Animal Conservation

Animal Conservation. Print ISSN 1367-9430

Effects of ranger stations on predator and prey distribution and abundance in an Iranian steppe landscape

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Keywords
camera trap; line transect; *Ovis vignei*; Persian leopard; poaching; law enforcement; rangers; protected areas.

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Editor: Res Altwegg
Associate Editor: Gurutzeta Guillera-Arroita

Received 12 December 2014; accepted 9 September 2015

doi:10.1111/acv.12240

Abstract

Ranger stations are essential to combat poaching in protected areas and the distance from ranger stations is sometimes used as a proxy for poaching levels and law enforcement intensity. However, the influence of the spatial distribution of ranger stations on wildlife abundance and population structure has rarely been investigated. We evaluated the abundance and distribution of urial sheep *Ovis vignei* and Persian leopard *Panthera pardus saxicolor* in the steppe of Golestan National Park in northeastern Iran. The spatial distribution of these species in regard to anthropogenic (distances to ranger stations, villages and park border) and environmental variables (distance to water resources, average slope and normalized difference vegetation index) was assessed using systematic line transect sampling (186 km) and camera trapping (1150 trap nights). The studied steppe area is divided into three management zones differing in the number of ranger stations and their position with respect to park boundaries. The results of multivariate analyses highlighted that the distance to ranger stations was negatively related to the size and density of urial clusters and the most important variable in explaining urial distribution. Moreover, the distance to park borders influenced urial cluster density. Leopard abundance was positively associated with urial density but was less affected by the other variables tested. We found urial densities in the three management zones to range from $0.15 \pm$ SE 0.09 individuals km^{-2} (zone with just one station outside the park) to $21.77 \pm$ SE 7.92 individuals km^{-2} (zone with three stations). Taking into account these results and historical data on ungulate abundance and distribution in these management zones, we conclude that law enforcement from ranger stations has shaped current patterns of ungulate distribution. These results confirm that a good coverage (both numbers and locations) of ranger stations is of central importance for the management of protected areas.

Introduction

Human exploitation can alter density, demography, distribution and behaviour of ungulates (Hay *et al.*, 2007; Jachmann, 2008; Averbeck *et al.*, 2012) and also affects large carnivores, which are highly dependent on their prey (Kilgo, Labisky & Fritzen, 1998; Karanth *et al.*, 2004; Balme, Slotow & Hunter, 2010). Poaching is one of the most eminent threats to wildlife (Gavin, Solomon & Blank, 2010) and rigorous law enforcement is important in tackling poaching pressure (Rowcliffe, de Merode & Cowlishaw, 2004; Hilborn *et al.*, 2006).

Essential park infrastructures related to law enforcement are ranger stations. Such stations, as well as infrastructure for researchers and tourists, can affect density and distribution of illegally harvested species (Campbell *et al.*, 2011; Jenks, Howard & Leimgruber, 2012; N'Goran *et al.*, 2012). The distance to ranger stations is sometimes used as a proxy for hunting pressure or law enforcement (Hunter & Cresswell, 2015). Such an assumption is due to the fact that most patrolling is done up to a certain distance from patrol posts (Plumptre *et al.*, 2014), resulting into better coverage of effectively patrolled areas near those stations, decreasing gradually with distance from the ranger posts (Hunter & Cresswell, 2015).

The establishment of ranger stations is often linked to logistical considerations or touristic requirements (Campbell *et al.*, 2011). Sometimes ranger posts are created at the border or even outside protected areas and that may leave some parts of the protected area effectively unprotected (Dajun *et al.*, 2006; Jenks *et al.*, 2012; N'Goran *et al.*, 2012). Ideally, ranger stations should be located in places where they can deter illegal activities most effectively and where protection of target species is most required (Plumptre *et al.*, 2014). Therefore, analyses of the relationships between the distributions of ranger posts and those of wildlife abundance or population structure (Dajun *et al.*, 2006; Jenks *et al.*, 2012) should be of interest to conservation management.

In this study, we use data from line transect counts and camera trapping in Golestan National Park (GNP), Iran, to test whether urial sheep (*Ovis vignei*; also formerly known as *O. orientalis*; Valdez, 2008; Rezaei *et al.*, 2010) and Persian leopard (*Panthera pardus saxicolor*; Khorozyan, 2008) distribution, abundance and demography are associated with the distance to ranger stations in comparison with other anthropogenic and environmental variables. Assuming that poaching deterrence is higher near ranger stations, we hypothesized that (1) wildlife abundance and distribution would be negatively related to distance from ranger stations and that (2) differing coverage of ranger stations (both numbers and locations) would result into population responses in harvested species.

Materials and methods

Study area

Located in northeastern Iran (Fig. 1), GNP was the first area to be designated as a national park in Iran in 1957. The park is in a mountainous terrain with landscapes ranging from deciduous forest to steppe and arid plains, which have mean annual precipitation of 142 and 866 mm in the east and west, respectively (Akhani, 2005). This UNESCO Biosphere Reserve comprises an area of 874 km^2, with an elevation range of 450 to 2411 m above sea level (Akhani, 2005). GNP is famous for its diverse landscapes and vegetation types and is the reserve holding the largest (sub)population of Persian leopards [$n = 27 \pm$ standard error (SE) 4.61 individuals; Hamidi *et al.*, 2014]. The area is well connected with three buffer protected areas in the east, west and northwest (Fig. 1). Urial sheep is the main ungulate species of the

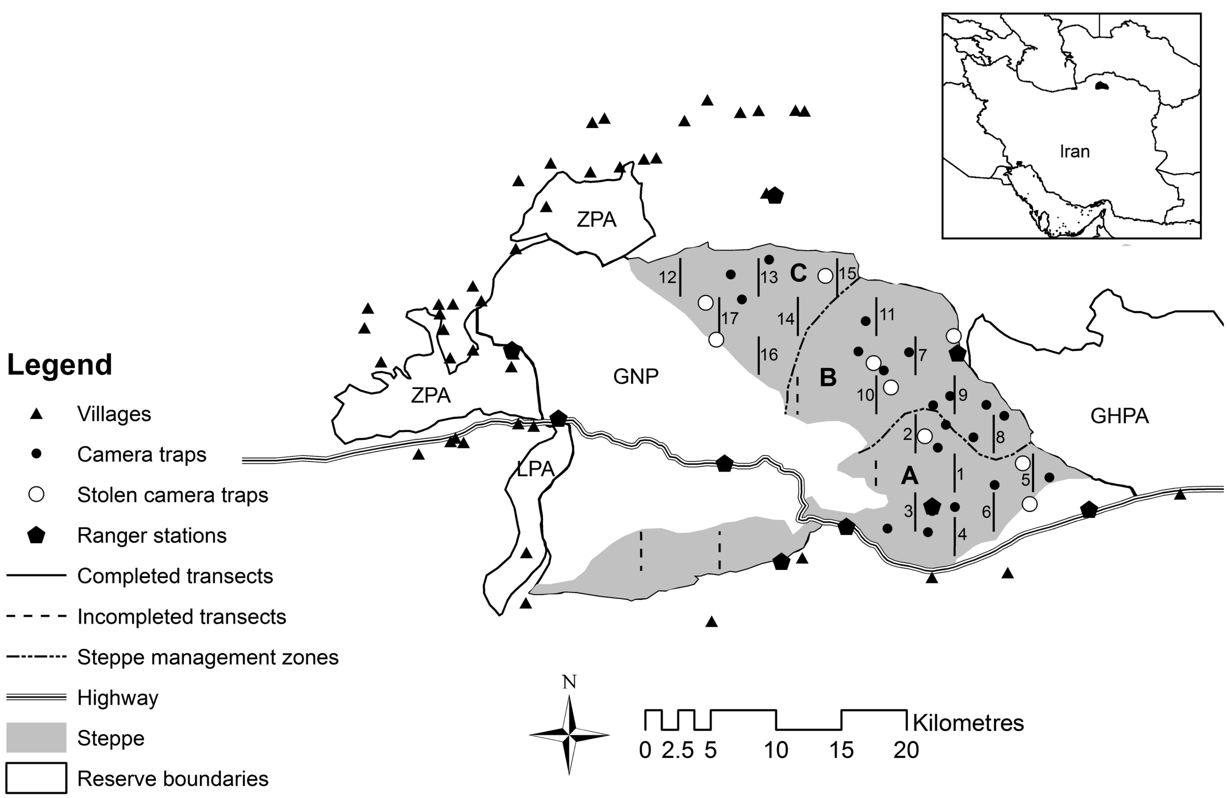

Figure 1 Location of Golestan National Park in the northeast of Iran (black area; inset map) and distribution of line transects (with ID number), camera traps and management zones (A, B and C) in the steppe area, as well as villages and ranger stations of the park (GNP, Golestan National Park; GHPA, Ghorkhod Protected Area; LPA, Loveh Protected Area; ZPA, Zav Protected Area).

steppe region, highly sought after among local poachers and serving as one of the staple prey species of Persian leopard (Decker & Kowalski, 1972; Farhadinia, Moqanaki & Hosseini-Zavarei, 2014; Hamidi *et al.*, 2014). Despite a ban on hunting in all Iranian national parks, insufficient acceptability among local communities and lack of enough conservation measures have led this park to face dramatic ungulate declines in the past decades (Decker & Kowalski, 1972; Kiabi, 1978; Hamidi *et al.*, 2014). The steppe area of the park is divided into three management zones: the zone A has three stations, the highest ranger station density compared to the zones B and C with each having one ranger station (Fig. 1). The station in the zone C is located 5 km outside of the park borders. Over 22 000 inhabitants live in 34 villages surrounding GNP, which are concentrated mostly in the northwest of the park, near the zone C (Fig. 1).

Survey design

We applied distance sampling using line transects to estimate the density of urial sheep (Buckland *et al.*, 2001). After classification of the park into forest, steppe and semi-arid plain landscapes using satellite imagery (Landsat7, 2000) in ArcGIS 10.1 (ESRI, Redlands, CA, USA), the steppe area was stratified for the urial survey (Fig. 1). The approximate total sampling area was 340 km^2. Superimposing a systematic grid cell of 3×3 km over the study area using Hawth's Tools in ArcGIS 9.3 (ESRI), we randomly selected 22 transects throughout the sampling area. Transects were chosen in the north-south direction, which complies with the general density gradient of the species from the core of the park to its boundaries (Buckland *et al.*, 2001). One transect was later removed from the study as it crossed the rugged areas (slope > 45°; 30 m digital elevation map DEM) of the park and therefore was inaccessible. We walked or rode on horseback transects by groups of two to five people including the team members (A.G., A.K.H. and M.S.), park rangers, volunteers and local ex-poachers. To offset high conservation costs and gain local support, in this study, we applied line transect counts using a participatory approach (involving local people), which is increasingly adopted in natural resource management (Holmern, Muya & Røskaft, 2007; Ransom *et al.*, 2012). All groups were equipped with laser rangefinders, binoculars, compasses and handheld GPS units for optimum urial detection and calculation of distances to the transect line. All observers were trained on the use of field protocols and equipment, and each group consisted of at least one ranger or ex-poacher and one volunteer to minimize the observer bias in detection capabilities. Transects were conducted during winter and summer (as the detection probability of urials does not differ in the steppe habitat throughout the year; this study), for 13 fieldwork days (from 22 January to 19 February 2013, from 15 August to 8 September 2013 and from 21 to 24 February 2014). Total survey effort was 186 km. In total, 31 volunteers, 19 park rangers and nine local ex-poachers participated. The total number of sampled transects was 17, as four out of the 21 designed transects were not surveyed because of hard accessibility or deep snow cover during winter (Fig. 1). Most transects ($n = 12$) were surveyed four times; four transects were surveyed three times, and one transect was surveyed twice (Supporting Information). All three management zones are located in suitable urial habitats (Pahlevani, 2004) and were almost proportionately populated by this species with the estimates around 10 000–15 000 individuals during the last systematic population surveys conducted in the park in the 1970s (Decker & Kowalski, 1972; Kiabi, 1978).

Camera trapping

We used camera-trapping data from the leopard population assessment project carried out in GNP by Persian Wildlife Heritage Foundation (Hamidi *et al.*, 2014). In the steppe area, 29 passive camera traps (Deercam, Park Falls, WI, USA) with 35 mm film were installed for 50 consecutive days from January to March 2011 along the main trails and ridge tops of the park, wherever leopard signs were present (Hamidi *et al.*, 2014). One camera trap was installed per station at the height of 40 cm, operating day and night with 1-min delay between consecutive photographs (Hamidi *et al.*, 2014). Nine camera traps were stolen during this period; from six of them, data were available only for 25 days, and from three cameras, no data were retrieved.

Data analysis

We used Distance 6.0 software (Thomas *et al.*, 2010) for estimation of urial sheep density while accounting for detectability and selected the best model based on the lowest Akaike's Information Criterion (AIC) score (Buckland *et al.*, 2001; Waltert *et al.*, 2008). To reduce potential errors in counting and measuring distances to remote observations, a 5% right truncation was performed (Buckland *et al.*, 2001; Thomas *et al.*, 2010). This truncation resulted in omitting observations farther than 587 m distance.

We used the urial cluster density (CD, number of clusters km^{-2}) as the metric of urial abundance (Muchaal & Ngandjui, 1999). We defined the cluster as a group of urials encountered together in a site. The CD was estimated in distance, using transect-specific encounter rate (ER; number of cluster observations per kilometre of transect surveyed) and a global detection probability estimate. The best-fitting detection function was derived from a half-normal key with transect effective strip width of 344.43 m. Since the numbers of urial observations on each transect were not sufficient to fit transect-specific detection functions, the global detection probability estimate was also used for estimation of CD at transect level. We also used the cluster size (CS, individuals) to assess the effects of different variables on the urial demography (Manor & Saltz, 2003; Averbeck *et al.*, 2012). On each transect, we calculated the median of CS and its 95% confidence interval (CI) based on 2000 bootstrap replicates in Flocker 1.1 software (Reiczigel *et al.*, 2008).

Leopards were recognized individually from their unique coat pattern on flanks and limbs (Ghoddousi *et al.*, 2010).

Because of using one camera per station, only photos from either flank (left or right) with the highest numbers of identified individuals were used (Hamidi *et al.*, 2014). We used the rounded mean number of leopard individuals identified from camera traps in the radius of 3 km from the central point of each transect as the metric of leopard abundance. Data from between zero and four camera traps were used for each transect to estimate leopard abundance. No camera traps were available near two transects (ID 14 and 16; Fig. 1), and therefore, these transects were removed from the leopard analysis.

We used the population metrics of urial CD, CS and leopard abundance as the response variables to analyse effects of different variables (Muchaal & Ngandjui, 1999; Hopcraft, Sinclair & Packer, 2005; Averbeck *et al.*, 2012). In order to assess the edge effect (Woodroffe & Ginsberg, 1998) and the impacts of ranger station distribution (Dajun *et al.*, 2006) and human population (Jachmann, 2008; Metzger *et al.*, 2010) on response variables, we considered three anthropogenic predictor variables (closest distances from park border, ranger stations and villages). Additionally, we considered three environmental predictor variables [closest distance to water resources, normalized difference vegetation index (NDVI) and slope] to assess the effects of habitat features on response variables (Ransom *et al.*, 2012). We measured the distance predictor variables (distances to park border, ranger stations, villages and water resources) from the transect centre points and the average NDVI and slopes in the 1.5 km radius of each transect centre point in ArcGIS 10.1. When GNP borders adjoined the neighbouring protected areas (Fig. 1), we measured the nearest distance to the borders with unprotected areas. We measured slopes from the 30 m DEM map of GNP. We obtained NDVI data from a Moderate Resolution Imaging Spectroradiometer (MODIS) scene available from the National Aeronautics and Space Administration, NASA (http://modis.gsfc.nasa.gov/data). We tested the relationship between leopard abundance and urial density (interaction of transect-specific urial CD and CS) on transects.

As two to three rangers per shift (7–10 days) were based in each ranger station of GNP during our research, patrolling intensity was not biased by numbers of rangers. Rangers normally do few daily patrols per shift by vehicle, horse or walking to detect poachers and return to their stations, rarely staying in the field overnight. Despite an effort to widely spread the patrols in each management zone, almost all patrolling trips begin from the stations and the areas around them are the most intensively patrolled. Rangers' patrolling intensity, efficiency and motivations can be variable, but difficult to measure. Because of the lack of spatial records of the patrolling routes, we were not able to measure patrolling intensity and efficiency (Linkie *et al.*, 2015), and therefore, they were assumed to be equally distributed between all the stations. Usually, rangers in GNP rotate their positions between shifts and stations every few months, which mitigates potential biases caused by variations in team performance.

We tested multicollinearity of predictor variables using the collinearity diagnostics test (variance inflation factor; VIF; Kutner, Nachtsheim & Neter, 2004). We identified the influential data points of outliers by regressing the Cook's distance against the centred leverage values for each transect. We visually identified the data points far from the regression line with high Cook's distance and centred leverage values and excluded them from the analysis (Supporting Information). We used two-tailed Z test for comparison of differences in seasonal ER. For assessing the combined effects of environmental and anthropogenic variables, we tested the relationship between the response variables (urial CD and CS and leopard abundance) and all possible combinations of all predictor variables using the generalized linear modelling (GLM) framework. We acknowledge that in the GLM analysis, estimates of variance in our response variables (urial CD, CS and leopard abundance) are not taken into account.

As all villages are located outside the GNP borders, we assumed that a potential interaction between the distance from park border and the distance from villages might exist. Therefore, we included the interaction of these two variables as an additional predictor. We used Tweedie distribution and log link function, which is a highly flexible family of distributions (Jørgensen, 1987). The Tweedie distributions have been only recently used in ecological studies and are applicable for monitoring data, as they account for non-negative data with the spike at zero (Swallow *et al.*, 2016). We used Poisson distribution for the leopard abundance GLM analysis. Because of small sample size, we chose the best model based on second-order corrected AIC (AIC_c) values and their differences (Δ) using the multi-model inference R package 'MuMIn' (Burnham & Anderson, 2002; Barton, 2009). The best models with $\Delta < 2$ were chosen to explain the relationships between the response and predictor variables (Burnham & Anderson, 2002). We applied the Akaike weights to rank models (Burnham & Anderson, 2002). We conducted the analyses using R statistical software 2.15.1 (R Development Core Team, 2012) and SPSS 17.0 (SPSS, 2008).

Results

During the line transect counts, we observed 1981 urials in 70 groups and found no difference in ER between the seasons ($Z = 1.18$, $P = 0.23$). The average ER was 0.37 groups km^{-1} (coefficient of variation CV% = 30.80). Urial clusters varied from one to 191 individuals. The global mean CS after truncation was 29.06 ± SE 3.98 animals/cluster in 66 observations. The global density of urials was estimated at 12.57 individuals km^{-2} (CV% = 35.51; 95% CI = 6.22–25.38) and the population size was estimated at 4275 individuals (95% CI = 2117–8632). By incorporating an effort of 1150 camera trap-nights, we took 35 leopard captures and identified nine leopard individuals based on left flank pictures.

There was no multicollinearity of predictor variables (VIF < 10), but one outlier transect was removed from the urial CD and CS analyses (ID 3; Fig. 1) because of high

Table 1 Summary of the top four generalized linear models (GLM) for urial cluster density, urial cluster size and leopard abundance in Golestan National Park (GNP), Iran

Rank	Model covariates[a]	Parameter[b]				Coefficient (± standard error)				
Urial cluster density (after exclusion of one outlier transect)										
		K	AIC_c	Δ	AIC_w (%)	Intercept	RSt	Bor	NDVI	
1	RSt	3	38.33	0.00[c]	22.32	1.51 ± 0.69	–0.40 ± 0.12			
2	RSt + Bor	4	39.22	0.89[c]	14.32	0.50 ± 0.76	–0.44 ± 0.11	0.27 ± 0.13		
3	NDVI	3	40.95	2.62	6.03	0.66 ± 0.56			–0.06 ± 0.02	
4	RSt + NDVI	4	41.19	2.86	5.35	1.67 ± 0.75	–0.33 ± 0.14		–0.02 ± 0.03	
Urial cluster size (after exclusion of one outlier transect)										
		K	AIC_c	Δ	AIC_w (%)	Intercept	RSt	Bor	Wat	Vil
1	RSt	3	97.68	0.00[c]	31.94	4.40 ± 0.60	–0.38 ± 0.10			
2	RSt + Bor	4	100.66	2.98	7.19	4.03 ± 0.77	–0.40 ± 0.11	0.11 ± 0.13		
3	RSt + Wat	4	100.68	3.00	7.14	4.20 ± 0.69	–0.41 ± 0.11		0.18 ± 0.22	
4	RSt + Vil	4	100.75	3.07	6.88	4.82 ± 0.79	–0.38 ± 0.10			–0.04 ± 0.06
Leopard abundance										
		K	AIC_c	Δ	AIC_w (%)	Intercept	D	Bor	Wat	Vil
1	D	3	42.97	0.00[c]	25.43	–0.24 ± 0.33	0.03 ± 0.01			
2	D + Vil	4	45.15	2.17	8.55	–0.96 ± 0.82	0.03 ± 0.01			0.06 ± 0.06
3	D + Wat	4	45.56	2.58	6.98	–0.65 ± 0.65	0.03 ± 0.01		0.20 ± 0.25	
4	D + Bor	4	45.98	3.01	5.68	–0.05 ± 0.58	0.03 ± 0.01	–0.06 ± 0.15		

[a]Abbreviation of covariates: RSt, shortest distance from transect centre points to ranger stations (km); Bor, shortest distance from transect centre points to GNP borders with unprotected areas (km); NDVI, normalized difference vegetation index in a 1.5 km radius of transect centre points; Wat, shortest distance from transect centre points to water resources (km); Vil, shortest distance from transect centre points to villages; D, urial density on transects (cluster size CS × cluster density CD, individuals km^{-2}).[b]Abbreviation of parameters: K, number of model parameters; AIC_c, second-order corrected Akaike's information criterion; Δ, difference in AIC_c scores between a given model and the best model; AIC_w, Akaike weight (%).[c]Best fitting models with Δ < 2 (Burnham & Anderson, 2002).

Cook's distance and centred leverage values. This transect is located in a different habitat than the rest of the study area on the only high plateau of the park in the transition area between the forest and steppe landscapes. This area was not occupied by urials during our study, possibly because of heavy winds and far distances from available shelters. For urial CD and CS, we used the Tweedie distribution, which revealed that the compound Poisson–Gamma distribution (power = 1.5) best fitted our continuous non-integer non-negative zero-inflated data. Model ranking based on the lowest AIC_c values showed a negative relationship between the distance to ranger stations and urial CD in the top models (see above; Table 1). In the two top models, the distance to ranger stations was significantly inversed to urial CD, while the distance to borders showed a positive relationship only in one of these models (Table 1). A similar influence of the distance to ranger stations was observed in the models of urial CS and other predictor variables (see above; Table 1). The distance to ranger stations was the single best predictor in explaining urial CS. Among the leopard models, urial density was the most influential predictor of its abundance, whereas other predictor variables were not presented among the best models (see above; Table 1).

On average, transect-specific urial density was 145.13 times higher in the zone A with three ranger stations (21.77 ± SE 7.92 individuals km^{-2}) than in the zone C with one station outside of the park (0.15 ± SE 0.09 individuals km^{-2}). The zone B with one station at the boundary of the park (6.90 ± SE 2.97 individuals km^{-2}) had 46 times higher average urial density than the zone C.

Discussion

Several regional studies show that large mammal populations can respond positively to law enforcement over time (Hilborn *et al*., 2006; Jachmann, 2008), but the role of ranger stations and associated patrolling effort in shaping the spatial distribution and population structure of species in predator–prey complexes has been rarely documented (Dajun *et al*., 2006; Jenks *et al*., 2012; Hunter & Cresswell, 2015). A few studies show that high hunting pressure can force sought-after species to move to safer areas (Kilgo *et al*., 1998; Jenks *et al*., 2012), such as conservation stations, monitoring or tourism centres (Dajun *et al*., 2006; Campbell *et al*., 2011; N'Goran *et al*., 2012). Here, we also documented a spatial relationship between the distribution of ungulates and conservation infrastructure. Given the secondary importance of environmental variables in our dataset, we are inclined to relate the distribution and density difference of urials to an increased deterrence of poachers in the stations' vicinity. Urials in GNP live in an open homogenous habitat, mainly consisting of juniper woodland and steppe (*Stipa*, *Artemisia* and *Artemisia*-*Stipa* steppes), with constant accessibility to perennial water resources (Decker & Kowalski, 1972; Pahlevani, 2004; Akhani, 2005). Therefore, a lack of an influence of water availability and vegetation cover on urial distribution is plausible.

According to suggested levels of enforcement to control illegal activities (staff ratio of one ranger per 23.8 km^2 in savanna woodland of southern Africa; Jachmann & Billiouw, 1997), GNP's steppe area has relatively sufficient law enforcement personnel (one ranger per 28.3 km^2 in every shift). However, the recent urial population (4275 individuals, 95% CI = 2117–8632; 2013–2014) is 57.3–71.5% lower in comparison to the earlier estimates in the 1970s in GNP (but see different methodologies used by Decker & Kowalski, 1972 and Kiabi, 1978). These, in combination with our recent data, suggest that the spatial arrangement of ranger stations might have resulted into different levels of poacher deterrence, with the management zone C (Fig. 1) without ranger station inside the park experiencing an especially sharp decline in urial numbers. This suggests that placing ranger stations in or near villages may not reduce poaching from these villages in a significant way (Dajun *et al.*, 2006). In contrast, establishing ranger stations in villages can expose ranger's patrolling efforts to local poachers. The establishment of ranger stations should happen primarily inside protected areas in order to be able to detect non-compliance directly. The decision on the exact location should be based upon considerations of patrolling efficiency, poaching levels or concentrations of most threatened conservation targets (Campbell *et al.*, 2011; Plumptre *et al.*, 2014).

Mapping the distribution of effective law enforcement (e.g. patrol routes and poacher detentions/unit of effort) is beneficial for the identification of patrolling gaps. For a robust assessment of law enforcement distribution, it is recommended to record patrolling routes and intensity, as well as conservation non-compliance and wildlife encounters (Plumptre *et al.*, 2014; Linkie *et al.*, 2015).

Cluster size and sex/age composition of ungulates have been suggested as metrics of human nuisance assessment (Manor & Saltz, 2003; Averbeck *et al.*, 2012). The cluster size of ungulates may indicate poaching pressure when robust population estimates cannot be obtained and is believed to detect subtle human disturbances even before changes in abundance occur (Averbeck *et al.*, 2012). Being a simple demography metric, which can be easily collected by rangers, cluster size may be monitored for such purposes. Also in our study area, the social organization of urials may have varied with hunting pressure as exemplified by the low cluster sizes in areas far from stations. Unfortunately, sex/age ratio monitoring was not possible in our study site as the urial sheep clusters are timid and the differentiation of young males from adult females in remote distances may be affected by observer skills.

In contrast to other studies (Holmern *et al.*, 2007; Jachmann, 2008; Balme *et al.*, 2010; Metzger *et al.*, 2010; N'Goran *et al.*, 2012), which found effects of villages (as sources of poaching) on game species abundance, we found little influence of distance to villages on urial abundance and demography. Urial cluster density, however, was affected by the distance to park borders, which positively contributed to the second best GLM model. We suspect that the lack of a strong relationship with distances to villages and borders is due to the relatively small size of GNP and an ease of access to its core areas by poachers.

We found a strong evidence for the dependence of leopard abundance on urial densities, but we could not detect a direct relationship between leopard abundance and the location of ranger stations. Herbivore abundance is known to be the main determinant of large carnivore distribution at a broad scale, although at a finer scale, prey 'catchability' may be a more important predictor (Hopcraft *et al.*, 2005; Balme, Hunter & Slotow, 2007). Being affected by lack of prey (Henschel *et al.*, 2011), the spatial distribution of carnivores may rely on prey distribution at a spatial scale different than those of poaching and law enforcement (Karanth *et al.*, 2004; Jenks *et al.*, 2012). Studying leopard distribution at a finer scale may reveal other patterns between leopard, prey and poaching in our study area.

With this research, we show that distance to ranger stations may serve as a valuable variable in distribution and abundance of harvested species in steppe areas of northeastern Iran. It also shows that the coverage (both numbers and locations) of ranger stations is of special importance for management planning. In addition, the study shows that the use of population metrics such as cluster size may help to uncover otherwise cryptic illegal behaviour and effects of law enforcement on wildlife.

Acknowledgements

German Academic Exchange Service (DAAD) and Panthera's Kaplan Graduate Award funded the PhD research of A.G. Erasmus Mundus and Alexander von Humboldt Foundation funded M.S. and I.K., respectively. We are grateful for the support of the Iranian Department of Environment, Golestan provincial office of Department of Environment and Golestan National Park management for collaborating with us at every stage of this study. We thank the Persian Wildlife Heritage Foundation for securing access to their camera-trapping data and logistical support during the fieldwork. Turkmen Ecolodge provided generous logistical support throughout this project, which we sincerely appreciate. This work would have not been possible without the support from Golestan National Park's rangers, as well as all the local guides and volunteers who participated in this project. We thank them all. We would like to thank E.J. Milner-Gulland and three anonymous reviewers for their useful comments on the earlier version of the manuscript.

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Supporting information

Additional Supporting Information may be found in the online version of this article at the publisher's web-site:

Fig. S1. The Cook's distance and leverage values of transect-specific urial cluster density and its regression line.
Fig. S2. The Cook's distance and leverage values of transect-specific urial cluster size and its regression line.
Fig. S3. The Cook's distance and leverage values of transect-specific leopard abundance and its regression line.
Table S1. Detail of all generalized linear models (GLM) of urial cluster density, urial cluster size and leopard abundance in Golestan National Park (GNP), Iran.
Table S2. Summary of data on each transect regarding urial population metrics (CD and CS) and leopard abundance and the related anthropogenic and environmental variables.