing camera traps while using Space-to-Event models. My results that I derived from an estimated abundance for elk and deer were slightly higher than most recent surveys, my precision was higher.

My study provided evidence winter habitat use by elk in tallgrass prairies is influenced by temporal variation in burn timing, suggesting elk will use burned areas with fresh growth as a high-quality food resource. Elk could be using these areas to replenish body composition losses due to a multitude of factors such as loss of fat reserves after vigorous activity during the breeding season or decreased milk productivity in lactating females during wintering months (Hudson et al. 2002; Cook et al. 2013). Looking at my results along with supporting literature, wildlife managers could optimize habitat for elk by also applying rotational prescribed fires in their habitat management regiment. I also uncovered the potential for alternative methods of using Space-to-Event models to estimate abundance for multiple unmarked species. Although my methods were not traditional with previous studies, future research could focus on repeating this research in varying seasons to determine if consistent abundance estimates are achieved.

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