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Distribution and Population Dynamics of Ungulates in the Mongolian Gobi

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**DISTRIBUTION AND POPULATION DYNAMICS OF UNGULATES IN THE
MONGOLIAN GOBI**

A Dissertation Presented

by

BUUVEIBAATAR BAYARBAATAR

Submitted to the Graduate School of the
University of Massachusetts Amherst in partial fulfillment of
the requirements of the degree of

DOCTOR OF PHILOSOPHY

SEPTEMBER 2016

WILDLIFE AND FISHERIES CONSERVATION

**DISTRIBUTION AND POPULATION DYNAMICS OF UNGULATES IN THE
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A Dissertation Presented

By

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DEDICATION

To the memory of my father, Bayarbaatar Galdanbemberel.

To my beloved family, wife Mandukhai Batbaatar, a daughter Uyanga,
for their love and endless support to make it possible.

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ABSTRACT

DISTRIBUTION AND POPULATION DYNAMICS OF UNGULATES IN THE MONGOLIAN GOBI

SEPTEMBER 2016

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The Mongolian Gobi desert ecosystem is one of the most spectacular and important regions in Central Asia, which supports a large assemblage of migratory plains ungulates. In recent years, a growing human population, expanding exploitation of natural resources, and the development of infrastructure in the region place increasing pressure on these species and their habitats. However, the current status and ecology of many of these species remains unknown or data deficient, requiring an urgent need for research to assess how they are responding to an increasing human footprint on the landscape. The overarching goal of my research is to increase understandings of Mongolian saiga (*Saiga tatarica mongolica*), Asiatic wild ass (or khulan, *Equus hemionus*), and goitered gazelles (*Gazelle subgutturosa*) inhabiting Mongolian Gobi desert, whose populations are threatened by the recent and rapid environmental and economic changes occurring across their ranges.

Habitat selection for calving by ungulates is an important behavioral trait as it affects neonate survival. To identify factors that influence birth location selection of Mongolian saiga, I used four years of data collected in and around Sharga Nature Reserve in western Mongolia. I found multiple factors explain calving location selection by saiga antelopes and individual saiga females preferred calving locations that were away from settlements and closer to water sources and avoided steeper slopes. These results demonstrate that the choice of calving locations and grouping patterns of saiga antelope is driven by both internal and external factors.

To identify group size and composition, saiga groups were observed monthly while conducting monitoring surveys, and weekly while tracking radio-collared animals during 2009–2012. Saiga group sizes were highly variable throughout year and seasonality exerted strong effects with the smallest groups in June and largest in December. According to Generalized Linear Model, the changes in group size of saiga influenced by Normalized Difference Vegetation Index, predation rate and season. Understanding which factors influence calving location selection and variation in group size for saiga provides insights to the management of this endangered antelope.

Extensive distance sampling line transect surveys were implemented during 2012–2015 across a much of the Southern Gobi, Mongolia. The primary objective of this survey was to provide data on the density and abundance of khulan and goitered gazelle. The density and abundance of the two species was estimated using the distance sampling analysis. The distance sampling results suggest that Mongolia's Gobi desert holds the largest population of khulan and goitered gazelle in the world. These findings provide a crucial update on the

status of the plains ungulates. It's also suggested that the distance sampling line transects can reliably applied to other plains ungulates inhabiting Mongolian Gobi desert.

Ground-based surveys conducted in the Southern Gobi for four years to provide an assessment of factors affecting the distribution of khulan and goitered gazelles. The Generalized Linear Mixed Model (GLMM) was used to provide information on the key drivers of their distribution. The GLMMs suggest human-associated factors were more important than environmental variables in explaining the seasonal distribution of the two species. In addition, the spatial models of both species indicate about a half of the study area is suitable habitat for khulan and goitered gazelles. In absence of appropriate mitigation measures, the growing development and associated increase in the human footprint in the region will inevitably result in diminished the range available for these species. These findings can be used to plan mitigation measures and reduce the impacts of human influences.

The most important conclusion is that the choice of calving locations and grouping patterns of saiga antelope is driven by both internal and external factors. Understanding which factors influence calving location selection and variation in group size for saiga provides insights to the management of this endangered antelope. Lastly, the Mongolian Gobi accommodates the world's largest populations of khulan and goitered gazelles. However, disturbance associated human activities negatively influence the distribution of the two species, resulting about a half of the study area is non-suitable habitat for khulan and goitered gazelles. In addition, much of the seasonal range of the two species fall outside of protected areas and thus there is urgent need to expand the existing reserves to account for their nomadic movement patterns across a wide landscape.

PREFACE

This dissertation is comprised of five chapters, the first four of which have been published in or submitted to peer reviewed scientific journals, as indicated below.

In Chapter 1, I examine impacts of environmental and human associated on calving site selection behavior of Mongolian saiga in western Mongolia (Buuveibaatar, B., T.K. Fuller, A.E. Fine, B. Chimeddorj, J.K. Young, and J. Berger. 2013. Changes in grouping patterns of saiga in relation to intrinsic and environmental factors in Mongolia. *Journal of Zoology*. 291:51–58).

Chapter 2 reports factors that affect group sizes of saiga antelope in western Mongolia (Buuveibaatar, B., T.K. Fuller, J.K. Young, and J. Berger. 2014. Calving location selection patterns of saiga antelope in Mongolia. *Journal of Zoology*. 294: 241–247).

In Chapter 3, I present results of extensive distance sampling line transect surveys implemented during 2012-2015 across a much of the Southern Gobi, Mongolia (Buuveibaatar, B., S. Strindberg, P. Kaczensky, J. Payne, B. Chimeddorj, G. Naranbaatar, S. Amarsaikhan, B. Dashnyam, T. Munkhzul, T. Purevsuren, D.A. Hosack, T.K. Fuller. In press. Mongolian Gobi supports the world's largest populations of khulan and goitered gazelles. *Oryx*).

Chapter 4 provides the results from the ground-based surveys conducted in the Southern Gobi for four years to provide an assessment of factors affecting the distribution of khulan and goitered gazelles (Buuveibaatar, B., T. Mueller, S. Strindberg, P. Leimgruber, P. Kaczensky, T.K. Fuller. In revision. Disturbance associated with human activities have a negative influence in predicting suitable habitat for ungulates in the Mongolian Gobi. *Biological Conservation*).

In Chapter 5, I summarize the key findings of all chapters and synthesize them into common conclusions.

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CHAPTER 1

CALVING LOCATION SELECTION PATTERNS OF SAIGA ANTELOPE IN MONGOLIA

Abstract

Habitat selection for calving by ungulates is an important behavioral trait because it affects neonate survival. Generally, ungulate calving site selection varies by vulnerability to predators, local topography, habitat quality, and level of human disturbance. The Mongolian saiga (*Saiga tatarica mongolica*) is endemic to Mongolia where a threatened population of ~7,000 exists in the northern Gobi Desert. We analyzed factors that could affect selection of saiga calving locations in the Sharga Nature Reserve, western Mongolia, using data obtained from ground surveys over 4 years between 2008 and 2012. Multiple factors explain calving location selection by saiga antelopes, based on the results of a Generalized Linear Mixed Model within a use–availability framework. Individual saiga females preferred calving locations that were away from settlements, closer to water sources, and avoided steeper slopes in comparison to random locations. These results demonstrate that the choice of calving locations for saiga antelope is driven by both internal and external factors. Understanding which factors influence calving location selection for saiga provides insights to protect important habitats.

Introduction

Variation in recruitment rate affects population trajectories of large herbivores (Gaillard et al., 2000; Coulson et al., 2005). Juvenile survival of large herbivores is generally low and more variable relative to that of adults (Gaillard et al., 1998), thus investigating potential causes of recruitment variability is important for effective management of large herbivores, especially if the population size is small. Where predators are present, depredation is the primary cause of neonatal mortality in large ungulates (Linnell et al., 1995). In response to the high risk of predation to neonates (Garrot et al., 1985), ungulates have evolved two main behavioral strategies, hiding or following (Estes, 1974; Lent, 1974; Leuthold, 1977), depending on whether the newborns lie concealed for their first few days or actively follow their mothers (Lent, 1974; Ralls et al., 1986). Following has been viewed as a strategy for avoiding predators in open habitats, while hiding is thought to reduce the predation risk in closed habitats (Lent, 1974; Estes, 1974).

Ungulate females often face challenges of selecting areas for calving that simultaneously provide sufficient forage to meet high energy demands of lactation and reduce exposure to predators (Lima and Dill, 1990; van Moorter et al., 2009). Consequently, selection of calving areas by female ungulates reflects trade-offs between minimizing risk of predation and maximizing nutrition (Berger, 1992; Rachlow and Bowyer, 1998). Birth site selection of ungulates is also shaped by the need to reduce neonatal predation, including concealment cover (Barbknecht et al., 2011; Pinard et al., 2012), density of predators (Bergerud et al., 1984), topographic variables associated with visibility (Bowyer et al., 1999; Mysterud and Ostbye, 1999), and experience in the previous year (Wiseman et al., 2006). In addition to the environmental factors, anthropogenic impacts such as proximity to settlements

and roads can determine birth site selection of ungulate females (Berger, 2007; Singh et al., 2010).

The Mongolia saiga (*Saiga tatarica mongolica*) is endemic to the semi-deserts of western Mongolia (Bannikov, 1954), with an estimated population of 7,000 individuals (Young *et al.*, 2010), and is categorized as an endangered species by IUCN (Mallon, 2008). Like most ungulates, the saiga antelope segregate at the time of parturition and maternal females become solitary in late spring (e.g. end of May) to seek secluded areas for giving birth (Bekenov et al., 1998). Newborns remain hidden for the first few days of life (Bannikov, 1954; Bekenov et al., 1998). The life history of saiga is characterized by early female reproductive maturity, unusually large neonates relative to female body size, and frequent twinning (Kuhl et al., 2007, Buuveibaatar et al., 2013a). Group size of Mongolian saiga is highly variable throughout the year and seasonality exerts strong effects, with the smallest groups forming in June (calving) and largest in December (Buuveibaatar et al., 2013b). The calving period of Mongolian saiga is highly synchronous and most calving occurs over a short period (7-10 days) in early June; calves are highly vulnerable to predators (mainly raptors and foxes) during the first two months of life (Buuveibaatar et al., 2013a).

Factors affecting calving site selection of saiga antelope (*Saiga tatarica*) are well documented in Kazakhstan, where saiga select sites with lower than average vegetation productivity, low inter-annual variation, intermediate distances from water sources, and far from human settlements (Singh et al., 2010). It is unknown if environmental and human variables similarly influence calving locations in the Mongolian subspecies, but such information would be useful for management of this endangered species (Clark et al., 2006).

Expansion of the existing reserves to protect key areas like calving grounds is recognized as a high priority for saiga conservation (Convention on Migratory Species, 2010).

Our goal herein is to identify factors that influence birth location selection of Mongolian saiga and assess the general relationship between spatial patterns of calving locations to habitat characteristics and thus areas in need of protection. We predicted that saiga would avoid higher elevations and steeper slopes during the calving period because these environmental traits increase predation risk to neonates (Bowyer et al., 1998). We also predicted that saiga females would prefer areas with high vegetation productivity to maximize energy gain (Bowyer et al., 1999; Kie, 1999). During the birth period ungulates are highly sensitive to human disturbance as it reduces reproductive success (Phillips and Alldredge, 2000), so we expected calving locations to be located away from human settlements. Lastly, as availability of water is essential for lactating females (Singh et al., 2010), we expected mothers should use areas closer to natural standing waters in this very dry region.

Study area

We studied saiga calving location selection in a 3,000-km² part of the Sharga Nature Reserve (SNR) in western Mongolia (Figure 1.1). The study area included ~30% of the entire Mongolian saiga range and is bounded by the Altay Mountains to the west. Elevations within the study area ranged from 1,300 to 1,900 m. The study area received ~100 mm precipitation annually and average air temperatures in summer and winter were 18°C and –20°C, respectively (Buuveibaatar, 2011). The main human populations in the area are concentrated in soums (villages/towns). The abundance of livestock, particularly goats, in and adjacent to

the SNR steadily increased over 30 years and now are the most dominant herbivore (Berger et al., 2013). The region is characterized by constant fluctuations in precipitation patterns resulting in a constant change in availability of quality forage (von Wehrden et al., 2012). There is a lack of permanent surface water and local herders rely heavily on hand-drawn wells. A few alkaline lakes present near the towns are not potable for livestock and wildlife. Onions (*Allium* spp.), grasses (*Stipa* spp.), and anabasis (*Anabasis brevifolia*) are the most common plants in this region (Buuveibaatar et al., 2012). Some shrubs (*Caragana* spp.) and trees, such as saxaul (*Haloxylon ammodendron*), are sparsely distributed. Common predators in this system are red foxes (*Vulpes vulpes*), corsac foxes (*V. corsac*), and raptors, such as golden eagles (*Aquila chrysaetos*) and cinereous vultures (*Aegypius monachus*; Buuveibaatar et al., 2013a).

Materials and Methods

We conducted extensive searches across the Sharga Nature Reserve to determine spatial distribution of pregnant females prior to calving, as a part of saiga calf survival research during 2008 – 2010, and 2012 (Buuveibaatar et al., 2013a). Observations of pregnant females were made during the early morning and late evening, when saiga were most active. Systematic searches for newborn calves also were conducted when we observed females exhibiting distinct postpartum behavior. We identified calving locations based on direct observation of births, newborn calves, or by presence of common calving location characteristics, including placenta and disturbed or cleared vegetation in a circular pattern. Geographical coordinates (UTM – Universal Transverse Mercator, zone 46N) were recorded

with a Garmin GPS 60CSX unit (with ± 3 m error) at each location where calving was observed.

Defining used and available habitat

A major difficulty in assessing habitat use concerns the definition of available habitat. This becomes problematic, especially when quantifying used relative to theoretically-available areas (Hjermann, 2000). For this study, we operationally defined ‘used habitat’ as the location of each observed calf birth. The number of calving locations recorded for saiga population in the Sharga Nature Reserve in different years during 2008 – 2010 were 27, 31, and 36, respectively (Table 1.1); due to a shorter field season in 2012, we recorded only 16 calving locations. To define available habitat, we created Minimum Convex Polygons (100% MCP) based on all observations of calving locations for each year over the study period to delineate the extent of yearly calving area within which to sample random points (Table 1.1). Random locations were separately sampled from within each of the yearly calving areas (2008 – 30, 2009 – 30, 2010 – 40, 2012 – 20), to fulfill assumptions of the use and availability framework (Manly et al., 2002). There is no established rule to decide the minimum sample sizes for random points (Peng et al., 2002); however, we tried to keep our samples symmetric (e.g. equal number of used and random points).

Spatial landscape features

Spatial landscape feature values for used and random calving locations were calculated using ArcMap 10.2 and Erdas Imagine 2010 (Leica Geosystems GIS & Mapping, LLC, Heerbrugg, Switzerland). We calculated six spatial landscape features for each used

and random location: vegetation productivity, elevation, slope, and distances to nearest surface water, and town. Vegetation productivity was estimated using the Normalized Difference Vegetation Index (NDVI) acquired by the Moderate Resolution Imaging Spectroradiometer (MODIS) on board the TERRA satellite. For each survey period, we obtained a 16-day NDVI composite in 250-m resolution from NASA's Earth Observing System Gateway (<http://reverb.echo.nasa.gov>) and re-projected the data to the UTM (zone 46 N). Elevation values for the locations were extracted from a 30-m resolution Digital Elevation Model (DEM). We used the surface tool in Spatial Analyst toolbox to create a slope raster map from the DEM. Using extraction tool in the Spatial Analyst toolbox we also extracted NDVI, slope, and elevation values for each used and random location. The nearest Euclidean distances to surface water, and towns were calculated for each used and random location point using the proximity tool in the Analysis toolbox in ArcMap 10.2. Spatial distribution of natural standing water was mapped using data collected during the study period and a GIS database of Gobi-Altai province.

Statistical analysis

To examine spatial patterns of saiga calving locations, average nearest neighbor (ANN) function of Spatial Statistics toolbox in ArcMap 10.2 (ESRI, California) was used to calculate Euclidean distance between calving locations for each year. The nearest neighbor index is expressed as the ratio of the observed mean distance to the expected mean distance. The expected distance is the average distance between neighbors in a hypothetical random distribution. If the index (Z value) is < 1 , the pattern exhibits clustering; if the index is > 1 , the trend is toward uniform dispersion. Annual relationships between nearest neighbor

distances of calving locations, density of calving locations, and vegetation productivity (e.g. the mean NDVI value of the calving areas for each year) were tested using linear regression (Montgomery and Pack, 1982).

We used a Generalized Linear Mixed Model (GLMM) with a binomial error distribution to test for differences between saiga calving and the random locations. We quantified the collinearity among the environmental and human associated covariates using the Pearson's rho and did not include within the same model strongly correlated covariates ($\rho \geq 0.6$). Elevation was excluded from the model because it was negatively correlated with distances to town ($\rho = -0.67$) and positively correlated with NDVI ($\rho = 0.59$). Therefore, our final model included four explanatory variables including NDVI, slope, and distance to nearest town (town), and surface water (water). The second order polynomial ($y \sim x + x^2$) was used to test all variables because a similar study has shown saiga females select for intermediate values due to various trade-offs (e.g. Singh *et al.*, 2010). Since second order polynomial models were not significant for all variables, we eliminated them and the model was re-run in the non-polynomial form ($y \sim x$). The GLMM was run with the library "lme4" (Bates et al., 2011) in R statistical software (R Development Core Team, 2008) with year as a random term. We ran all possible model subsets of the 4 variables and ranked them using the Akaike Information Criterion for small sample sizes (AICc). The final set of models were the most parsimonious based on $\Delta AICc < 4$ (Anderson, 2008). Models with a ≤ 2 AICc unit difference (e.g. $\Delta AICc = AICc_i - \min AICc$) were considered equivalent (Burnham and Anderson, 2002). To quantify the influence of each covariate on calving location selection, we used model-averaging techniques to obtain parameter estimates, unconditional standard errors, and the relative support of each variable (Burnham and Anderson, 2002) within the

"MuMIn" library in R (Barton, 2012). In addition, the model AICc weights were calculated to measure the likelihood of a candidate model being the best among the set of fitted models. We used the area under the receiver operating characteristic curve (AUC) to measure the discrimination ability of the final models, with 0.5 showing no discrimination ability and 1.0 showing perfect discrimination ability of a model (Pearce and Ferrier, 2000).

Results

During 2008 – 2010, and 2012, we collected data on calving locations for 28 females that produced twins and 81 females that produced single calves, for a total of 109 calving locations (Figure 1.1). Calving areas ranged from 115 to 343 km² with densities of 0.08 – 0.14 calving locations/km² (Table 1.1). The average nearest neighbor distances between calving locations, pooled for four years, averaged 1,986 ± 351 m and was largest in 2012 and smallest in 2010 (Table 1.1). Calving locations were randomly distributed in 2009 ($Z = 0.29$, $P = 0.77$) and 2010 ($Z = 0.31$, $P = 0.97$), and highly dispersed patterns were observed in 2008 ($Z = 5.18$, $P < 0.001$) and 2012 ($Z = 4.08$, $P < 0.001$). Overall (e.g. cumulatively across all years), spatial distribution of calving locations were clustered ($Z = -3.08$, $P < 0.001$). There was no relationship between NDVI and the ANN distance between calving locations ($R^2 = 0.001$, $F = 0.003$, $P < 0.960$, $n = 4$), or between density of calving locations and NDVI ($R^2 = 0.35$, $F = 1.115$, $P = 0.401$) during 2008–2012.

Calving location selection of individual saiga females was best explained by a mixed model that included the factors of NDVI, slope, distances to nearest water, and distances to towns (Table 1.2). The exclusion of a covariate of NDVI from the best model produced the second-ranked competitive model (e.g. ΔAICc value was 1.21). These two models accounted

for ~ 87% of the AICc weight among the 16 possible subset models (Table 1.2). Model-averaged parameter estimates of the full model suggests parturient saiga females preferred locations that were farther from a town and closer to surface water (e.g. avoided farther distances from water), and avoided areas with steeper slopes (Table 1.3). By contrast, NDVI variable emerged as a non-significant predictor affecting saiga calving location selection (Table 1.3). Among the top models (e.g. models $\Delta\text{AICc} \leq 4$), distances to nearest town, surface water, and slope were always included and consequently have maximum relative variable importance values (e.g. the relative importance of these variables was 1.0%); whereas the relative importance of NDVI was 0.65% (Table 1.3). The AUC for the final averaged model was 0.81, indicating good discriminate ability.

Discussion

In some years, the distribution of calving locations was dispersed across the calving area during the parturition period. Dispersing over large areas to distance themselves from other parturient females during calving season may be a strategy to enhance calf survival by reducing the risk of predation (Bergerud et al., 1984; Bowyer et al., 1999). We suspect that such unpredictable distribution of calving locations reduces search efficiency by predators by creating gaps across the landscape (McCauley et al., 1993). Perhaps dispersion is a good anti-predation strategy for Mongolian saiga females during calving period because density is low. In contrast, population densities of saiga antelope are much higher in Kazakhstan where they form large aggregations to give birth, perhaps a predator-swamping strategy to minimize neonate mortality. It remains unclear whether the dispersion of birth locations has a strong effect on saiga calf survival in Mongolia.

As we predicted, saiga calving locations were situated away from towns relative to available sites. The saiga calving sites in Kazakhstan were also located away from settlements (Singh et al., 2010). Singh et al. (2010) suggested this response was likely due to intensity of poaching (Kuhl et al., 2009), but law enforcement appears stronger in Mongolia (Chimeddorj, 2009). Instead, pasture depletion due to livestock grazing is a serious problem in Mongolia (Wesche et al., 2010; Berger et al., 2013), and levels of grazing show strong geographical variation with high impact areas near settled areas (Batkishig and Lehmkuhl, 2003). Consequently, heavily grazed areas in proximity to settlements may have insufficient forage for lactating females or provide less cover for saiga calves and increase exposure to predators. Alternatively, saiga may avoid settlements because free-ranging dogs of livestock herders kill saiga neonates (Buuveibaatar et al., 2010).

Access to water is a critical factor for large herbivores inhabiting arid environments (Bleich et al., 2010), particularly for lactating females during the calving period. The saiga calving sites were located an intermediate distance from nearest source of water in Kazakhstan (Singh et al., 2010), suggesting a trade-off between predation, disease, or disturbance risk and water requirements during parturition (Bowyer et al., 1999; Milner-Gulland et al., 2001; Morgan et al., 2005). However, in our study, calving locations were located closer to surface water in comparison to available habitat. Many rangeland studies have reported that the impacts of concentrated grazing by livestock near the water sources on vegetation dynamics generally lead to marked reductions in forage resources (Stumpp et al., 2005, Fernandez-Gimenez and Allen-Diaz, 2001). The density of livestock herders is lower in summer (e.g. during calving season) in the study area as they move up to the mountains (Buuveibaatar et al., 2010). It is possible that low level of human disturbance and

competition with livestock for resources allows saiga females to use areas close to surface water during the calving period.

Of all the variables measured, vegetation productivity (e.g. NDVI) had the smallest effect on selection of calving locations. Habitat choices of ungulates are associated with vegetation communities with distinctive nutritional properties (Wilmshurt et al., 1999), because nutrition level of plants affects growth rate and subsequent survival of neonates (Cook et al., 2004). Experimental study suggests that without selecting for forage plants that have high concentrations of minerals, saiga antelope in semi-desert range cannot meet their nutritional requirements for weight gain and lactation (Abaturov and Subbotin, 2011). Regions abundant with forbs and shrubs that have significantly greater concentrations of calcium, phosphorous, and magnesium, are important for Mongolian gazelles in Eastern Mongolia, especially shortly before and immediately after calving (Olson et al., 2010). Choice of saiga calving locations therefore is likely constrained by spatial distribution of vegetation communities that have high nutritional value. Future research should put more emphasis on calving location selection behavior at finer scales in relation to vegetation quality, rather than quantity indexed as NDVI.

We used individually observed, multi-year calving locations to determine their spatial patterns and identify factors influencing calving location selection of saiga in western Mongolia. Our findings have shown that the choice of calving locations for saiga antelope is driven by both internal and external factors. Understanding which factors affect calving location selection patterns of Mongolian saiga offers insights for prioritizing habitats for protection.

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Table 1.1 Number of saiga calving locations (single: twin), the Average Nearest Neighbor (ANN) distances among calving locations, extent of calving areas (100% MCP), density of calving locations/km², average (\pm SD) NDVI value (non-normalized) within the calving areas during 2008–2012 in the Sharga Nature Reserve, western Mongolia.

Year	Number of calving locations	ANN, m	Calving area, km ²	Density of calving location #/km ²	NDVI value, Mean \pm SD
2008	27 (13: 14)	2,189	343.4	0.079	926.7 \pm 252.5
2009	31 (22: 9)	1,755	279.2	0.111	875.8 \pm 213.7
2010	35 (34: 1)	1,558	270.8	0.129	743.2 \pm 202.1
2012	16 (12: 4)	2,366	155.0	0.103	699.7 \pm 125.9
Average	27 (20: 7)	1,986	251.9	0.107	811.3 \pm 198.5

Table 1.2 Top ranked models ($\Delta\text{AICc} \leq 4.0$) on basis of minimum AICc explaining difference in calving locations versus random sites in Sharga Nature Reserve, western Mongolia during 2008 –2012.

Model structure	LogLik	AICc	ΔAICc	Weights
<i>water + NDVI + slope + town</i>	-133.11	278.60	0.00	0.57
<i>water + slope + town</i>	-134.79	279.81	1.21	0.30

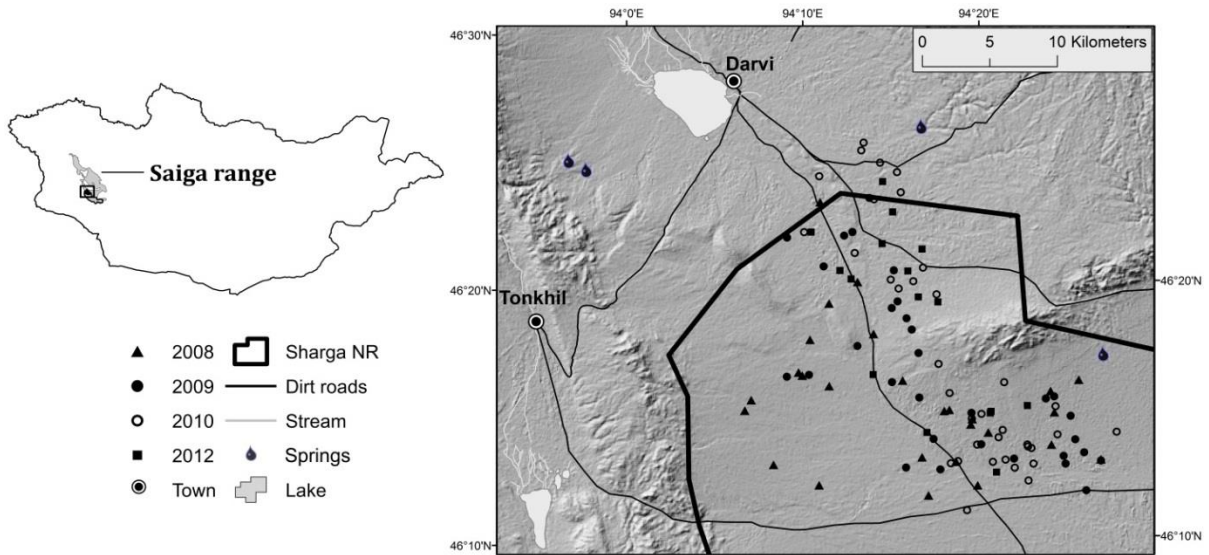
LogLik, Log Likelihood; AICc, corrected AIC; Delta AICc, difference between model AICc and the minimum AICc; Weights, model AICc weight

Table 1.3 Model-averaged parameter estimates of the full model for determining calving location selection of saiga antelope relative to available locations during 2008–2012, western Mongolia. Model-averaged estimates, adjusted standard errors, and relative importance of variables were obtained based on the Akaike Information Criterion for small samples sizes (AICc) statistics following Burnham and Anderson (2002) model averaging procedures.

	Estimate	SE	Z	P	Variable
Intercept	1.077	0.708	1.521	0.128	importance
Distance to water	-0.249	0.051	4.861	< 0.001	1.00
Slope	-0.184	0.082	2.229	< 0.05	1.00
Distance to town	0.126	0.027	4.674	< 0.001	1.00
NDVI	1.325	0.725	1.827	0.067	0.65

Random effect: Year, SD = 1.364e-06

Figure 1.1 A map of study area and calving locations of saiga antelope observed during 2008-2012 in western Mongolia. The box in a country map denotes the “Study site”.



CHAPTER 2

CHANGES IN GROUPING PATTERNS OF SAIGA IN RELATION TO INTRINSIC AND ENVIRONMENTAL FACTORS IN MONGOLIA

Abstract

Factors that affect group sizes in large ungulates are generally poorly understood for species from remote regions. Understanding grouping patterns is important for effective species management, but is lacking for the endangered Mongolian saiga (*Saiga tatarica mongolica*). We studied seasonal changes in the group size and social structure of saigas in relation to environmental and anthropogenic factors in western Mongolia during 2009 – 2012. To identify group size and composition, we observed saigas monthly while conducting monitoring surveys, and weekly while tracking radio-collared animals. We observed 9,268 individuals; median group size was 6.5 (SE = 1.7; range = 1 – 121), and groups of 1 – 5 animals were most common. Seasonality exerted strong effects with the smallest groups in June and largest in December. The largest mixed and nursery groups formed during pre-rutting and summer seasons, respectively, but no seasonal differences were detected for bachelor groups. The best fitting model, including Normalized Difference Vegetation Index, predation rate and season as covariates, explained ~76% of the variation in monthly ‘typical’ group size. Our results are concordant with studies of other arid-adapted ungulates and suggest vegetation productivity, predation rate and biological cycles are responsible for saiga grouping patterns in Mongolia.

Introduction

Several factors influence grouping patterns of ungulates and other social mammals. Grouping is a common response to predation, with a primary benefit of reduced risk of an individual being preyed upon, through increased vigilance (Berger, 1978; Roberts, 1996; Li et al., 2012). Social structure in ungulates is closely related to ecological factors such as habitat type and food availability (Jarman, 1974; Thirgood, 1996). Group size tends to increase with population density (Coulson, 1993; Borkowski, 2000), but habitat openness may also affect grouping patterns (Estes, 1974; Jarman, 1974). Further, grouping behavior is risk sensitive and group size is positively associated with both predation risk and vegetation productivity (Berger, 1988; Banks, 2001).

Understanding the interaction between social systems and life-history patterns is an essential prerequisite for effective conservation (Festa-Bianchet and Apollonio, 2003); it is the foundation upon which monitoring schemes, population models and management strategies are built. Because populations of large mammals are strongly structured (Gaillard et al., 1998), additional demographic indicators, such as sex ratios, group composition and recruitment rates, are often used to monitor populations (Ginsberg and Milner-Gulland, 1994; Milner-Gulland et al., 2003; Buuveibaatar, 2011).

The saiga antelope (*Saiga tatarica*) is a migratory herding species of semi-arid ecosystems of Central Asia (Bekenov et al., 1998). Two subspecies exist, the nominate form (*S.t. tatarica*) in Russia, Kazakhstan and Uzbekistan, and the Mongolian saiga (*S.t. mongolica*; Kholodova et al., 2006). The Mongolian saiga occurs in four subpopulations (Amgalan et al., 2008) in semi-desert or dry steppe depressions in western Mongolia (Bannikov, 1954). While the nominate subspecies undertakes large scale migration tracking

greenness of vegetation (Bekenov et al., 1998; Singh et al., 2010), the Mongolian subspecies does not show nomadic behavior with pronounced seasonal movements (Bannikov, 1954). Saigas are categorized as critically endangered globally (IUCN, 2011); however, Mongolian saigas have been assessed as endangered (Clark et al., 2006). The Mongolian saiga population appears stable in total size, probably owing to enhanced protection (Chimeddorj et al., 2009), and estimates suggest a population of 5000–7000 (Lushchekina et al., 1999; Young et al., 2010). While many aspects of saiga ecology in Mongolia, such as habitat requirements and neonate survival, are relatively well understood (Berger et al., 2008; Buuveibaatar et al., 2013a), little is known about variation in grouping patterns and how environmental and human factors may affect them. Monitoring programmes on Mongolian saigas began in the late 1990s, but efforts were largely confined to winter (Chimeddorj et al., 2009). There is a need for a year-round assessment of grouping patterns to better inform saiga conservation actions, especially because anthropogenic threats are increasing (Lkhagvasuren et al., 2012).

Here, we report seasonal changes in group size and composition of saiga in relation to biotic and abiotic factors. Our primary objectives were to determine (1) the extent to which group sizes differed between seasons, and (2) the relative importance of factors contributing to monthly variation in grouping patterns. The logical bases for our expectations are as follows. We expected grouping patterns of saigas to be positively correlated with vegetation productivity [indexed as Normalized Difference Vegetation Index (NDVI); Bon et al., 1990]. Also, animals in larger groups benefit from the dilution effect, as the individual predation risk per attack is reduced as a function of group size (Hamilton, 1971). Thus, we expected saiga group size would increase during periods when the predation rate is high. We also

expected group size to be negatively influenced by the population density of livestock; livestock probably displace saiga into marginal habitat, resulting in larger groups using these areas. Finally, we expected saiga group size to vary in accordance with seasonality and life history traits, such as calving, rutting and migration.

Materials and Methods

Our research was conducted in western Mongolia across the entire range of Mongolian saigas; we excluded the tiny Mankhan subpopulation as it has only 20–30 animals. Our dataset covered three main subpopulations of saigas: Shargiin Gobi, Khuisiin Gobi and Dorgon Plain (Figure 2.1). The main human populations in the area are concentrated in soums (villages/towns) and saiga range encompasses eight soum territories in the Khovd (Darvi and Chandmani soums) and Gobi-Altai Aimags (Figure 2.1). Semi-nomadic herders are at their highest density during autumn within the study area (Buuveibaatar et al., 2010). Domestic livestock consists primarily of goats and sheep with small numbers of camels and horses. There is a lack of permanent surface water and local herders rely heavily on hand-drawn wells or snow.

The study area is bounded by the Altay Mountains to the west; elevations range from 900 to 4070 m. The region is desert-like with a short growing season, long harsh winters and a strongly variable climate, which governs the availability of food plants (Yu *et al.*, 2004). During 1975–2007, average air temperature during summer and winter was 18 and -20°C, respectively (Buuveibaatar et al., 2013a). The study area receives ~100 mm precipitation annually. Vegetation is sparse and onions (*Allium* spp.), grasses (*Stipa* spp.) and anabasis (*Anabasis brevifolia*) are the most common plants (Buuveibaatar et al., 2012). Common

predators in this system are grey wolves (*Canis lupus*), red foxes (*Vulpes vulpes*), corsac foxes (*V. corsac*), lynx (*Lynx lynx*) and raptors, such as golden eagles (*Aquila chrysaetos*) and cinereous vultures (*Aegypius monachus*).

Data sources

Two data sources for saiga groups were analyzed. First, we used saiga rangers' monthly monitoring data collected during November 2009 to December 2011. As a part of the 'Saiga Conservation Project' implemented by World Wide Fund for Nature (WWF), a total of 11 saiga rangers (three in Dorgon Plain, four in Shargiin Gobi and four in Khuisiin Gobi) conducted monthly observations of saiga groups (Figure 2.1). All rangers received training on monitoring of saigas and were equipped with binoculars, GPS units, compasses, maps and rangefinders. Monitoring occurred at the same time in each month to avoid double observation of saiga groups by different saiga rangers. During the monthly monitoring, each ranger recorded group size and composition along *a priori* defined fixed transects. A saiga group was defined as one or more saigas at least 500 m from other conspecifics that moved cohesively. The WWF protocol did not include a method to obtain observer error, so for this study, we assume it to be constant across rangers, days, seasons and years. Second, 116 saiga calves were captured and fitted with a 70-g expandable VHF radio-collar to monitor their movements and survival during 2008–2010 (Buuveibaatar et al., 2013a). Animal handling methods were approved by the Institutional Animal Care and Use Committee at the University of Massachusetts Amherst (protocol 2010-0001). Up to three times per week, we located and recorded group size and composition of groups of saigas encountered while radio-tracking collared individuals. For the monitoring data of the marked animals, we

assumed all observations to be independent because no more than one observation was made for each group on any given day. Weekly observations of saiga group size were pooled into months to determine average monthly group size.

Group size definitions

We describe the distribution of group sizes using median values for comparison with other studies, and because group sizes, in general, typically exhibit an aggregated (right-skewed) distribution (where most groups are small, few are large and a very few are very large), and thus are not normally distributed or accurately described by mean values. For modeling purposes, we calculated and used ‘typical group size’ to examine how saiga grouping patterns respond to environmental and social factors. Typical group size is defined as $\sum Gi^2 / \sum Gi$, where Gi is the size of the i th group (Jarman, 1974); this identifies the group size in which the most animal lives (also ‘mean crowding’; Reiczigel et al., 2008). Typical group size has advantages over mean or median group size because it is less sensitive to the number of records of solitary animals (White et al., 2012).

Predictor variables

We evaluated vegetation productivity, predation rate, mean temperature and precipitation, distribution and number of livestock herders, and a dummy predictor variable. To estimate vegetation productivity in saiga range, we acquired NDVI scenes from the 10-day SPOT (Satellite Pour l’ Observation de la Terre) dataset (1 x 1 km spatial resolution – <http://www.spotimage.fr>), corresponding to the study period during November 2009–March 2012 (i.e. 31 months x 3 = 93 scenes). To estimate monthly variation in NDVI value, random

points ($n = 10\,000$) were generated within the entire saiga range, using the Hawth's Tools extension of ArcGIS 9.3 (ESRI), of which 9257 were left after removing the points from water bodies and high altitudes, where saigas were unlikely to occur. The NDVI values were extracted for all randomly generated points using the Extraction tool in the Spatial Analyst extension. Ten-day NDVI values for each random point were pooled by month to estimate average monthly NDVI values for all years.

A predation rate index was developed using survival data from the radio-collared saiga calves during 2008–2010 (Buuveibaatar et al., 2013a). During the 3-year survey, 56 (48%) of the marked animals died from five sources of mortality, including raptors (36%), foxes (18%), lynx (2%), parasites (2%) and unknown causes (43%). To calculate predation rate, we estimated monthly mortality rates for each cohort by determining the proportion of the marked animals killed by predators (e.g. raptors, foxes and lynx) and averaged these estimates across years. Predation rate on calves was highest in July (e.g. ~30% of marked animals killed by predators during 2008–2010; Buuveibaatar et al., 2013a).

Mean temperature and total precipitation were calculated for each month; data were derived from local meteorological stations in the eight soums adjacent to saiga range (Figure 2.1).

Data on seasonal distribution and number of nomadic livestock herders in the study area throughout the year were obtained from WWF Mongolia databases. Because livestock data were incomplete for each herder, we used density of livestock herders as a proxy for livestock population density. The density of livestock herders was calculated by dividing the total number of herders by the size of the study area for four seasons: winter (December–February), spring (March–May), summer (June–September) and autumn (October–

November). Seasons were based on movements of the livestock herders within the study area as determined from our own observations and from interviews with rangers monitoring livestock herder activity.

Finally, to account for variation in group size in relation to seasonal sociality (e.g. calving, rutting and migration), we created a dummy variable using seven biological seasons: calving (June), summer (July–August), autumn (September–October), pre-rutting (November–December), rutting (January), winter (February–March) and spring (April–May), based on behavioral and climatic considerations (Buuveibaatar, 2011).

Data analysis

To examine variation in social structure, groups were defined as bachelor groups (males), nursery groups (females and subadults or juveniles of unknown sex) and mixed-sex groups. Cases, in which sex or group composition could not be determined reliably, were excluded from the analysis. There were no differences in median group sizes between the monitoring and tracking data for all types of groups (Table 2.1). Therefore, we combined both datasets for analysis. Group compositions were identified for 619 (~74%) of the groups across years, and included 197 mixed, 324 nursing and 98 bachelor herds (Table 2.1). Monthly observations were pooled into the seven biological seasons. A Kruskal–Wallis one-way analysis of variance (ANOVA) was used to test for differences in monthly group size and seasonal group composition of saigas. Differences in monthly median and typical group sizes were examined using Mann–Whitney tests. Median and typical group sizes are reported with standard errors (SE).

Generalized linear models (GLM) with Gaussian error structure were used to evaluate effects of NDVI, predation rate, climate, density of livestock herders and seasonality on monthly variation in typical group size of saiga. We excluded monthly total precipitation ($r^2 = 76.4$, $F = 32.5$, $P < 0.001$) and average temperature ($r^2 = 90.1$, $F = 90.8$, $P < 0.001$) because they were positively related to average monthly NDVI value. Therefore, our final model included four explanatory variables (NDVI, predation rate, livestock herder density and season). All variables were assessed for correlation, using variance inflation factors (VIF), which show how much the variance of the coefficient estimate is being inflated by multicollinearity; a value exceeding five is considered as evidence of multi-collinearity (Das and Chatterjee, 2011). None of the variables considered showed a high VIF (NDVI = 1.61; predation rate = 1.98; herder density = 1.32; and season = 1.31); hence, they were included in the same model. We used the Akaike Information Criterion, corrected for small sample sizes (AICc) and Akaike weights for model selection (Burnham and Anderson, 2002). We considered the model with the smallest AICc value to be the best model to fit the data and any model within 2 AICc values as a competing model (Burnham and Anderson, 2002). Akaike weights were used to assess the strength of evidence of one model versus another. Relative importance of variables affecting saiga group size was evaluated using the method of hierarchical variance partitioning (Walsh and MacNally, 2004). All the statistical analyses were done in program R 2.15.2 (R Development Core Team, 2008).

Results

In total, 836 groups and 9268 individuals were observed between November 2009 and March 2012. Groups of 1 – 5 animals were the most (36.6%) frequent, followed by 6 – 10

individuals (26.4%) and 11 – 15 individuals (13.2%); groups with >30 individuals were rare (e.g. 5.5% of the total; Figure 2.2). Overall median group size was 8.0 ± 1.7 (range, 1 – 121 individuals). Median group sizes varied monthly (Kruskall–Wallis one-way ANOVA: $H_{11} = 103.35$, $P < 0.001$) with smallest groups forming in June (1.5 ± 0.3) and largest in December (12.5 ± 1.7 ; Figure 2.3). Similarly, typical group size was smallest in June (3.0 ± 0.4) and largest in November (31.0 ± 1.8 ; Figure 2.3). Typical group sizes varied monthly (Kruskall–Wallis one-way ANOVA: $H_{11} = 115.57$, $P < 0.001$). Overall typical group size (16.0 ± 2.3) was twice larger than median group size (Figure 2.3; Mann–Whitney test: $W_I = 194.5$, $P < 0.01$).

Median size of mixed groups was largest during the pre-rutting period (14.5 ± 1.6) and smallest during calving (7.5 ± 1.5 ; Figure 2.4), and varied between seasons (Kruskall–Wallis one-way ANOVA: $H_6 = 14.17$, $P < 0.01$). For nursery groups, median group size was the lowest during calving season (1.5 ± 0.1) and largest during summer (6.5 ± 1.2 ; Kruskall–Wallis one-way ANOVA: $H_6 = 9.34$, $P = 0.05$). Median bachelor group size ranged from 2.0 to 5.0 males (SE = 0.2 to 1.2) and median group size did not differ seasonally (Kruskall–Wallis one-way ANOVA: $H_6 = 1.21$, $P = 0.26$; Figure 2.4).

The best model determined by GLM, including the factors NDVI, predation rate and season, explained ~76% of the variation in typical monthly group size of saigas (Table 2.2). This model accounted for 56% of the AICc weights among the 14 subset models we considered. The exclusion of predation rate from the best model produced the second-ranked model, which was competitive (AICc weight = 24%; Table 2.2). Relative support of NDVI (39%) was the greatest, followed by predation rate (34%), season (22%), and herder density (5%), in explaining changes in typical group size. In the top model, typical group size was

negatively related to NDVI and predation rate, and season was positively associated with monthly variation (Table 2.3).

Discussion

Although the range of saiga group size in this study was large (range = 1 – 121), the distribution was heavily skewed towards smaller groups. Groups of 1–5 animals were encountered most frequently (36.6%). Small groups are more common in arid environments, where food is likely to be sparsely distributed (Berger, 1988). Goitered gazelles (*Gazella subgutturosa*) also occupy areas of Mongolia where high-quality food items are sparse, and individuals tend to be dispersed in relatively small herds (Blank et al., 2012).

Changes in monthly group size appeared to be dependent on the annual biological cycle of saigas. This was expected since many large and medium-sized ungulates congregate during the rut and then form smaller groups at other times (Borkowski and Furubayashi, 1998; Blank et al., 2012). The median mixed-group size of saigas was lowest during summer, similar to populations in Kazakhstan and Russia (Bekenov et al., 1998; Kuhl, 2008). Grouping in smaller herds in summer is probably associated with calving. Pregnant females leave their groups and remain solitary during this period (Sokolov, 1974). Saiga groups reached maximum size during the pre-rutting season (e.g. November–December), with a secondary peak in group size observed in February. These observations probably reflect saiga reproductive behavior. Males start establishing harems at the beginning of the rut (Dulamceren and Amgalan, 1994), and harem herds join together to form larger groups once the rut ends (Bekenov et al., 1998).

Food ultimately limits the formation of groups in many ungulates across different landscapes (e.g. bighorn sheep *Ovis canadensis*: Payer and Coblenz, 1997; sika deer *Cervus nippon*: Borkowski and Furubayashi, 1998). However, contrary to our prediction, we found a negative association between group size and NDVI. In other words, large groups were more common when food availability was lowest during winter and spring (November–April). Food availability is limited by snow cover during winter and saigas may be forced to concentrate in small patches of good vegetation, resulting in formation of larger groups. Similarly, a mega-herd of Mongolian gazelles was observed in eastern Mongolia, when availability of suitable foraging patches was severely reduced by severe drought (Olson et al., 2009). Also, human disturbance or hunting may influence grouping dynamics of saigas. Poaching of saigas seems to occur when they form larger groups during November–April when food availability is low (B. Batsaikhan, pers. comm.), but data on actual poaching intensity throughout the year are not available.

As we predicted, typical group size responded negatively to increased predation rate. Animals in larger groups generally benefit from the ‘dilution effect’, as the individual predation risk per attack is reduced as a function of group size. Thus, forming large groups in saigas is probably a behavioral response to mitigate predation risk. Further, herd formation reduces search efficiency by predators by creating gaps in prey availability across the landscape, analogous to the effects of weak diffusive movements by predators or prey (McCauley et al., 1993; Keeling et al., 2000). Thus, sociality may have reduced the frequency at which predators encounter prey (Cosner et al., 1999; Nachman, 2006). Negative effects of large group size, such as increased competition (i.e. reduced foraging success) and

risk of disease transmission, can act against the formation of larger groups, but are outweighed by the advantages when predation risk is high.

There was a positive, but insignificant, relationship observed between typical group size and the density of livestock herders. Although overall density of the saiga population is low (Young et al., 2010), group size may be affected by sympatric livestock herds, given that livestock biomass currently exceeds that of saigas by nearly 50:1 in the study area (Berger et al., 2013). Livestock herder density may not have reflected livestock population densities sufficiently to detect any impacts on saigas; however, we suggest the scale at which saiga and livestock occupy the landscape is too different to influence saiga grouping behavior. In our study area, there is a lack of permanent ground water and local herders rely heavily on hand-drawn wells that are limited in their spatial distribution. The distribution of herders is highly restricted to wells, while saigas move over large areas (Berger et al., 2008).

In summary, we found factors involved in explaining changes in saiga group size in western Mongolia could have implications for saiga conservation efforts. Our results suggest saiga form large groups to reduce predation risk and increase reproductive opportunities. Large groups are also easier for humans to find, especially those interested in poaching, and are likely to cause disruptions in saiga grouping behavior.

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Table 2.1 Descriptive statistics of median size of group types (pooled across months and years), based on observations during VHF tracking and monthly population monitoring. Median group sizes were compared using nonparametric Kruskal–Wallis tests.

Group type	Tracking			Monitoring			Difference	
	Median \pm SE	Range	N	Median \pm SE	Range	N	<i>H</i>	<i>P</i>
Mixed	9.5 \pm 1.4	2 – 39	110	11.0 \pm 1.7	2 – 56	195	1.92	0.19
Nursing	7.0 \pm 1.1	2 – 35	157	5.5 \pm 0.8	2 – 21	84	2.07	0.06
Bachelor	4.0 \pm 0.5	1 – 8	18	4.5 \pm 0.7	1 – 13	55	1.04	0.55
Overall	6.8 \pm 1.0	2 – 39	285	7.0 \pm 1.1	2 – 56	334	0.85	0.69

Table 2.2 Model selection results for estimation of factors affecting grouping dynamics of saigas in western Mongolia during 2009–2012. We present results of top 10 ranked models that have AICc weight >0.002, although 14 subset models were considered.

Model structure	AICc	Delta AICc	AICc weights	Deviance
<i>Season + NDVI + predation</i>	83.961	0.000	0.562	0.761
<i>Season + NDVI</i>	85.593	1.632	0.248	0.815
<i>Season + NDVI + livestock</i>	86.701	2.740	0.143	0.797
<i>Predation</i>	91.735	7.774	0.041	0.379
<i>Season + predation</i>	92.329	8.368	0.009	0.519
<i>NDVI</i>	92.654	8.693	0.007	0.329
<i>NDVI + livestock</i>	93.024	9.063	0.006	0.490
<i>NDVI + predation</i>	93.050	9.089	0.006	0.489
<i>Predation + livestock</i>	95.076	11.115	0.002	0.395
<i>Season</i>	95.185	11.224	0.002	0.172

AICc – corrected AIC, Delta AICc – difference between model AICc and the minimum AICc, AICc weights – model AICc weight, Deviance – proportion of deviance explained by the model.

Table 2.3 Parameter estimates of the top model explaining changes in monthly typical group size of saigas in western Mongolia, during 2009 – 2012.

	Estimate	SE	<i>t</i> value	<i>p</i>
Intercept	20 .31	4.79	4.23	< 0.002
NDVI	-283.05	60.17	-4.70	< 0.001
Predation rate	-4.241	1.50	-3.47	< 0.001
Season	3.69	0.91	4.06	< 0.002

Figure 2.1 A map of the study area in western Mongolia. Monitoring and patrolling areas of 11 saiga rangers are shown as polygons.

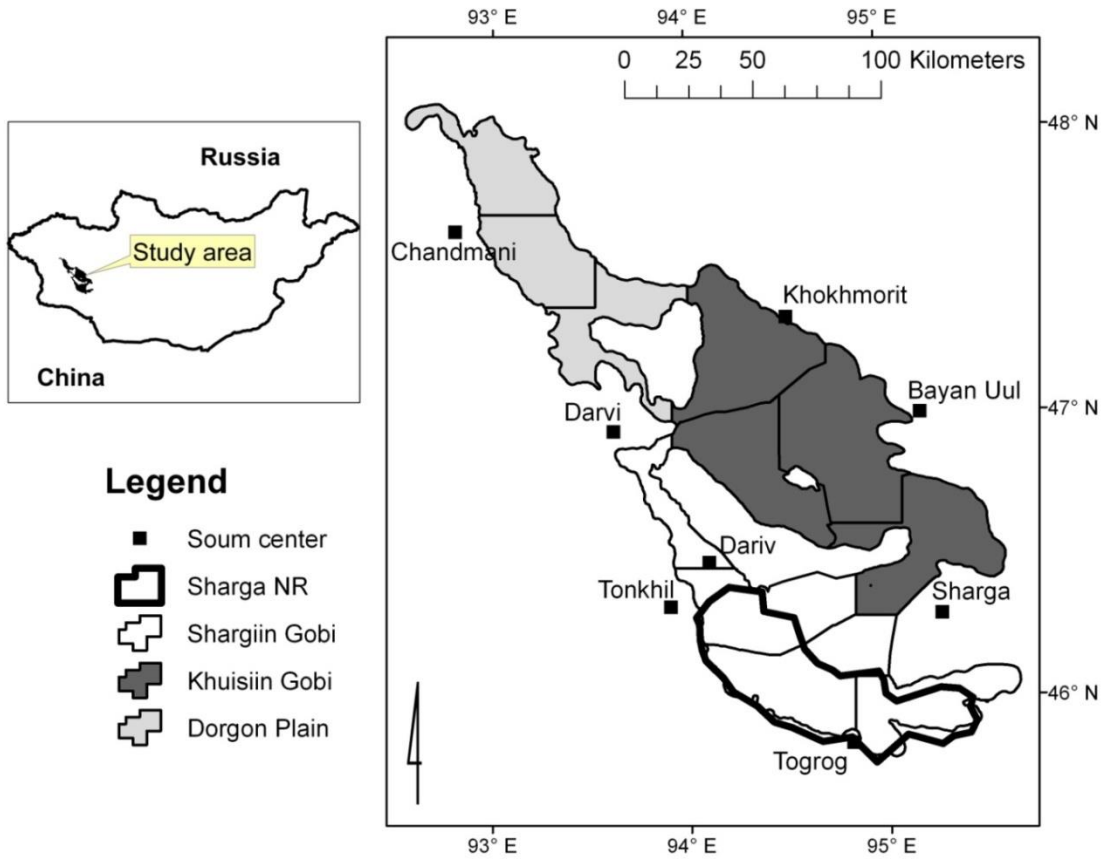


Figure 2.2 Frequency occurrence of group sizes in Mongolian saiga during 2009–2012 in western Mongolia.

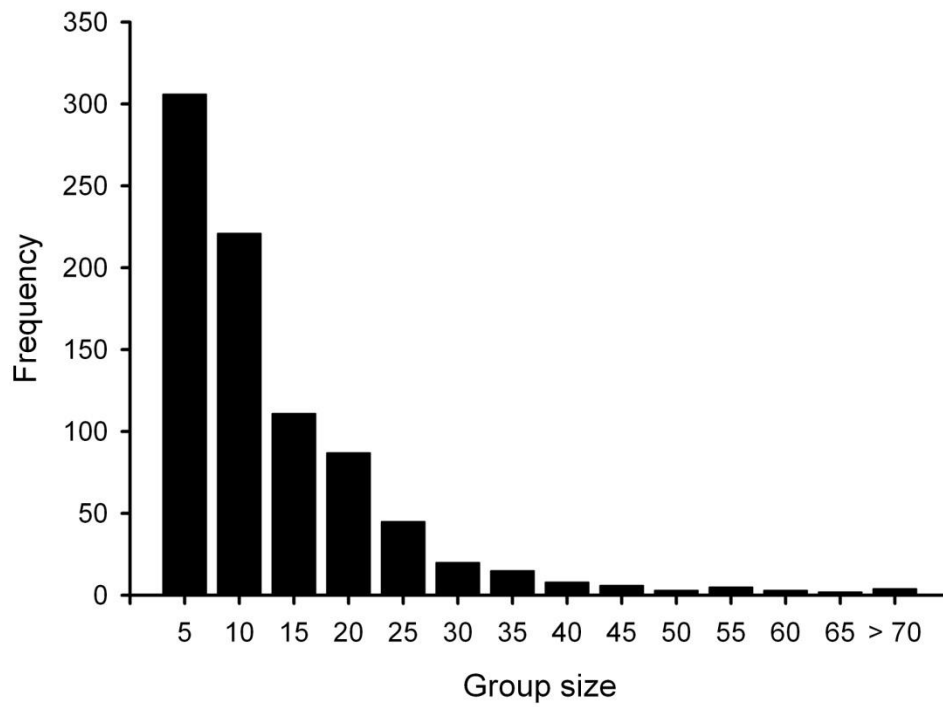


Figure 2.3 Monthly variations in typical and median group size of Mongolia saiga (*Saiga tatarica mongolica*) during 2009 – 2012 in western Mongolia.

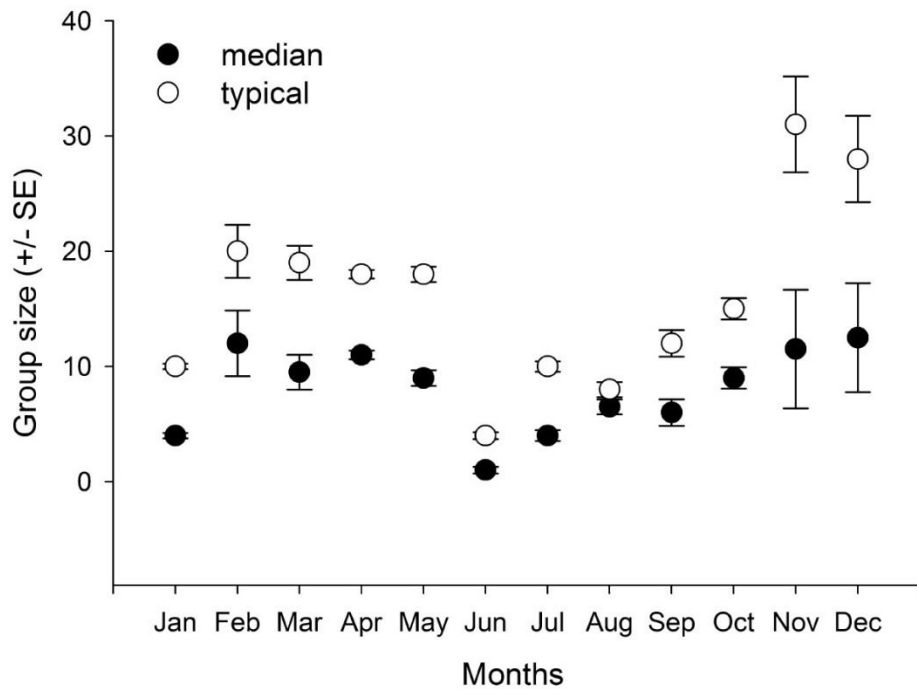
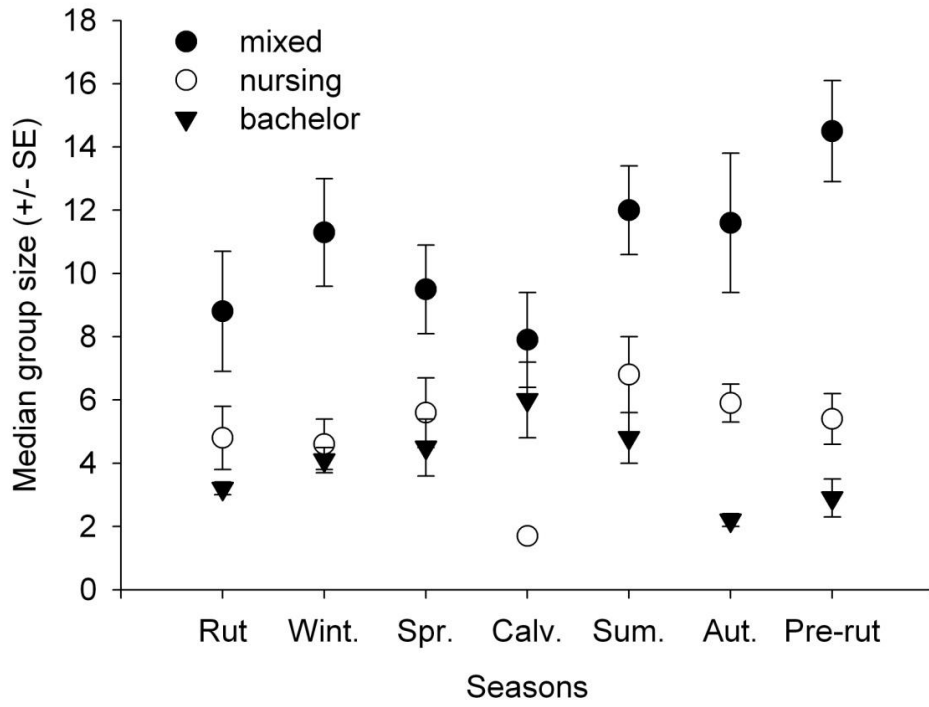


Figure 2.4 Seasonal changes in group composition of Mongolian saiga (*Saiga tatarica mongolica*) during 2009–2012 in western Mongolia.



CHAPTER 3

MONGOLIAN GOBI SUPPORTS THE WORLD'S LARGEST POPULATIONS OF KHULAN AND GOITERED GAZELLE

Abstract

Mongolia's Gobi desert ecosystem, a stronghold for substantial populations of Asiatic wild ass (or khulan; *Equus hemionus*) and goitered gazelle (*Gazelle subgutturosa*), currently faces conservation challenges from rapid economic development, including mining-related infrastructure projects. There is a paucity of reliable data on population abundance for these ungulates in the region, which makes it difficult to assess how they are responding to an increasing human footprint on the landscape. Our aim was to obtain abundance estimates of khulan and goitered gazelles to inform their management and form the basis of a long-term monitoring program. Each year during 2012-2015, we surveyed a total of 64 line transects spaced 20 km apart, with a total of 3,464 km of survey effort across a 78,717-km² area. Distance sampling analysis provided annual estimates of density and abundance while an aerial survey in 2013 allowed us to cross reference the results of the two independent survey methods. Overall, we observed 784 groups (14,608 individuals) of khulan and 1,033 groups (3,955 individuals) of goitered gazelles during the four surveys. The abundance estimates for 2013 were 35,899 (95% CI = 22,680 – 40,537) khulan and 28,462 (95% CI = 21,326 – 37,987) goitered gazelles. These estimates were highly congruent with the results from the aerial survey that overlapped with our ground-based survey in space and time. Our findings

confirm that Mongolia's Gobi desert supports the largest population of khulan and goitered gazelle in the world, and we provide a critical update on the status of the two species.

Introduction

The vast rangelands of Central Asia provide habitat for a large number of migratory ungulates (Mallon and Jiang, 2009). One of the most spectacular and important regions for the conservation of these ungulates is the Gobi-Steppe ecosystem of Mongolia, comprising the largest area of intact grassland in the world (Batsaikhan et al., 2014). Mongolia's southern Gobi area, in particular, is an iconic ungulate stronghold that supports substantial populations of the Asiatic wild ass (khulan in Mongolian – *Equus hemionus*) and the goitered (or black-tailed – *Gazella subgutturosa*) gazelle (Reading et al., 2001).

A growing human population, expanding exploitation of natural resources, and the development of infrastructure in the region place increasing pressure on these species and their habitats (Kaczensky et al., 2011a; Ito et al., 2013; Batsaikhan et al., 2014). As the footprint of human development continues to expand in the landscape, conservation management to ensure the survival of these species requires long-term datasets that provide accurate information on changes in abundance and distribution over space and time.

Khulan are listed as Near Threatened on the IUCN Red List and the species has lost as much as 70% of its range (Kaczensky et al., 2015a). Poaching represents the primary driver of past population declines (Stubbe et al., 2012), although habitat loss and fragmentation across the species' range may also be responsible (Batsaikhan et al., 2014). Goitered gazelle are categorized as Vulnerable in the IUCN Red List (Mallon, 2008), with a decreasing population trend attributed primarily to exploitation, habitat degradation, and

human disturbance (Clark et al., 2006). Past population surveys of khulan and goitered gazelles in Mongolia were based on limited survey efforts and non-standardized survey protocols, produced imprecise and potentially biased estimates (Lhagvasuren, 2007; B. Lhagvasuren and S. Strindberg, unpubl. data).

The effectiveness of mitigation measures with the aim to reduce impacts caused by mining-related activities and infrastructure developments cannot be evaluated without proper monitoring. Consequently, obtaining unbiased and precise estimates of density or abundance for the plains ungulates inhabiting this large landscape is crucial. Despite population size estimation being one of the most critical prerequisites for conservation planning, long-term monitoring is often hindered by lack of funds and logistical constraints. For this reason robust monitoring programs using contemporary scientific methods are frequently still not being employed in Central Asia (Singh and Milner-Gulland, 2011).

Distance sampling techniques are well-established for the estimation of wildlife density and abundance (Buckland et al., 2001) with associated survey design and analysis software (Thomas et al., 2010). Distance sampling line transect surveys are increasingly used for population estimates of wild ungulates in Mongolia (e.g. Olson et al., 2005; Young et al., 2010; Wingard et al., 2011), and has been shown to be an appropriate and cost-effective method for estimating the abundance of relatively large and conspicuous species at low population densities over vast open areas (Sutherland et al., 2006). However, most surveys have only been conducted once or twice and were rarely if ever compared to simultaneous surveys employing a different method.

Our overarching goal was to develop research techniques and implement field surveys that can provide baseline and long-term data. Here we present the results of

extensive ground-based distance sampling surveys conducted for khulan and goitered gazelles for consecutive years during 2012-2015 in the Southern Gobi of Mongolia. We also make recommendations for further improvements to the ground-based ungulate survey efforts to ensure a robust long-term monitoring program. Lastly, we compare the distance sampling results to an independent aerial survey conducted in 2013 across a larger area using different methods.

Study area

The extent of the 98,216 km² study area in the Mongolia's Southern Gobi (Figure 3.1) was defined using existing information on the distribution of khulan, as they have the larger species range (Kaczensky et al., 2011a). Elevations in the study area range from 683 m to 1,884 m. Precipitation averages ~ 150 mm per year and annual temperatures range between -35⁰C and +40⁰C. Surface water is restricted to springs, some of which are permanent, primarily located in or near mountain ranges. There are few tree species, including saxaul *Haloxylon ammodendron* and elm *Ulmus pumila*, which are confined to the river valleys and basins. The most common vegetation includes *Stipa* spp., *Artemisia* spp., *Allium* spp., and *Anabasis* spp. Argali sheep (*Ovis ammon*) and ibex (*Capra sibirica*) are present in the mountainous areas, and larger mammalian predators include wolves (*Canis lupus*), lynx (*Lynx lynx*), red foxes (*Vulpes vulpes*), and corsac foxes (*V. corsac*).

There are four protected areas which comprise approximately 20% of the study area (Figure 3.1). Human populations in the region are concentrated in *soums* (villages/towns; Figure 3.1), with the rural population primarily consisting of semi-nomadic livestock herders. The Southern Gobi is the center of the cashmere goat industry in Mongolia, the key source of

income for local herders (Berger et al., 2013). Thus, domestic livestock consists primarily of goats and sheep with small numbers of camels and horses. Our study area is bordered to the south and east by two impermeable linear infrastructures constructed in the 1950s, namely a fenced border with China, and the Trans Mongolian Railroad corridor (fenced on both sides as well). In addition, two parallel paved roads that connect major mines with the Chinese border crossing are present to the west.

Materials and Methods

We conducted distance sampling line transect surveys using a systematic survey design with a random start and a spacing of 20 km between transects (Strindberg et al., 2004). A survey design with 29 transect lines and 4,820 km total survey effort was generated using the Distance 6 software (Thomas et al., 2010). Transects were oriented north-south to facilitate their coverage in the field and to minimize potential problems in sightability caused by glare. Due to ruggedness of the topography (mountains and sand dunes), we truncated 28% of the total transect length, resulting in 64 shorter transects (range = 4.25 – 204.78 km). This gave a total of 3,464 km of survey effort across 78,717 km² excluding those portions of the original 98,216 km² area that were inaccessible (Figure 3.1). The final design provided a sufficient number of replicate lines to ensure that variation in encounter rate could be estimated with adequate precision, and ensured sufficient observations per ungulate species to fit the detection function (Buckland et al., 2001).

The ground-based surveys were conducted in the first half of October in 2012, and in late May and early June in 2013, 2014, and 2015; each survey lasted 14–18 days. Transects were driven at a rate of 20–30 km per hour during daylight hours using a global positioning

system (GPS) for orientation. Observers scanned the area in front of them and out to 90° on either side. When a group of the ungulates was detected, the location, group size, radial distance (r), and sighting angle (θ) were recorded using a GPS, compass, binoculars, spotting scope and rangefinder. Ungulate groups often began to run after we detected them, and so we used a landscape feature at the point of detection to measure r and θ . The unit of observation was the group and thus measurements were recorded to the center of the group. The perpendicular distance of observation from the transect line, calculated as $x = r\sin(\theta)$ within the Distance software, was used to estimate a detection probability function. The detection function gives the probability that an animal group is detected, as a function of distance from the line $g(x)$.

Data analysis

Differences in ungulate group sizes between autumn and spring seasons were examined using a t -test. One-way analysis of variance (ANOVA) was used to test for variability in ungulate group size among the three spring surveys. The ungulate group sizes were log transformed because the data violated an assumption of normal distribution.

Data were analyzed using the Distance software to obtain density and abundance estimates for each of the ungulate species (Thomas et al., 2010). Ungulate densities were estimated as:

$$\hat{D} = \frac{n\hat{f}(0)\hat{E}(s)}{2L},$$

where L denotes the aggregate length of the transects, n is the number of ungulate groups observed, $f(0)$ is the probability density function of observed perpendicular distances

evaluated at $x = 0$, and $E(s)$ is the estimated expected group size (Buckland et al., 2001). The density of individuals is multiplied by the surface area of the study area to obtain the corresponding abundance estimate (N).

Exploratory analyses were conducted to examine options for truncation and grouping intervals to improve model fit for the detection probability function. Following Buckland et al. (2001), a variety of key functions and adjustment term combinations were considered to model the detection function, with data either stratified by survey season or pooled across seasons. Histograms of the data and goodness of fit tests were used to identify any violations of assumptions. Akaike's Information Criterion for small sample sizes (AICc) was used in model selection, with particular attention paid to model fit at distances near zero, which is most important for robust estimation (Buckland et al., 2001). The data were always stratified to permit estimation of encounter rate and expected group size specific to each survey season. To deal with potential size bias in estimation of group size, we used the expected group size rather than the average group size when the regression line fit to the natural logarithm of group size versus $g(x)$ was significant at a 15% alpha level. We used a z -test to determine if a change in density could be detected between surveys (Buckland et al., 2001) at the 10% significance level (more conservative for management purposes).

Due to poaching activities in some areas, ungulates sometimes moved before distance and angle measurements could be taken. To avoid potentially negatively biased density estimates caused by responsive movement away from the observer, the survey teams looked far ahead to detect animals and attempted to obtain measurements before movement occurred. In contrast, heaping at zero, where there are more observations than expected on or close to the line, can produce positively biased estimates of density.

An aerial survey was completed in 2013 and covered an area of 150,000 km² across much of the Southern Gobi (Norton-Griffiths et al., 2015). This gave the unique opportunity to contrast the results produced by two independent survey methods. We assessed accuracy of the 2013 ground survey by comparing it to the aerial survey results that used a photograph-based method. The plane followed north-south transects at 5- and 10-km spacing, with photographs (n = 101,000) taken every 250 m, on average. Photos were taken with the camera angled straight down at an altitude of about 427 m, and each photo covered an area of approximately 125 m x 185 m. The aerial survey included the entire ground survey area, except for a 15-km-wide strip with an area of 10,781 km² along the Mongolia-Chinese border where the plane was not allowed to fly (Figure 3.2). No type of terrain was excluded. In total, the overlap zone between the aerial and ground surveys comprised 75,281 km², when the areas inaccessible to the ground survey were removed. The aerial survey was conducted between May 23 and July 02 in 2013, covering the May 24 – June 07 period during which the ground survey was conducted in 2013. Population estimates of the ungulates for the aerial survey data were made using method 2 of Jolly for unequal sized sampling units (Jolly, 1969). There were difficulties distinguishing between goitered and Mongolian gazelles in aerial survey photographs. However, there is no reason to assume that the total gazelle count is inaccurate. During the ground survey, 279 groups of goitered gazelles and 75 groups of Mongolian gazelles were seen with an average group size of 3.1 and 9.1 individuals, respectively. Very few, if any, adjacent photos taken during the aerial survey contained gazelles. Therefore, we can assume that each group of gazelles is captured by a single photo. Thus a ratio of Mongolian to goitered gazelle groups obtained from the temporally coincident ground survey (where species identification is not a significant problem) was applied to the

total gazelle estimate from the aerial survey to obtain separate estimates for each of the gazelle species.

Results

Overall, 784 khulan groups and 14,608 individuals, and 1,033 goitered gazelle groups and 3,955 individuals were observed during 2012 – 2015 (Table 3.1). Khulan and goitered gazelles formed significantly larger groups in autumn (*t* test; khulan: $t = 5.62, p < 0.001$, goitered gazelle: $t = 5.84, p < 0.001$) compared to spring. There was no difference in group size of khulan among the three spring surveys ($F = 1.48, p = 0.22$); however, the average group size of goitered gazelles was significantly larger in Spring 2015 ($F = 6.68, p < 0.001$). For the pooled data across four years, overall mean group size was $20.26 \pm \text{SD } 36.78$ for khulan and $3.88 \pm \text{SD } 3.70$ for goitered gazelles (Table 3.1).

Khulan groups of 2 – 5 individuals were seen most frequently during 2012 – 2015 (range = 24.0 – 42.1%; Figure 3.3a), whereas only 4 – 12 groups (i.e. 2.0 – 9.3% of overall khulan groups) of >100 individuals were observed. Distribution of goitered gazelle group size was also heavily skewed towards smaller groups; groups of < 5 individuals comprised 67.1 – 88.5% of all groups recorded during the four surveys (Figure 3.3b). Goitered gazelle groups with >10 individuals were rare, comprising only 3.4 – 8.4% of groups observed for all years.

Pooling the data across the first three survey seasons provided a larger and better data set to deal with the likely inaccuracies due to responsive movement and heaping at zero and to fit an unbiased detection function. A separate detection function was fit to the Spring 2015 data as there was no evidence of substantial assumption violations. The pooled data for the

first three surveys were right-truncated at 1,400m (4% “loss” of data) and placed in 6 equal-sized intervals for the final model (Figure 3.4a). The resulting estimate for the detection probability was 0.42 (95% CI = 0.38 – 0.46), with an associated effective strip width ($\hat{\mu}$) of 584.53 m (95% CI = 526.41 – 649.08). The Spring 2015 data were right-truncated at 1,250m (8% of the data) and placed in 7 equal-sized intervals (Figure 3.4b). The subsequent estimate for the detection probability was 0.48 (95% CI = 0.42 – 0.54), with an associated $\hat{\mu}$ of 596.42 m (95% CI = 530.22 – 670.88). Average group size (\bar{s}) was smaller than the expected group size $\hat{E}(s)$ for the first two surveys, while \bar{s} was larger than $\hat{E}(s)$ for the last two surveys. Size bias was statistically significant for all Spring surveys, and thus $\hat{E}(s)$ was used in the estimation for these surveys and \bar{s} for the Autumn 2012 survey. Encounter rate (n/L) was lowest (i.e. 0.03 groups/km) and the group sizes markedly larger in the Autumn 2012 (Table 3.2), with the latter leading to the largest individual density for that survey (Table 3.3). Density estimates varied from 0.45 khulan/km² in Spring 2013 to 0.83 khulan/km² in Autumn 2012, with total population estimates of 35,899 – 65,739 khulan, respectively (Table 3.3). Despite the larger density and abundance estimate for the Autumn 2012 survey, differences among the surveys failed to be statistically significant due to large confidence intervals. Uncertainty in the density estimates in all four surveys was mostly due to estimation uncertainty in the encounter rate (63.6 – 67.4% of the variance), followed by the group size (29.4– 31.2%), and detection probability (3.0 – 6.4%).

The data were pooled across four survey seasons and distances grouped into 8 equal-sized intervals to help fit an unbiased detection function because heaping at zero and responsive movement was evident in all surveys. The pooled data from the four surveys were right-truncated at 700m (< 4% of the data) for the final model (Figure 3.4c). The best fitting

model was a half-normal function with cosine adjustment terms giving an estimated detection probability of 0.37 (95% CI = 0.34 – 0.40), with an associated $\hat{\mu}$ of 258.45 m (95% CI = 241.11 – 277.05). Encounter rate was similar between the four surveys (around 0.06 groups/km), although the expected group size was largest in Autumn 2012 (Table 3.2). For all seasons there was significant size bias with \bar{s} being consistently larger than $\hat{E}(s)$ likely due to smaller groups being missed at larger distances. Thus $\hat{E}(s)$ was used in the estimation of density and abundance. Density estimates varied between 0.36 and 0.50 gazelles/km², with total population estimates of 28,462 and 39,602 goitered gazelles, respectively (Table 3.3). There was no detectable difference in density estimates among the surveys, although the larger $\hat{E}(s)$ making the final individual density and abundance estimate larger for the Autumn 2012. As for khulan, most of the variance in the abundance estimate was due to the encounter rate (68.8 – 81.8%); whereas group size and detection probability contributed 13.0 – 23.6% and 5.8 – 7.6% of the variance, respectively.

The reliability of the abundance estimates of khulan from the ground survey data was supported by the highly congruent estimate obtained from the aerial survey, i.e. 25,838 from ground survey compared to 26,969 from the aerial survey in the 75,281 km² overlap area ($z = 0.21, p > 0.83$). Furthermore, the precision (CV = coefficient of variation) of the estimates are very similar: 21% compared to 19%, respectively. Dealing with the problem of gazelle species identification by applying the ratio of gazelle groups ($75/279 = 0.269$) during the ground survey to the total gazelle estimate from the aerial survey (39,112) yields an estimate of 28,598 goitered gazelles in the overlap zone. The estimates of goitered gazelle in the overlap area estimated by the two survey methods are also very comparable: 27,892 from the

ground survey compared to 28,598 from the aerial survey ($z = 0.14, p > 0.88$). The precisions of the estimates are also very similar, 18% compared to 17%.

Discussion

This is the first attempt to produce consecutive population estimates for khulan and goitered gazelle in the Southern Gobi using a standardized survey protocol. Our results confirm that Mongolia's Gobi ecosystem contains the largest population of khulan and goitered gazelle in the world. Our population estimates for khulan are about 2-3 fold larger than a previous estimate in 2009 (B. Lhagvasuren and S. Strindberg, unpubl. data). Although we cannot rule out that very high poaching pressure in the early 2000s reduced the khulan populations in the region (Wingard and Zahler, 2006), we believe that these low estimates were primarily caused by a limited survey effort, and non-standardized survey protocols. Thus the current higher estimates need to be interpreted as reflecting improved monitoring methods rather than a true increase in population owing to conservation interventions.

Our density estimates for khulan and gazelles are in the same order of magnitude but somewhat lower than those recently obtained in Great Gobi "B" Strictly Protected Area in south-western Mongolia (Ransom et al., 2012). Differences in population density and the ratio of khulan to gazelles likely reflect differences in habitat, human land use and protection. In particular, the range of khulan in the south-western Mongolia largely falls into the Great Gobi "B" Strictly Protected Area (~ 9,000 km²), where mining activities are illegal and infrastructure development is minimal. Furthermore, herders and livestock presence is only seasonal, and resource (e.g. water and pasture) availability tends to be more predictable (D. Nandintsetseg unpubl. data).

The precision of wildlife population surveys is generally low, with coefficients of variation (CVs) of around 30% not uncommon (Kaczensky et al., 2015b). In this context, the goitered gazelle estimates, in particular, are extremely good, with CVs below 15%. Even the precision for the khulan estimates, ranging between 18% and 27%, is acceptable. However, improving the precision of population estimates of ungulates in the Southern Gobi would facilitate evaluation of the effectiveness of conservation measures because wide confidence intervals make the detection of trends more challenging. Such evaluations also depend on the magnitude of fluctuations in population size due to various environmental and human associated factors including unfavorable weather conditions (e.g. droughts and harsh winters) and poaching (Kaczensky et al., 2011b; Stubbe et al., 2012).

Power analyses show that it is extremely difficult to detect change, even with a CV of 15%. In 10 years, an exponential decline of 1%, 5% or 10% per year would result in losses of about 10%, 40% or 65% of the population, respectively (Table 3.4). Changes of this magnitude would be difficult to detect, in part because population changes are usually masked by process error; that is, variation in true population size (Ahrestani et al., 2013). Moreover, the demographics of khulan and gazelles in Mongolia are not well studied. This highlights the need for a cautious management approach, as well as triangulation of results (i.e., considering the results from the telemetry studies, habitat models, and khulan carcass surveys) rather than relying solely on the ground survey results to signal a statistically significant decline.

Three components contribute to the variance of the density or abundance estimate, namely the variance associated with the estimation of 1) encounter rate, 2) the detection function, and 3) expected group size (Buckland et al., 2001). The distribution of ungulates in

the study area is closely tied to seasonal variability in resources and anthropogenic disturbances. Thus, not unexpectedly, during the ground-based surveys, for both ungulate species, variance associated with encounter rate and expected group size were the largest contributors to the overall variance in density or abundance. Khulan and goitered gazelle formed smaller groups during the spring relative to the autumn survey, resulting in higher encounter rates particularly for khulan. Consequently, surveys conducted in late spring would likely provide more precise and more accurate population estimates.

When sufficient funds are available, the more costly photo-based aerial surveys (~US\$500,000) have certain advantages over the more cost-effective ground-based surveys (~US\$50,000). Due to the speed of the plane responsive movement is generally less of a problem, although this is clearly a function of survey altitude and the characteristics of the aerial platform. Aerial photographs help to accurately estimate group size, even though species identification issues arose in terms of distinguishing Mongolian gazelles from goitered gazelles. The existence of digital photographs permits tests using independent photo interpreters to ensure that animals are not missed or miscounted, as well as tests for biases caused by changing sightability with time of day, height, and ambient light conditions. The aerial photographs also contain a wealth of ancillary data on vegetation, vehicle tracks, and other variables that may be valuable to future analyses. Our analyses show that the two survey methods produced comparable results for both khulan and goitered gazelle, indicating the ground-based distance sampling methods are able to provide sufficiently reliable and precise estimates. Thus, plains ungulates in the region can continue to be monitored in future by ground-based distance sampling, when funds are limited.

Given the rapid mining and industrial development in the region, the uncertainty surrounding their impact, and that the Southern Gobi supports ~75% of the global khulan population, we recommend the following. Firstly, landscape-scale monitoring of khulan and goitered gazelles should be implemented using the ground-based distance sampling every 2 years. To obtain the best accuracy and precision through increased encounter rates and smaller group sizes, surveys should be conducted at the time of year when the ungulates are most widely distributed throughout the area. Secondly, the aerial surveys should be repeated every 6 or 12 years as they provide an important check on the ground survey estimates and provide valuable ancillary data at the landscape scale. Lastly, complementary studies that assess other vital population parameters, such as recruitment and mortality rates, are needed to provide a more complete picture of the status of these ungulate populations. Having reliable and multiple sources of information will support timely management actions that ensure the survival of khulan and goitered gazelles in the Southern Gobi in the long-term.

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Table 3.1 Results of driving line transect surveys for groups and individuals of ungulates in the Southern Gobi, Mongolia during 2012 – 2015.

Season	Year	# groups	# individuals	Median	Mean \pm SD	Range
Khulan						
Autumn	2012	140	4,941	12.0	35.29 \pm 52.65	1 – 275
Spring	2013	247	3,001	4.0	12.07 \pm 25.71	1 – 250
	2014	192	3,571	5.0	18.59 \pm 34.37	1 – 302
	2015	205	3,095	5.0	15.09 \pm 34.41	1 – 351
	Overall	784	14,608	5.0	20.26 \pm 36.78	1 – 351
Goitered gazelle						
Autumn	2012	234	1,140	3.0	4.89 \pm 4.28	1 – 30
Spring	2013	279	863	2.0	3.10 \pm 3.25	1 – 33
	2014	269	894	2.0	3.34 \pm 3.13	1 – 23
	2015	251	1,058	3.0	4.21 \pm 4.15	1 – 32
	Overall	1,033	3,955	3.0	3.88 \pm 3.70	1 – 33

Table 3.2 Estimates of encounter rates (n/L in groups/km), estimates of expected group size $\hat{E}(s)$, and average groups size (\bar{s}) with their 95% confidence intervals (95% CI) for each species.

Season	Year	n/L	95% CI	$\hat{E}(s)$	95% CI	(\bar{s})	95% CI
Khulan							
Autumn	2012	0.03	0.02 – 0.05	28.96	19.31 – 43.41	27.05	20.59 – 35.53
Spring	2013	0.06	0.04 – 0.08	8.24	6.78 – 10.01	11.21	8.25 – 14.97
	2014	0.04	0.03 – 0.07	12.62	9.32 – 17.08	17.85	13.32 – 23.91
	2015	0.05	0.03 – 0.07	11.01	8.29 – 14.63	14.71	10.78 – 20.05
Goitered gazelle							
Autumn	2012	0.06	0.05 – 0.08	4.23	3.77 – 4.74	4.77	4.23 – 5.37
Spring	2013	0.07	0.05 – 0.09	2.79	2.52 – 3.10	3.13	2.73 – 3.59
	2014	0.06	0.05 – 0.08	3.14	2.78 – 3.54	3.40	3.01 – 3.84
	2015	0.06	0.05 – 0.08	3.45	3.05 – 3.90	4.17	3.65 – 4.75

Table 3.3 Estimates of density of individuals (\hat{D} per km²) and abundance (\hat{N}) with their percent coefficient of variation (%CV), and 95% confidence intervals (95% CI) for each species.

Season	Year	\hat{D}	95% CI	\hat{N}	95% CI	(%CV)
Khulan						
Autumn	2012	0.83	0.52 – 1.29	65,739	40,462 – 106,810	24.92
Spring	2013	0.45	0.31 – 0.67	35,899	22,680 – 40,537	18.47
	2014	0.50	0.30 – 0.86	39,998	25,234 – 42,153	26.81
	2015	0.46	0.27 – 0.78	36,298	21,447 – 61,434	27.08
Goitered gazelle						
Autumn	2012	0.50	0.38 – 0.68	39,602	29,638 – 52,916	14.69
Spring	2013	0.36	0.27 – 0.48	28,462	21,326 – 37,987	14.62
	2014	0.39	0.30 – 0.50	30,744	23,833 – 39,658	12.92
	2015	0.43	0.33 – 0.55	33,627	26,090 – 43,340	12.88

Table 3.4 Example of a 6-, 12-, or 18-year monitoring program where surveys take place every two years. The power (the probability of being able to detect a certain change – with values in bold indicating acceptable power) is given for a range of different positive or negative changes (% change/year) in population size with different potential precision expressed as percent coefficient of variation (%CV) associated with the survey estimate. We assume exponential population changes and a significance level of 10% or 15%, the latter being more conservative for management purposes.

Total	Annual	%CV = 15		%CV = 20		%CV = 25	
Monitoring	Population						
Duration	Change	$\alpha = 10\%$	$\alpha = 15\%$	$\alpha = 10\%$	$\alpha = 15\%$	$\alpha = 10\%$	$\alpha = 15\%$
(years)	(%)						
6	1	10	15	10	15	10	15
	5	14	21	12	18	12	17
	10	23	33	18	26	15	23
	-1	10	15	10	15	10	15
	-5	14	21	12	19	12	17
	-10	25	36	19	29	16	24
12	1	14	20	12	18	11	17
	5	73	82	52	63	40	50
	10	100	100	95	98	84	91
	-1	13	19	12	18	11	17
	-5	76	85	55	66	42	53
	-10	100	100	97	99	90	95
18	1	24	32	18	25	15	22
	5	100	100	96	98	87	92
	10	100	100	100	100	100	100
	-1	23	31	18	24	15	21
	-5	100	100	97	99	89	94
	-10	100	100	100	100	100	100

Figure 3.1 A map of the study area and the line transects surveyed during 2012-2015, in the Southern Gobi, Mongolia.

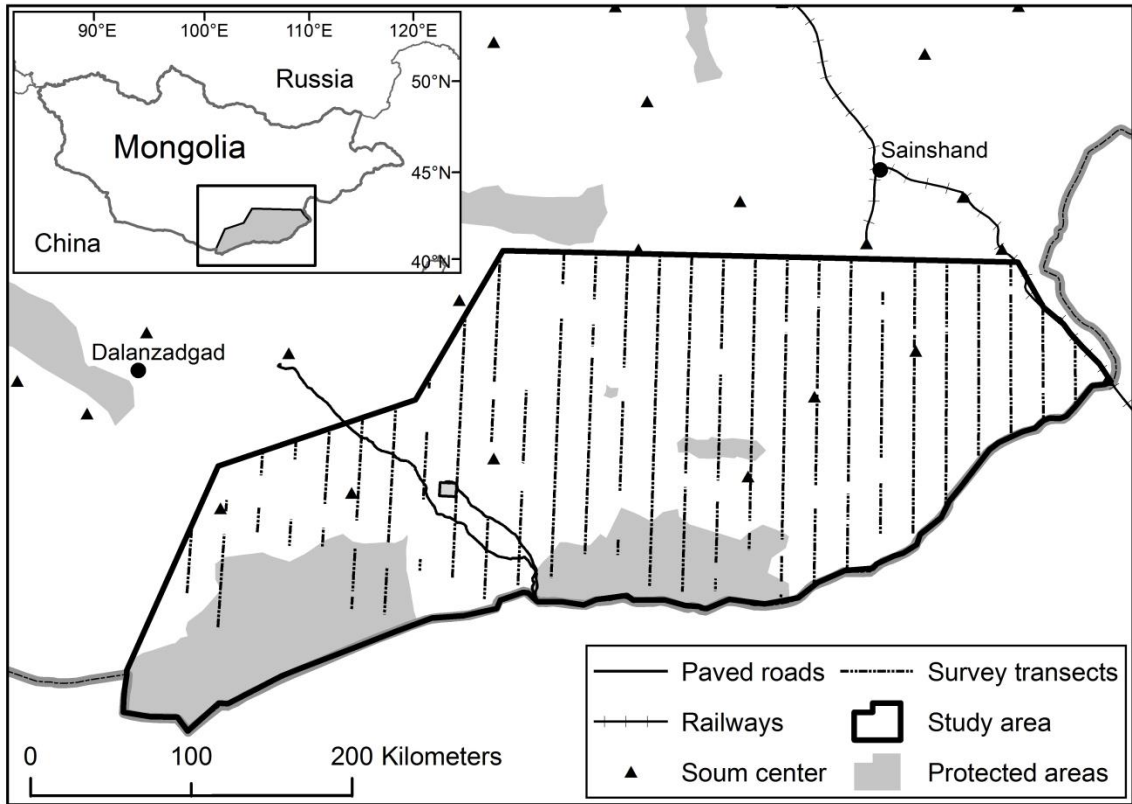


Figure 3.2 Overlap area between the aerial and ground-based surveys conducted in 2013.

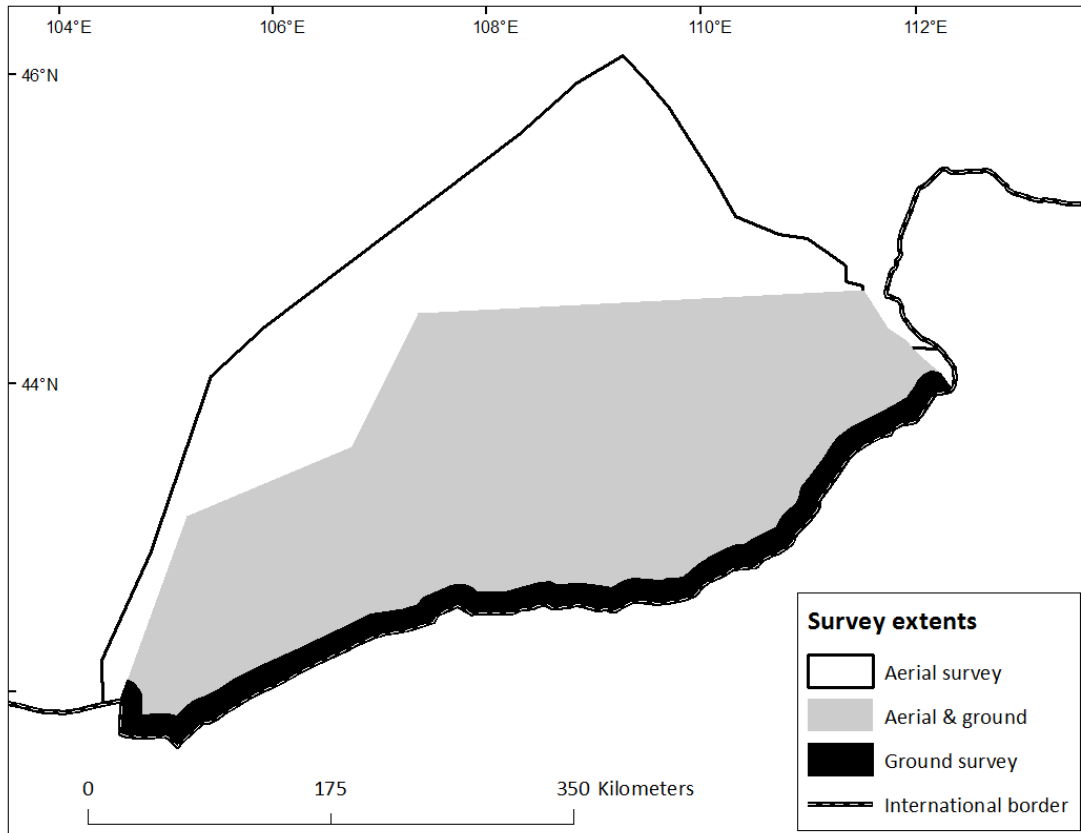


Figure 3.3 Grouping patterns of khulan (A) and goitered gazelles (B) observed during ground surveys in 2012 – 2015 in Southern Gobi, Mongolia.

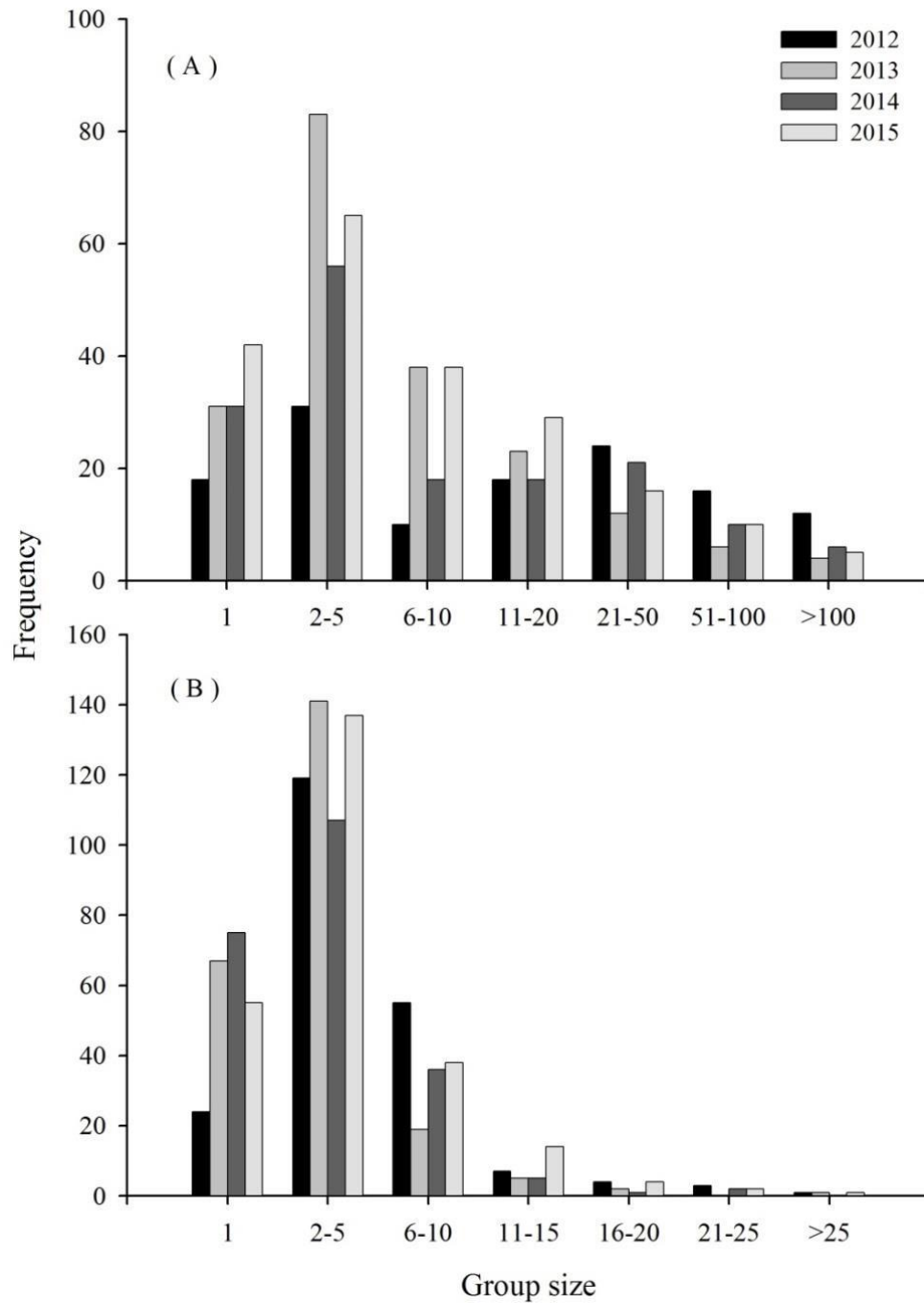
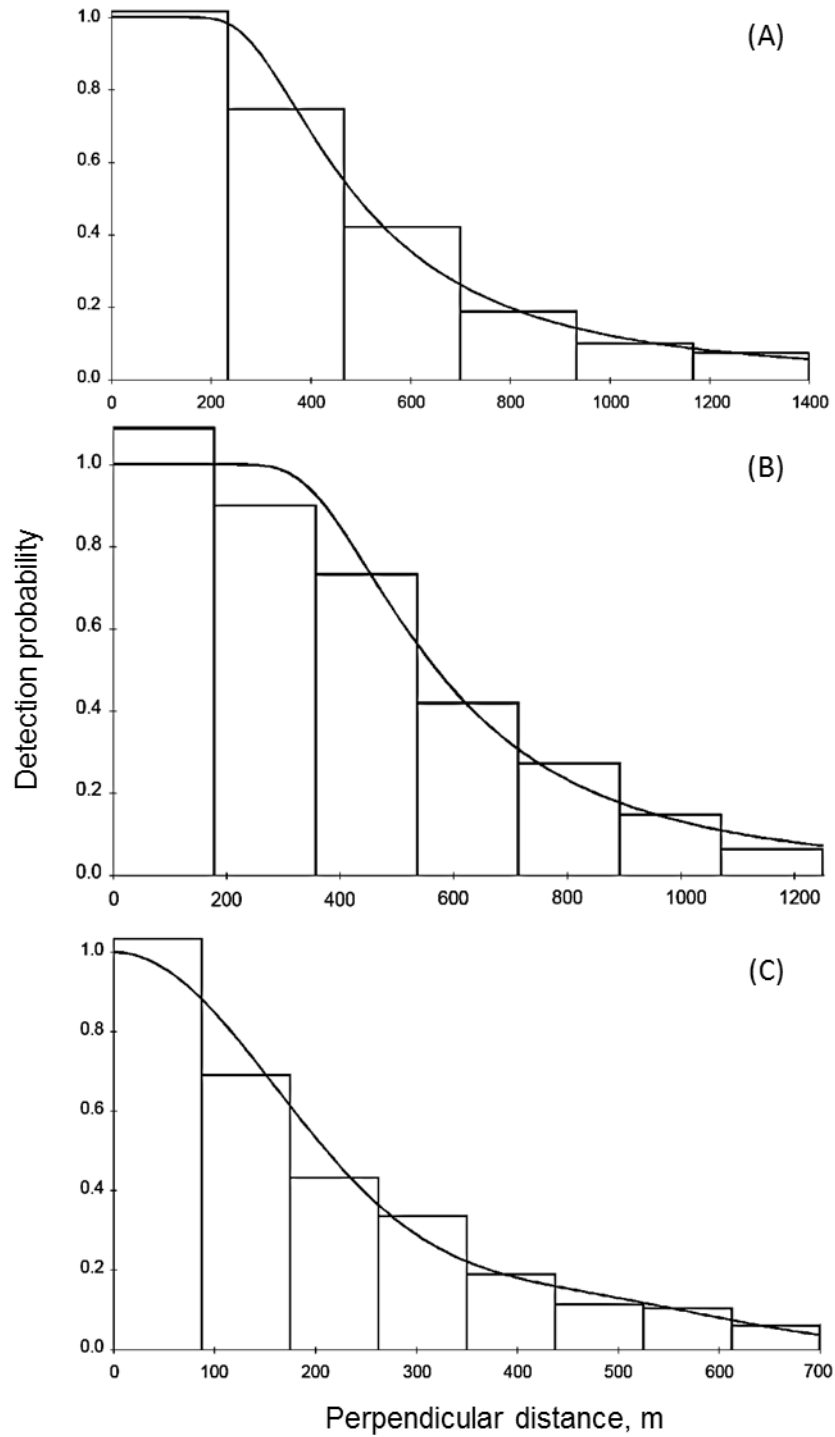


Figure 3.4 The detection probability function fit the pooled data during 2012-2014 (A) and Spring 2015 (B) for khulan, and the detection function fit the pooled data from 4 surveys for goitered gazelle (C).



CHAPTER 4

DISTURBANCE ASSOCIATED WITH HUMAN ACTIVITIES HAVE A NEGATIVE INFLUENCE ON SUITABLE HABITAT OF UNGULATES IN THE MONGOLIAN GOBI

Abstract

The Southern Gobi of Mongolia is an iconic ungulate stronghold that supports the world's largest populations of Asiatic wild ass (or khulan – *Equus hemionus*) and goitered gazelle (*Gazella subgutturosa*). A growing human population, intensifying exploitation of natural resources, and the development of infrastructure in the region place increasing pressure on these species and their habitats. During 2012-2015, we studied factors influencing the distribution of these two ungulate species in the Southern Gobi to better inform management. We built Generalized Linear Mixed Models (GLMMs) to predict the spatial distribution of both species using environmental and human-associated factors and validated these models using independent telemetry data for each species. The GLMMs suggest that the probability of ungulate presence decreased with increasing human influence and increased in areas with intermediate values of elevation and Normalized Difference Vegetation Index (except for goitered gazelle). Habitat models predicted between 45-55% of the study area to be suitable for khulan and between 50-55% suitable for goitered gazelles. Models for both species had good predictive power, as nearly 90% of khulan and 100% of goitered gazelle telemetry locations from separate data sets were found within the predicted preferred areas. Notably, human-associated factors were more important than environmental

variables in explaining the seasonal distribution of the two species. Our findings provide a critical baseline on the key drivers of their distribution and can be used to plan mitigation measures and reduce the impacts of development.

Introduction

The Mongolia's Southern Gobi desert is among the world's largest and most intact grasslands, and thus is of global importance (Batsaikhan et al., 2014). This region supports a unique assemblage of native wildlife, including the largest populations of Asiatic wild ass (or khulan, *Equus hemionus*) and goitered (or black-tailed *Gazella subgutturosa*) gazelle in the world (Mallon, 2008; Kaczensky et al., 2015). For both species, poaching is primary driver of population declines throughout their range (Mallon and Zhigang, 2009; Stubbe et al., 2012), although habitat loss and fragmentation across the species' range may also be important (Clark et al., 2006; Ito et al., 2013a; Batsaikhan et al., 2014). The khulan is categorized as Near Threatened (Kaczensky et al., 2015), while goitered gazelles are listed as Vulnerable on the IUCN Red List (Mallon, 2008).

The desert ecosystem is characterized by seasonal extremes of heat and cold, unpredictable precipitation, and accompanying low and dramatically variable pasture productivity (von Wehrden et al., 2012). The overall sparse environment with tremendous interannual variability in high-quality pasture resulted in the development of a nomadic ungulate-grassland system. Well-adapted ungulate species survived because of their ability to move long-distances to find suitable habitat (Olson et al., 2010; Kaczensky et al., 2011a). Conservation of this highly dynamic system is particularly challenging because of the large areas required to provide enough pasture for viable populations (Ito et al., 2013b).

The Southern Gobi also is rich in mineral deposits (World Bank, 2006), and a number of mining-related development and infrastructure projects are underway or planned (Walton, 2010; Batsaikhan et al., 2014). As the level of extractive industry developments expand across the region, they disrupt migratory movements, fragment habitat, and cause direct or indirect habitat loss (Ito et al., 2008; Kaczensky et al., 2011a). However, little is known about the impacts of mining development and operations on khulan and goitered gazelles and their habitats. Consequently, determining key variables influencing distribution of and the amount of potential habitat available to khulan and goitered gazelle in the vast landscape of the Southern Gobi is crucial to developing mitigation measures and planning landscape-level conservation strategies (Kaczensky et al., 2008; Mallon and Zhigang, 2009).

In this study, we present the results from extensive ground-based surveys conducted in the Southern Gobi during 2012–2015 to provide an assessment of factors affecting the distribution of khulan and goitered gazelles. We were particularly interested in determining whether environmental or human associated factors are the main drivers for the distribution of both species. We used spatially explicit models to quantify the amount and spatiotemporal heterogeneity of ungulate habitats to identify and prioritize areas for conservation and management. These results are important for understanding of what the current drivers of distribution for these species.

Material and Methods

We conducted our study across a 98,216-km² area in Mongolia's Southern Gobi (Figure 3.1), where elevation ranges from 683 m to 1,884 m. Average annual precipitation is 150 mm in the southeast part of the study area, but considerably less (≤ 100 mm) toward the

north and west. The average annual temperature is around 5⁰C, but daily means may reach 40⁰C in summer and drop to -35⁰C in winter. Vegetation is sparse and in many areas is dominated by drought-adapted central Asian desert species, particularly *Artemisia spp.*, *Allium spp.*, *Stipa spp.*, and *Anabasis brevifolia* (von Wehrden et al., 2009). There are a few tree species, including saxaul (*Haloxylon ammodendron*) and elm *Ulmus pumila*, which are confined to the river valleys and basins. Surface water is restricted to springs, some of which are permanent, primarily located in or near mountain ranges. Khulan are capable of accessing water by digging in dry riverbeds where the ground water table is high, thereby also creating temporary water points for other wildlife, including gazelles. In addition to the two study ungulates, there are Mongolian gazelle (*Procapra gutturosa*), argali sheep (*Ovis ammon*) and ibex (*Capra sibirica*) present. Mammalian carnivores include the wolf (*Canis lupus*), lynx (*Lynx lynx*), red fox (*Vulpes vulpes*), and corsac fox (*Vulpes corsac*).

The study area is bordered to the south and east by a fenced border with China, and the Trans Mongolian Railroad corridor, which create nearly impermeable barriers to ungulate movement. In addition, two parallel paved roads connecting major mining activities and the Chinese border crossings are present to the west (Figure 3.1). There are four protected areas which comprise approximately 20% of the study area (e.g., 18,949 km²). Human populations in the region are concentrated in *soums* (villages/towns), with the rural population primarily consisting of semi-nomadic livestock herders. The region is at the center of the cashmere goat industry in Mongolia, and livestock products generate the main income of local herders (Berger et al., 2013).

Data collection

Each year during 2012-2015, we surveyed a total of 64 transect lines (transect length range = 4.25–204.78 km), with a total of 3,464 km of survey effort across the 98,216-km² area (Figure 3.1). The transect lines were randomly located and systematically spaced 20 km apart using the Distance software (Strindberg et al., 2004; Thomas et al., 2010). The survey was conducted using distance sampling line transect approaches, in accordance with guidelines recommended by Buckland et al. (2001). The ground-based surveys were conducted in the first half of October in 2012, and in late May and early June in 2013 and 2014; each survey lasted 14–18 days. Transects were driven at a rate of 20–30 km per hour during daylight hours. During each survey, observers scanned the area in front of them and out to 90° on either side. When groups of khulan and goitered gazelles were detected, the location and group size were recorded, using a global positioning system (GPS), binoculars, and spotting scopes. Locations of other features of interest (e.g. human settlements, households, surface water) encountered along the survey transects were also recorded.

Habitat modelling

To develop a habitat suitability model for each species, the entire length of each transects driven was divided into 724 5x5 km blocks. We then calculated presence/ absence of ungulates in each block to derive a binary response variable. Predictor variables used in the spatial modelling included the Normalized Difference Vegetation Index (NDVI), elevation, slope, distribution of households, human disturbance, and distance to the nearest surface water. Of these predictor variables, NDVI and distribution of households vary among years. We acquired NDVI data from the MODIS (Moderate-resolution imaging

spectroradiometer) sensor on board the TERRA satellite. For each survey period, we obtained a 16-day (May 25 - Jun 09 for Spring and Sep 22 - Oct 06 for Autumn) NDVI composite in 250-m resolution (<http://reverb.echo.nasa.gov>). We calculated the mean NDVI value for each block using Neighborhood function in ArcGIS version 10.2. Elevation and slope data were averaged and extracted for the each block, as well, based on Digital Elevation Models obtained from Shuttle Radar Topography Mission (<http://srtm.usgs.gov>) with 90 m resolution. To estimate the broader effect of households on the distribution of ungulates, we used their presence or absence in each block, using data collected during the ground survey. To measure cumulative human impacts on ungulate distribution, we used the human disturbance data layer created by The Nature Conservancy for the Southern Gobi (Heiner et al., 2013). This layer was created using a wide range of human associated factors, including road and railroad density, population centers and associated area of impact, existing mines and existing petroleum development and infrastructure. We determined the spatial distribution of surface water using different sources such as our own observations during the ground surveys, other field work, and the geodatabase of a mining company in the region (i.e., Oyu Tolgoi mine). However, many water sources in our study area are ephemeral and our water layer probably is incomplete when considering the wider landscape.

We used binomial Generalized Linear Mixed Models (GLMMs) to predict ungulate distribution in relation to our set of environmental and human influence variables (Hedley and Buckland, 2004; Marques and Buckland, 2004). We quantified the collinearity among the environmental and human influence covariates using Pearson's correlation. All the variables were included in the spatial modelling as none showed a strong correlation (Figure 4.1). To eliminate sample asymmetry (e.g., more absent than present data) and balance

statistical analysis, we randomly subsampled the absence blocks to equal the number of presence samples in each survey. We used 'z-score' standardization to give all continuous predictor variables a mean of 0 and a standard deviation of 1 to allow the magnitude of coefficient estimates to be compared across variables. The GLMMs were fitted using the library 'lme4' (Bates et al., 2014) in the R statistical software (R Development Core Team, 2014). The year of each survey was included as a random term in the GLMM models to account for potential variability between years. The square terms of the continuous variables were included in the GLMM models to consider whether the ungulates showed a preference of intermediate variable values (e.g. Mueller et al., 2008; Kaczensky et al., 2014). Additionally, we explicitly modelled spatial autocorrelation (Augustin et al., 1996) by including as an autocovariate the number of neighboring blocks where the ungulate species did occur (Figure 4.2).

Final spatial models were selected on the basis of minimum AIC (Burnham and Anderson, 2002). The relative importance of variables explaining distribution of ungulates was evaluated using hierarchical variance partitioning within the R library 'hier.part' (Walsh and MacNally, 2013), which examines all model combinations jointly to identify average influences of predictor variables rather than just considering the single best model (MacNally, 2002).

Habitat variability

We created predictive habitat surface layers which assign a probability as to the presence or absence of khulan and goitered gazelle. These surface layers were based on parameter estimates of the unscaled variables from reduced spatial GLMMs that excluded the

autocovariate term. Although probabilities are generally more informative, thresholds are a helpful tool in conservation management and for simple and applied assessments (e.g., Mueller et al., 2008). We used a 0.5 probability threshold to classify predicted probability values into ungulate presence/absence areas in the four season data sets. Spatiotemporal heterogeneities in ungulate habitats were examined by overlapping predicted suitable habitat across four season datasets for the two species. We then qualitatively compared the proportion of available habitats with selected habitats.

Model validation

We validated predictive performance of the GLMMs for the two species using independent data set collected from individuals fitted with GPS collars. We had a total of 8,638 GPS locations from 18 khulan (Mean \pm SD, 479.9 ± 87.8) and 1,051 GPS locations from 4 goitered gazelles (262.8 ± 10.5), for model evaluation that matched the ground survey periods in 2014 and 2015, respectively. We calculated the mean of all probability surface values corresponding to actual locations of the collared animals. To test whether this mean was significantly higher than expected by chance, we simulated 1,000 random toroidal shifts (Fortin and Dale, 2005) of the ungulate movement paths within the study area. For each shifted point pattern we calculated the mean of the extracted prediction surface values. We determined the significance of our model by determining how many of the simulated patterns had a higher mean than the mean calculated from actual ungulate locations. The extent of habitat utilized by the collared goitered gazelles during the period when the ground survey completed in 2015 was only 3,454 km² (e.g. <5% of the study area), in part due to small

sample size. Thus, we did not simulate the random shifts for the collared gazelle as the movement data insufficiently represents the study area.

Results

On basis of minimum AIC, the top-ranked model included the covariates NDVI, elevation, presence of households, and human disturbance (Table 4.1). NDVI and elevation emerged as significant terms in the model with their second-order polynomials, indicating the preference for areas associated with intermediate values of these variables by khulan (Table 4.1; Figure 4.3a). The model also suggested probability of khulan presence decreased with increasing human disturbance and aggregations of households. Slope and proximity to surface water were less important drivers of khulan distribution and did not appear in the top model. With the addition of the autocovariance term, the overall model fit improved somewhat; the AIC of the full model decreased from 2,241 to 2,074 and residual deviance decreased from 2,225 to 2,056 (Table 4.1; Figure 4.3a).

The best model explaining distribution of goitered gazelles included the covariates: disturbance index, presence of households, and the first- and second-order polynomials of elevation (Table 4.2). The probability of goitered gazelle presence decreased with increasing human disturbance and presence of households, and they preferred an intermediate range of elevation (Table 4.2; Figure 4.3b). Surprisingly, both the first- and second-order polynomial of NDVI did not appear in the top model. Similar to the spatial model of khulan, slope and proximity to surface water also were weak predictors explaining the distribution of goitered gazelles. With the addition of the autocovariance term, the AIC of the full model decreased

from 2,733 to 2,728 and residual deviance decreased from 2,721 to 2,714 (Table 4.2); only a slight change in the relative magnitude of estimated coefficients was observed (Figure 4.3b).

The relative importance of the disturbance index (59%) and presence of household (23%) were much greater than other variables for predicting khulan distribution (Figure 4.4). Similarly, disturbance index and household in addition to the elevation appeared to best explain goitered gazelle distribution. Notably, combined effects of human associated factors (82% for khulan and 65% for goitered gazelle) were higher than those for environmental variables in explaining the distribution of the two species.

During the 2012-2015 surveys, on average, 50 and 52% of the study area was delineated as khulan (Range = 45–55%) and goitered gazelle (Range = 50–55%) habitats, respectively (Figure 4.5 and 4.6). When overlapping suitable habitats for both species across four survey seasons, 71% and 60% of the study area was consistently predicted as khulan (69,733 km²) and goitered gazelle (59,055 km²) habitats during 2012-2015. With exclusion of the autumn data, proportion of habitat overlap between spring seasons increased particularly for goitered gazelles; e.g., habitat overlap across three spring surveys were 72% for khulan and 89% for goitered gazelle during 2013-2015. In addition, our model predicted 44% and 57% of the four protected areas as khulan and gazelle habitat, respectively, throughout all four surveys (Figure 4.5 and 4.6).

Using the independent tracking data to test the predictive power of the khulan distribution model we found that in only 50 out of 1,000 ($p = 0.05$) permutations had a higher mean than the mean for the predicted probability values corresponding to the actual khulan locations. About 88% (7,629 of 8,638 locations) of all tracking locations were found within this predicted area (Figure 4.7). The remaining khulan locations in lower suitability areas

were close to the predicted suitable habitat, with an average distance of 2.30 ± 1.99 km (Range = 0.001 – 13.77 km; Figure 4.7). For goitered gazelles, almost all gazelle locations (e.g. 99% of 1,051 locations) were located in the habitat that they were predicted to occur.

Discussion

Spatial distribution of khulan and goitered gazelles was influenced predominantly by human disturbance, the presence of households, and to a lesser degree by elevation preferring areas associated with intermediate values of this variable; for khulan intermediate values of vegetation productivity (NDVI) were an additional factor. The distribution of the two species was negatively correlated with presence of households, similar to the pattern observed for Mongolian gazelle in eastern Mongolia (Olson et al., 2011). Khulan and goitered gazelles are regularly poached (Stubbe et al., 2012) and both species have long flight distances. They may additionally avoid households because of free-ranging livestock guarding dogs (Buuveibaatar et al., 2009), or grazing livestock that likely compete for resources around households (Yoshihara et al., 2008; Sheehy et al., 2010).

Khulan were encountered more frequently in areas associated with intermediate values of NDVI, suggesting a probable quality–quantity or quality-security trade-off in the vegetation being selected. A preference for areas of intermediate NDVI has also been found for other species such as Mongolian gazelles (*Procapra guttorosa*; Mueller et al., 2008), saiga antelope (*Saiga tatarica*; Singh et al., 2010), and wild Bactrian camels (*Camelus ferus*; Kaczensky et al., 2014), comparable species in terms of ecology. Unlike for the khulan, NDVI did not appear in the top model that explained distribution of goitered gazelles. Goitered gazelles are dryland adapted browsers (Clauss et al., 2002), and are likely able to

feed on extremely sparse vegetation. We found very little interannual variation in spatial distribution of goitered gazelles, although there were considerable changes in vegetation biomass within and between seasons (data not shown). Furthermore, where vegetation is very sparse the strong reflection from bare ground swamps the NDVI signal (Huete et al., 2002); which may explain why NDVI was a weak predictor in the spatial model for goitered gazelles.

Human associated factors (household and human disturbance) had greater explanatory power than environmental variables in the spatial models influencing distribution of the two species. Consequently, the growing development and associated increase in the human footprint in the region will inevitably diminish the range available for these nomadic ungulates. Our results suggest that higher household densities, in addition to other human disturbance (e.g., mining, settlements, railroads and roads) can result in large regions becoming unfavorable for both species. For each year during the 2012-2015 period, the spatial models predicted that approximately half the study area (~50,000 km²) is unsuitable habitat for khulan and goitered gazelles, respectively. Further, about 29% and 40% of the study area was never classified as khulan and goitered gazelle habitat, respectively, during any of these four years. It is worth noting that these four years were “normal” years, and that the distribution of suitable habitat may shift considerably in years of droughts or harsh winters (e.g. Kaczensky et al., 2011b). Given that the Southern Gobi accommodates the world’s largest populations of khulan and goitered gazelles, and that the human pressure in the region is increasing there is urgent need to determine and implement conservation measures that ensure ungulate populations and their habitat can proliferate. Potential management solutions could include land-use regulations that limit the number of livestock

herders that can reside in a region and improved law enforcement to reduce poaching pressure. Regional planning and implementation of mine developments and linear infrastructure should also follow a mitigation hierarchy to avoid prime khulan and gazelle habitat, minimize potential disturbances, restore impacted habitats, and offset residual impacts by improving habitats elsewhere.

There was slight spatiotemporal variation in khulan and goitered gazelle habitats among the four surveys. A low degree of spatiotemporal heterogeneity of ungulate habitats in the region may facilitate enhanced conservation planning for these species. Furthermore, our results show that 27% of khulan and 23% of goitered gazelle suitable habitats were located within a protected area network consisting of the four nature reserves. Protected areas can benefit wildlife population by limiting poaching and development, however, effective management on individual protected areas may not necessarily guarantee successful conservation of these ungulates as they need to move across large areas. The telemetry study in the region revealed that khulan only spend a small fraction of their time within the protected area network (Kaczensky et al., 2011a). Consequently, conservation of plain ungulates in the region cannot focus on protected areas alone, but needs to incorporate the surrounding multi-use landscapes (Reading et al., 2006).

Our surveys were carried out during one autumn and three spring (and early summer) seasons; hence distributional data of the two species during summer and winter seasons are missing in our analysis. The winter period is particularly critical for both species, due to the limited food resource, cold temperatures, and deep snow cover, which can increase mortality (Tachiiri et al., 2008; Kaczensky et al., 2011b). There is a need for similar analyses to be conducted for the winter season, but this requires data on ungulate distribution at this time of

year, which are currently lacking. Adequate satellite telemetry could provide an effective way to gather this sort of data, given the logistical difficulties of ground surveys in the winter months. Further efforts therefore should target running habitat suitability analyses for the khulan and goitered gazelles using the tracking data. While the current approach provides a snapshot of the khulan and gazelle populations, the telemetry data allows for dynamic habitat suitability modelling across years, using a small subset of the population.

The results we presented here are important for understanding of what the current drivers of distribution for these species and can inform landscape-scale conservation initiatives in the Southern Gobi. We have used ground-based distance sampling data to model the likelihood of khulan and goitered gazelle presences and tested these models with independent datasets in a spatially explicit manner. This is a fundamental requirement for embarking on a systematic conservation planning process. One potential application of our approach is evaluation of various mitigation efforts such as reductions in human influences or projection of potential impacts from the planned developments in the future.

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Table 4.1 Parameter estimates of the top ranked full (e.g. spatial autocovariance included) and reduced models explaining spatial distribution of khulan in the Southern Gobi, Mongolia.

Coefficient	Full model			Reduced model		
	Estimate	SE	Z	Estimate	SE	Z
Intercept	-2.029	0.108	-18.779***	-1.451	0.107	-13.554***
NDVI	0.168	0.074	2.257*	0.318	0.074	4.268***
NDVI ²	-0.142	0.051	-2.783**	-0.225	0.052	-4.333***
Disturbance	-0.404	0.110	-3.673***	-0.543	0.110	-4.915***
Household	-0.592	0.156	-3.777***	-0.519	0.150	-3.452***
Elevation	0.011	0.073	-0.159.	0.026	0.070	-0.381.
Elevation ²	-0.181	0.063	-2.841**	-0.214	0.061	-3.497***
Autocovariate	0.077	0.059	12.917***			
Model AICc			2073.7			2240.9
Residual deviance			2055.7			2224.9
Random effect (SD)			9.286e-08			0.108
Degrees of freedom			2287			2288

The terms followed by ² denote second-order polynomials.
 Significance codes: '***' 0.001; '**' 0.01; '*' 0.05; '.' 0.1

Table 4.2 Parameter estimates of the top ranked full (e.g. spatial autocovariance included) and reduced models explaining spatial distribution of goitered gazelles in the Southern Gobi, Mongolia.

Coefficient	Full model			Reduced model		
	Estimate	SE	Z	Estimate	SE	Z
Intercept	-1.207	0.078	-15.370***	-1.126	0.071	-15.651***
Disturbance	-0.700	0.107	-6.494***	-0.721	0.107	-6.701***
Elevation	-0.431	0.065	-6.554***	-0.438	0.065	-6.659***
Elevation ²	-0.264	0.057	-4.584***	-0.275	0.057	-4.793***
Household	-0.979	0.148	-6.590***	-0.964	0.148	-6.500***
Autocovariate	0.148	0.055	2.696**			
Model AICc			2728.3			2733.4
Residual deviance			2714.3			2721.4
Random effect (SD)			0.00			2e-07
Degrees of freedom			2829			2890

The terms followed by ² denote second-order polynomials.
 Significance codes: '***' 0.001; '**' 0.01; '.' 0.1

Figure 4.1 Scatterplot between all pairs of continuous variables with a smoothed fitted curve. The pairwise correlation coefficients displayed in the corresponding upper-right panels, with the font size scaled proportionate to the absolute value of the correlation.

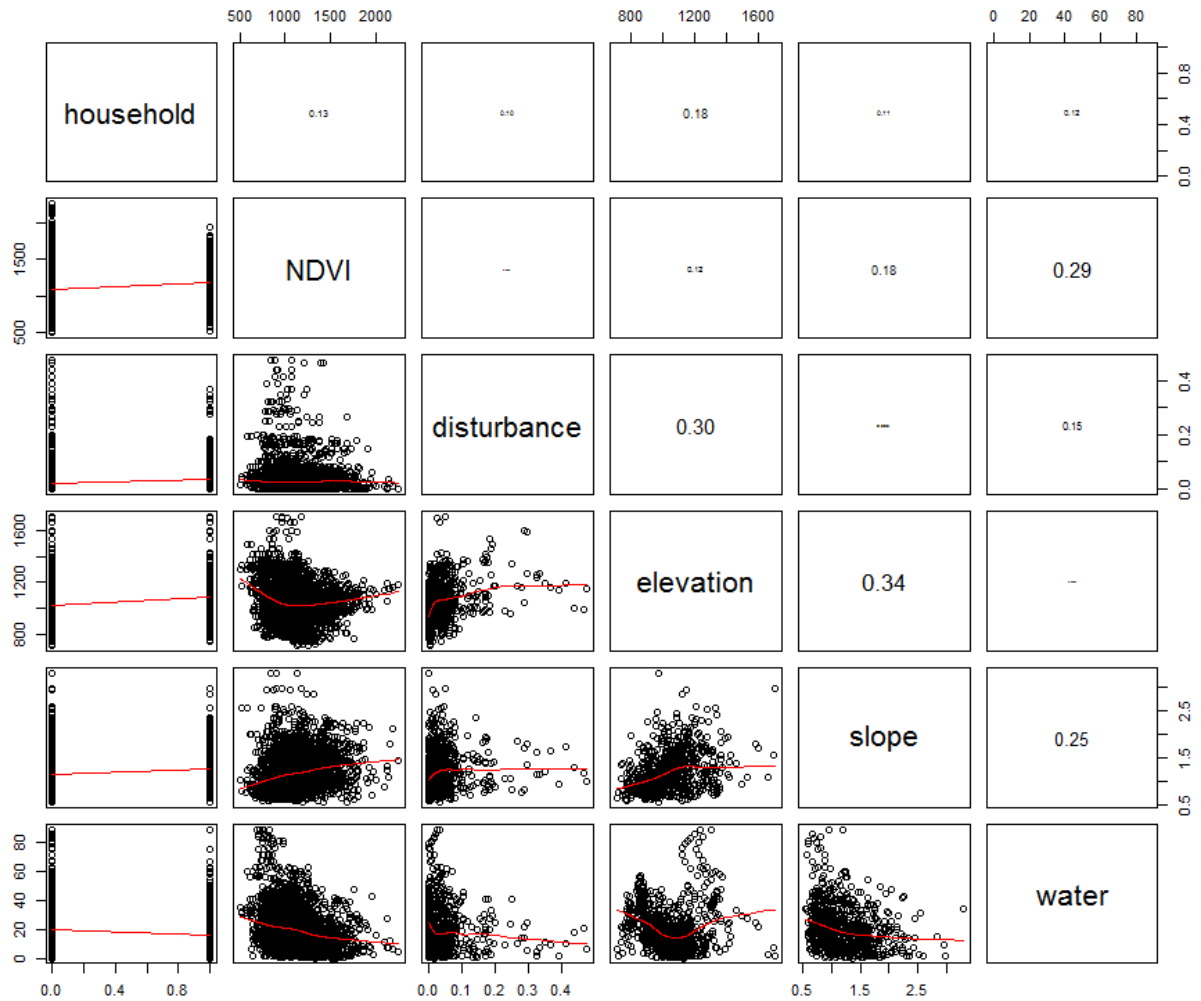


Figure 4.2 Correlograms of residual for the GLMMs to determine spatial distribution of khulan (top) and goitered gazelle (bottom) in Southern Gobi, Mongolia. The range of geographical distance was divided approximately 20 km in each distance class bin. Correlogram of ungulate presence/absence suggests that the presence of khulan and goitered gazelle occurrences have strong positive spatial autocorrelation (e.g. the Moran's I becomes near zero) until around 10 km.

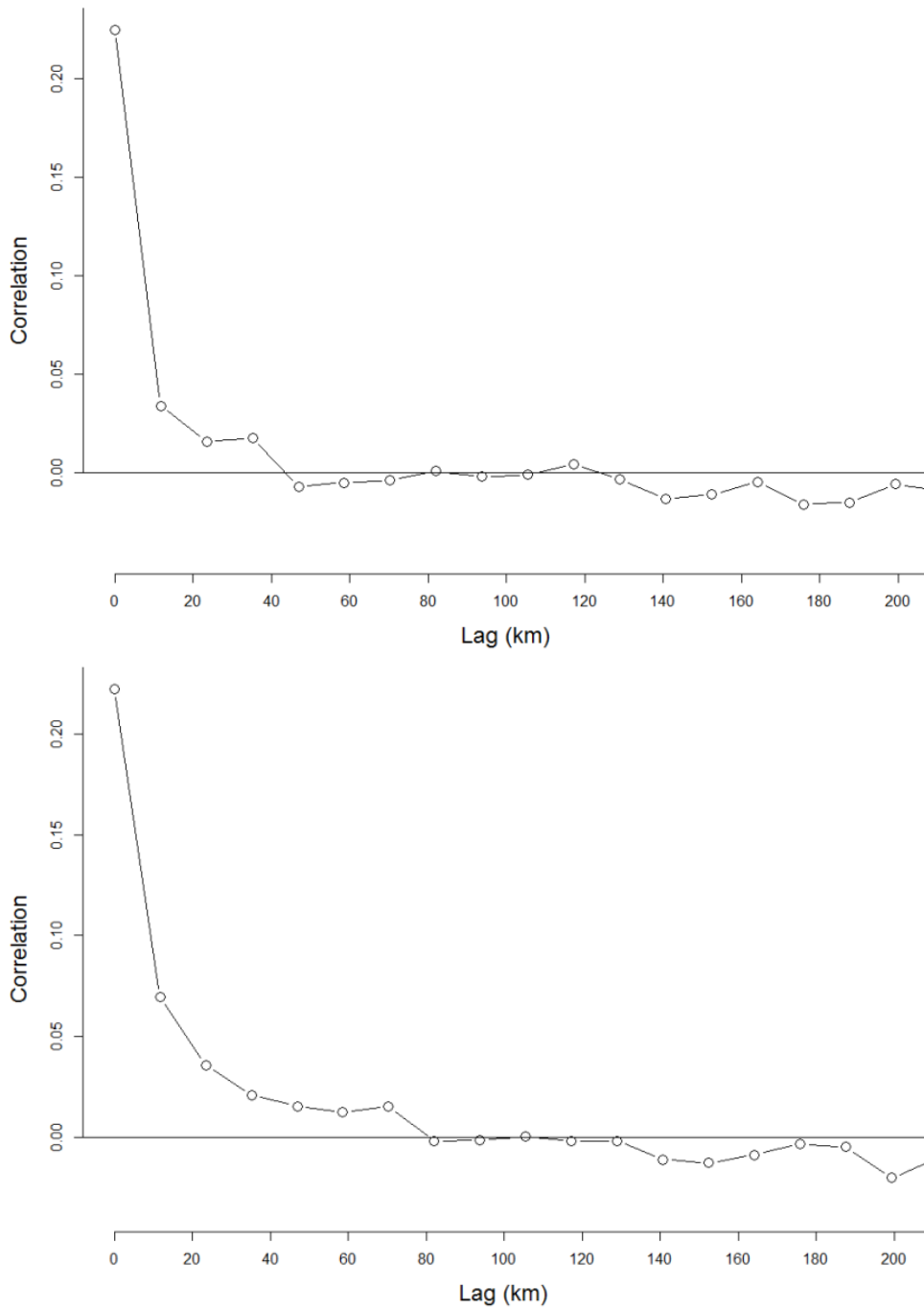


Figure 4.3 Coefficients (\pm SE) of fixed effects of full (i.e. spatial autocovariance included) and reduced generalized linear mixed models using environmental and anthropogenic covariates to predict spatial distribution of (A) khulan and (B) goitered gazelles in the Southern Gobi, Mongolia. All variables were scaled. (NDVI – Normalized Difference Vegetation Index; Elev. – elevation; Disturb. – disturbance index; House. – household; the covariate names suffixed by 2 denote second-order polynomials).

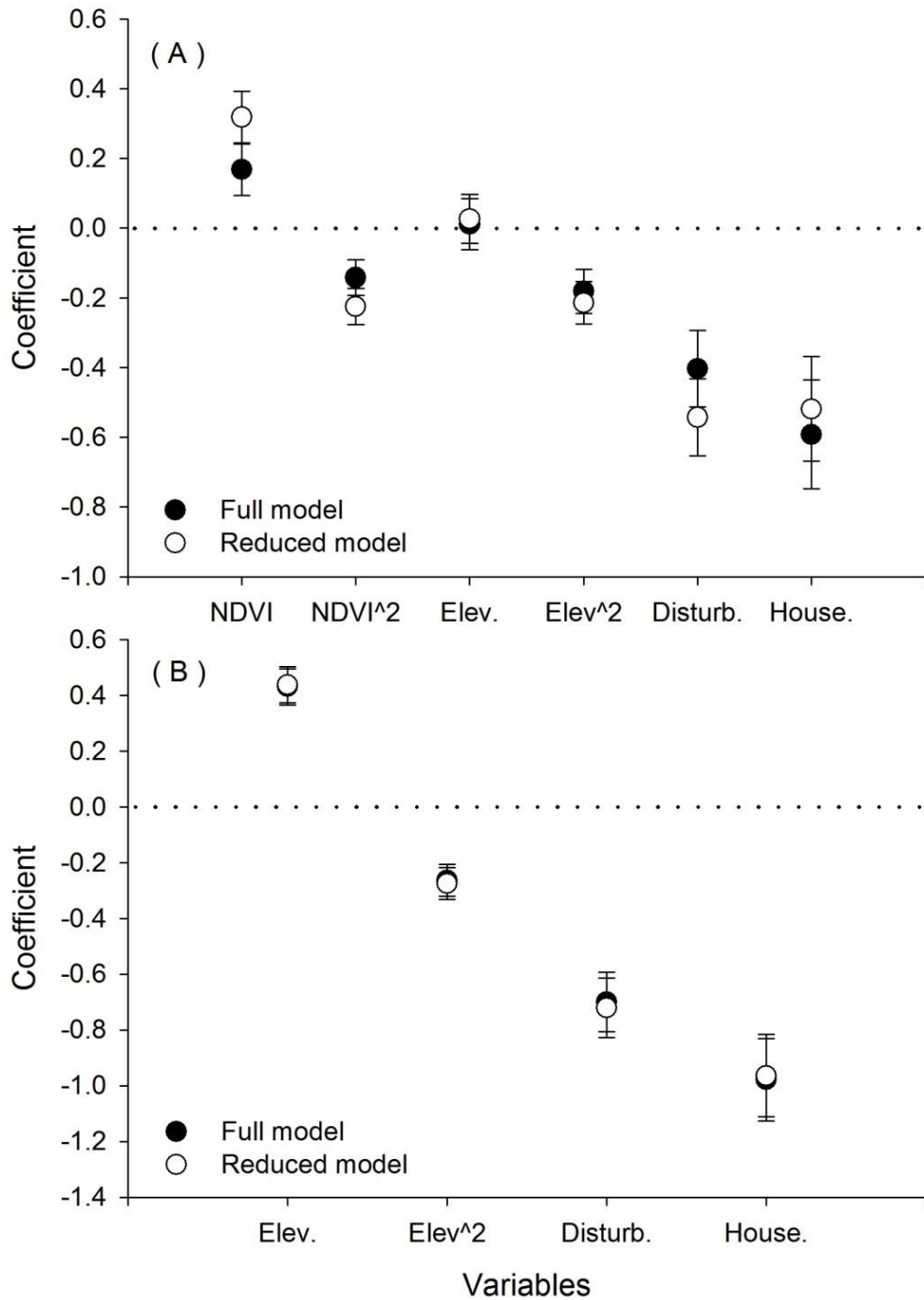


Figure 4.4 Relative importance of predictor variables explaining spatial distribution of khulan and goitered gazelles in relation to environmental and human factors in the Southern Gobi, Mongolia, during 2012-2014 (Disturb. – disturbance index; House. – household).

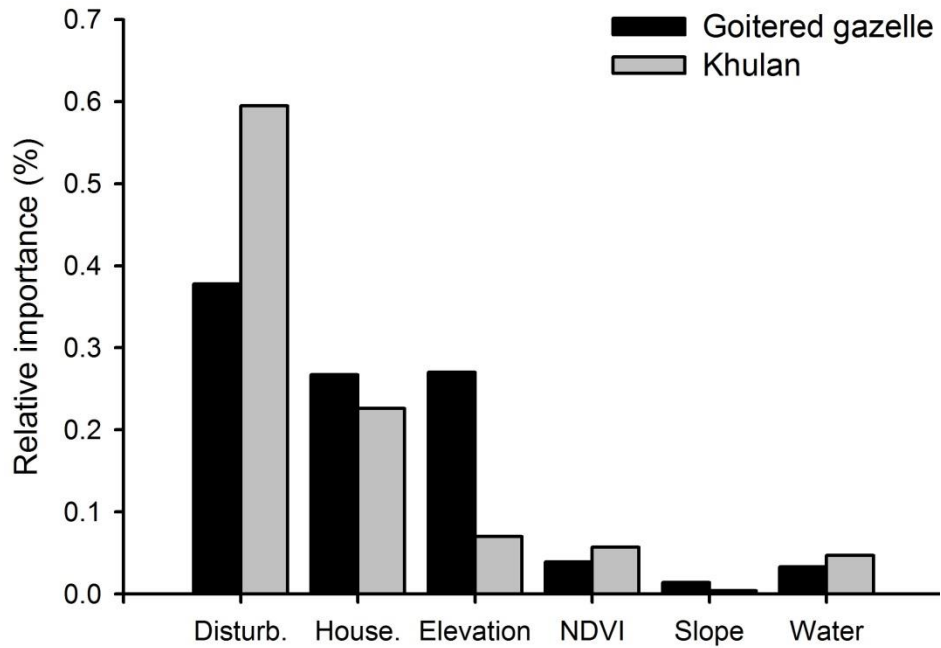


Figure 4.5 Spatially explicit models predicting khulan habitats for autumn 2012 and spring 2013-2015 seasons in Southern Gobi.

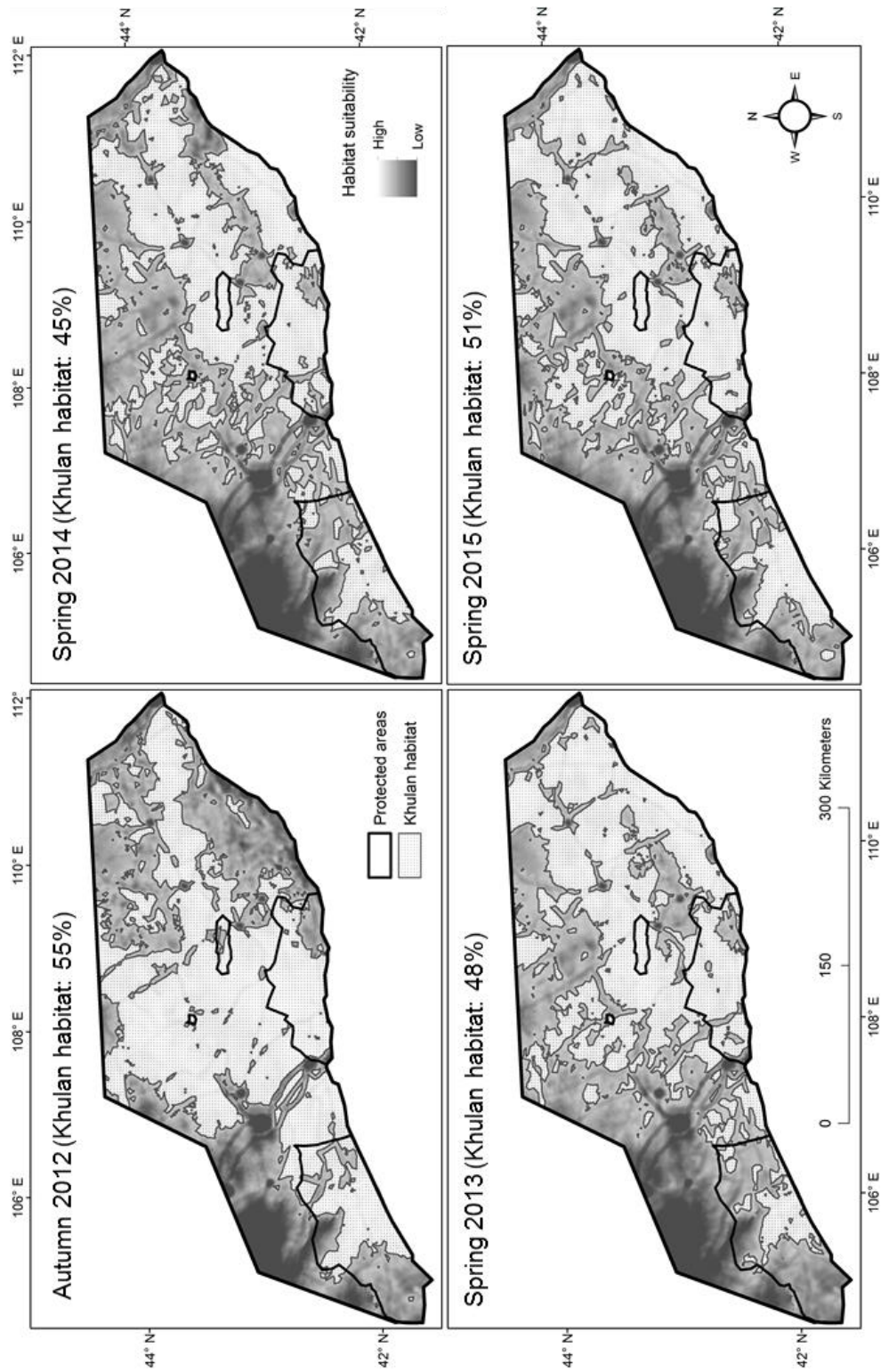


Figure 4.6 Spatially explicit models predicting goitered gazelle habitats for autumn 2012 and spring 2013-2015 in the Southern Gobi.

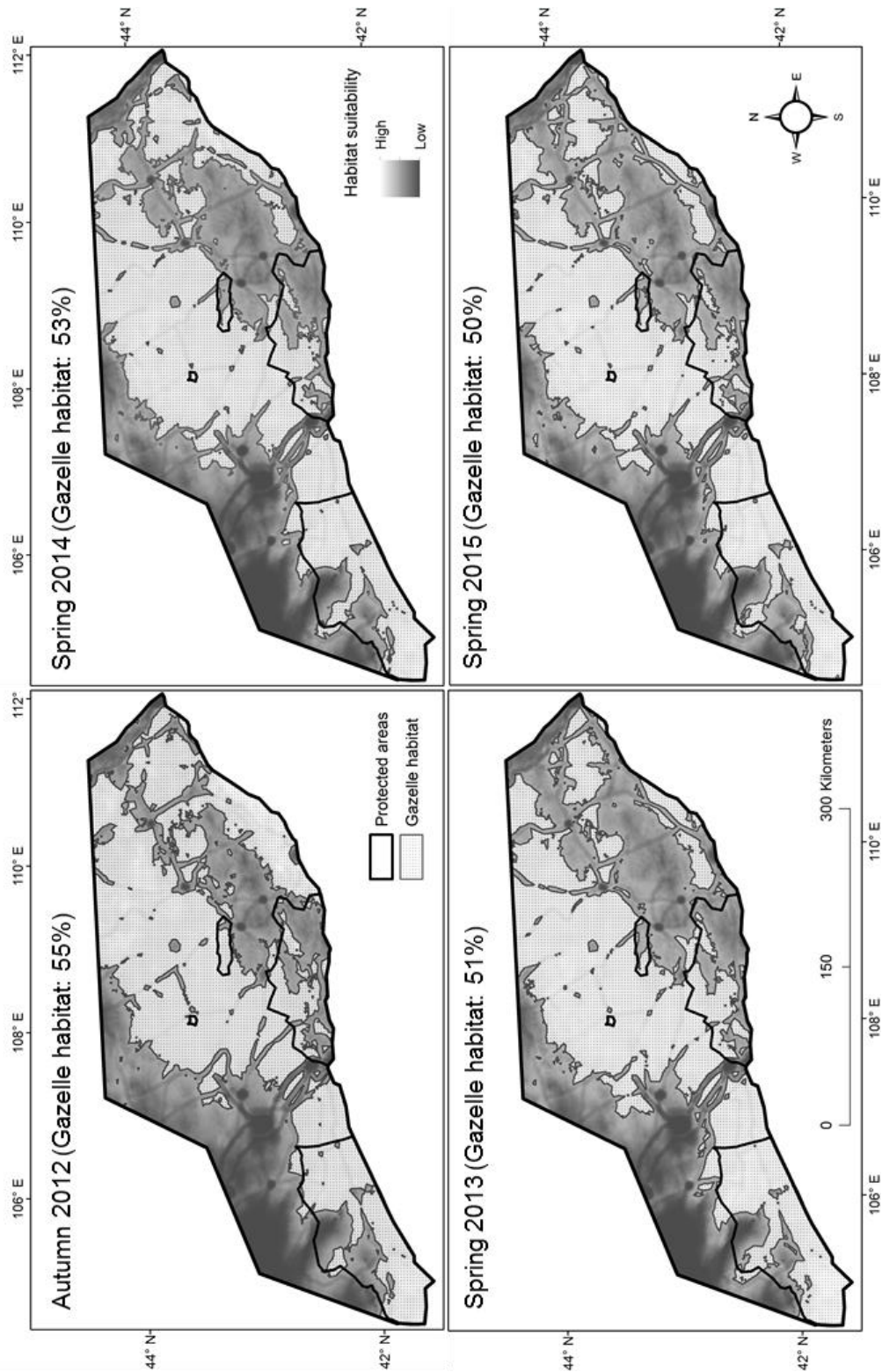
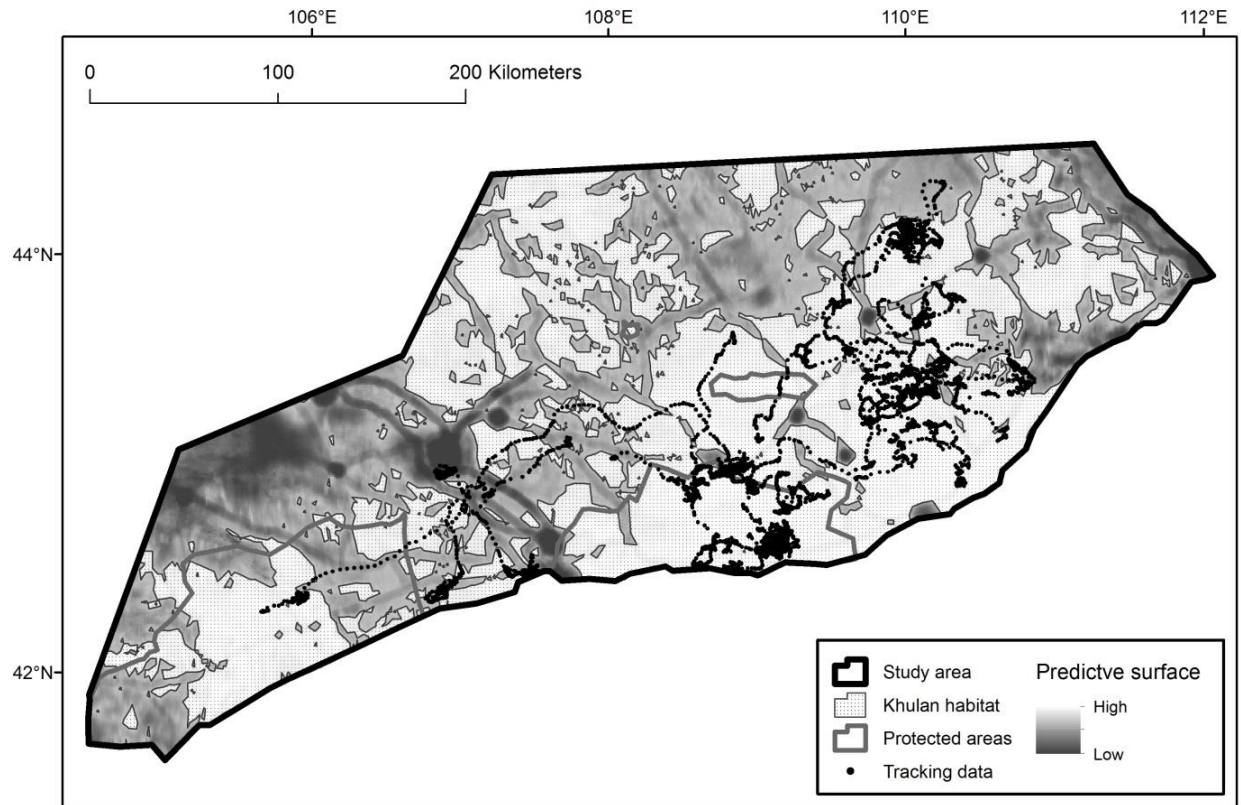


Figure 4.7 A probability surface map predicting khulan habitat (probability threshold of > 0.5) for the Spring 2014 ground survey period, overlaid with independent khulan tracking data for the survey period May 25 – June 10, 2014 in the Southern Gobi, Mongolia.



CHAPTER 5

CONCLUSIONS AND RECOMMENDATIONS

The Mongolian Gobi is one of the most spectacular and important regions in Central Asia (Mallon and Jiang, 2009), comprising the largest area of intact grassland in the world (Batsaikhan et al., 2014). This region supports a large assemblage of migratory plains ungulates including Asiatic wild ass (or khulan - *Equus hemionus*), Mongolian gazelle (*Procapra gutturosa*), goitered gazelle (*Gazella subgutturosa*), the Mongolian saiga (*Saiga tatarica mongolica*), wild Bactrian camel (*Camelus ferus*) and re-introduced Przewalskii horse (*Equus ferus*), some of which are the largest in the world (Reading et al., 2001; Hare, 2008; Mallon, 2008a; Mallon, 2008b; Kaczensky et al., 2015). The current status and ecology of many of these species remains unknown or data deficient. Consequently, there is an urgent need for fundamental ecological surveys of wildlife in the region (Clark et al., 2006; Batsaikhan et al., 2010).

In recent years, a growing human population, expanding exploitation of natural resources, and the development of infrastructure in the region place increasing pressure on these species and their habitats (Kaczensky et al., 2011; Ito et al., 2013). Overhunting and illegal trade increased after the collapse of the Soviet Union in 1991 and the opening of new markets led to the decimation of many populations of these mammals (Wingard and Zahler, 2006). At the same time, pastoral systems and grazing practices have changed (Fernandez-Gimenez, 2001), and overstocking of livestock results in competition for and degradation of pastures (Fernandez-Gimenez, 1999, Sheehy et al., 2010, Berger et al., 2013). Barbed wire

fences along the Chinese-Mongolian border and the Trans Mongolian Railroad affect movements and mortality of khulan and Mongolian gazelles (Ito et al., 2008; Ito et al., 2013). Moreover, a numerous ongoing and planned infrastructure projects such as roads and railways pose threats to these ungulates by impeding of their movements and causing direct and indirect habitat loss (Lkhagvasuren et al., 2012; Batsaikhan et al., 2014).

This dissertation focused on three species of ungulates residing in the Mongolian Gobi such as Mongolian saiga, khulan, and goitered gazelles. The endangered Mongolian saiga is endemic to Mongolia and their range enclosed by a massive Altai Mountains in western Mongolia (Lkhagvasuren et al., 2001; Buuveibaatar, 2011). Global population of saiga antelope plummeted from over 2 million to about 50,000 animals in only two decades primarily due to poaching for horns used in traditional medicine, as well as infectious disease (Milner-Gulland et al., 2001; Mallon, 2016). The khulan is near threatened, with Mongolia's Gobi region holding about 75% of the total global population (Kaczenksy et al., 2015). They move in a nomadic pattern, tracking unpredictable resources in their desert environment across large landscapes (Kaczenksy et al., 2011; Batsaikhan et al., 2014). Goitered gazelle are a medium sized antelope and have a wide range across Middle East and Central Asia (Mallon, 2008b). They are categorized as Vulnerable (Mallon, 2008b), with a decreasing population trend attributed primarily to exploitation, habitat degradation, and human disturbance (Clark et al., 2006).

The calving location selection of saiga antelopes influenced by multiple factors and individual saiga females preferred calving locations that were away from settlements and closer to water sources and avoided steeper slopes. While the variation in saiga group size predominantly determined by Normalized Difference Vegetation Index, predation rate and

season. These results demonstrate that the choice of calving locations and grouping patterns of saiga antelope is driven by both internal and external factors. Understanding which factors influence calving location selection and variation in group size for saiga provides insights to the management of this endangered antelope.

The ground-based ungulate survey in the Southern Gobi of Mongolia provides valuable update on population size of khulan and goitered gazelles in the region. My findings confirm that Mongolia's Gobi desert still supports the largest population of khulan and goitered gazelle in the world. Although variance needs to be reduced and the field protocol for distance sampling improved, the results demonstrate that distance sampling can be an effective technique for monitoring ungulate populations in Mongolian Gobi. As the first study in Mongolia used a standardized survey protocol across a large landscape, this research provides a basis of information on the species, and creates a foundation for the development of more specific management and conservation plans in the country. I suggest the distance sampling approaches should be applied to other large ungulates inhabiting Gobi desert to gather critical baseline data on their population size.

Spatial analyses results suggest disturbance associated with human activities have a negative influence on the amount and distribution of suitable habitats for khulan and goitered gazelles in Southern Gobi. Spatially explicit models also indicate approximately half the study area is unsuitable habitat for khulan and goitered gazelles. The data presented here contain valuable information on factors influencing distribution and calving site selection of the plains ungulates and these results can be used to plan mitigation measures and reduce the impacts of developments. The ways in which we approach this important question can also serve as a model for other systems.

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