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Prioritizing livestock grazing right buyouts to safeguard Asiatic cheetahs from extinction

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Abstract

Livestock husbandry exerts major pressures on wildlife across the world. Large carnivores are particularly at risk because they are often killed by pastoralists as a preventive or precautionary response to livestock depredation. Minimizing the overlap between pastures and carnivore habitat can thus be a conservation strategy, but it remains often unclear which pastures should be targeted to maximize conservation benefits given a limited budget. We addressed this question for the last viable population of the Asiatic cheetah (*Acinonyx jubatus venaticus*) in northeastern Iran. By combining species distribution modeling with a spatial prioritization framework, we aimed to identify where grazing right buyouts should take place to reduce cheetah killing by herders and their dogs. We assessed the Asiatic cheetah habitat using species distribution models, highlighting large, contiguous areas that overlap with livestock pastures (5792 km^2, equaling 72% of the total predicted suitable habitat). Subsequently, we used data on the number and distribution of livestock (~47,000 animals in 80 pastures) and applied a spatial prioritization method to identify pastures for grazing right buyouts for a range of budget scenarios (US $100,000–600,000). Pastures selected had a high level of irreplaceability and were generally stable across budget scenarios. Our results provide a novel approach to minimize encounter rates between cheetah and livestock, and thus the mortality risk, for one of the world's most endangered felids and highlight the potential of spatial prioritization as a tool to devise urgent conservation actions.

KEYWORDS

Acinonyx jubatus, action planning, human–carnivore conflict, human–wildlife coexistence, livestock husbandry, spatial conservation prioritization

1 | INTRODUCTION

Large carnivores are important for healthy ecosystems (Hoeks et al., 2020). Standing at the top of the food chain, large carnivores control prey populations, and through that, other elements of food webs (Ripple et al., 2014). The loss of large carnivores can have far reaching, and often unintended consequences on ecosystems and societies, including changes in carbon sequestration or nutrient cycling, the spread of invasive species, disease outbreaks, or rising wildfire risk (Estes et al., 2011). Unfortunately, it is often difficult to reconcile large carnivore conservation with land use in today's human-dominated landscapes (Di Minin et al., 2016; Ripple et al., 2014). Large carnivores naturally occur at low densities and require large tracts of habitats, making them vulnerable to habitat loss and fragmentation (Chapron et al., 2014; Newbold et al., 2020). Similarly, as defaunation progresses, large carnivores suffer from the decimation of prey in many places (Ripple et al., 2014). Large carnivores are often persecuted because of the real or perceived risk they pose to people and livestock (Treves & Karanth, 2003). Fostering the coexistence of large carnivores and people is therefore challenging, yet critically important for carnivore conservation (Boronyak et al., 2020; Chapron & López-Bao, 2016; López-Bao et al., 2017).

Livestock husbandry occupies more than 25% of the Earth's surface (Steinfeld et al., 2006), exerting widespread anthropogenic pressure on large carnivores (Feng et al., 2021; Ripple et al., 2014). Domestic herbivores compete over forage and water resources with wild herbivores, which are the main prey base for large carnivores (Ripple et al., 2015; Schieltz & Rubenstein, 2016). Declines in prey can also happen due to pathogen transmission from livestock as well as poaching by herders (Ripple et al., 2015). Importantly, depredation of livestock by large carnivores often leads to the retaliatory or precautionary killing of large carnivores (Broekhuis et al., 2017; Treves & Karanth, 2003). Finally, herding dogs are frequently used to protect livestock, which can cause considerable and indiscriminate mortality in wild carnivores, as well as their prey (Hughes & Macdonald, 2013; Nayeri et al., 2021). As a result, livestock husbandry often translates into serious threats to carnivores (Soofi et al., 2018), and reducing these pressures could benefit endangered large carnivores in major ways (Pudyatmoko, 2017).

Reducing the negative impacts of livestock husbandry can be addressed by an integrative approach, in which landscapes are shared by carnivores and humans (Carter & Linnell, 2016) or a segregated approach, in which both are separated (Packer et al., 2013). Integration could be achieved by reducing direct pressures in landscapes where herders and carnivores co-occur (e.g., banning of lethal control, prevention measures, compensation payments), whereby the segregated approach aims to spatially separate pastoralists and carnivores (Chapron & López-Bao, 2016; Morehouse & Boyce, 2017). The former is an option in densely settled landscapes, but can be very costly (e.g., European countries pay around 28.5 million Euros per year for damage caused by large carnivores; Bautista et al., 2019), partially suffer from low acceptance rates (Treves et al., 2009), and are not always effective in reducing pressure on carnivores (Eklund et al., 2017). Where possible, reducing the spatial overlap between large carnivore habitat and livestock pastures can minimize conflict (Xiao et al., 2022) and therefore, be a viable and economically attractive alternative. For example, Kuijper et al. (2019) recommend a fine-scale separation between humans and wolves to conserve the species in Europe, or in India and Nepal voluntary human resettlements to mitigate human–wildlife conflict has been successfully implemented (Dhakal et al., 2011; Karanth & Karanth, 2007). Likewise, compensating farmers for setting aside their land for wildlife, as in Karanth (2007), can be a key mechanism for reducing conflict between carnivores and people.

Spatial separation of wildlife and pastoralists can be achieved through relocating livestock from key habitat (Giuliano & Homyack, 2004; Torre et al., 2007). However, deciding on where such relocation could take place is a complex task with several ecological and socioeconomic considerations (Torri, 2011). Pastures vary in size, quality, and location (Molnár, 2012), all reflected in the value these pastures have for livestock herders. Compensation costs for such relocation campaigns can therefore vary hugely across a given landscape. Likewise, the conservation benefit of pastures varies across space, as habitat quality for large carnivores depends on a range of factors that vary themselves. Large carnivores require contiguous habitat, so proximity to other habitat patches must be considered when selecting areas for relocation. As a result, novel approaches for identifying the best places for livestock relocation are highly required.

Spatial prioritization (hereafter: prioritization), is a powerful framework to solve such conservation problems in order to maximize benefits to biodiversity (Wilson et al., 2009). Prioritization has traditionally been used to identify optimal sites for protected areas or their zonation (Kukkala & Moilanen, 2013). Recently, prioritization approaches have expanded to allocate a wider range of conservation actions in landscapes (Wilson et al., 2007), including identifying the best sites for anti-poaching measures (Li et al., 2020), for strategic land acquisitions (Carwardine et al., 2008), or for wildlife-friendly farming

techniques (Chadés et al., 2015). A prioritization framework should hence be useful to identify where a reduction in livestock pressure would benefit large carnivores the most, and where grazing right buyouts should take place given a certain budget. However, to our knowledge, prioritization has not been used for this purpose.

Asiatic cheetahs (*Acinonyx jubatus venaticus*) once roamed over vast areas in central and southern Asia but have disappeared from 98% of their historical range with less than 50 individuals remaining, all of them in Iran (Durant et al., 2017; Khalatbari et al., 2017). The species naturally occurs in arid and semiarid steppes at low densities and requires large home ranges due to low prey availability (Hunter et al., 2007). The last viable population of the species occurs in and around Touran Biosphere Reserve and Miandasht Wildlife Refuge in northeastern Iran (Khalatbari et al., 2017). Livestock pastoralism is widespread in both protected areas and threatens Asiatic cheetahs for all the reasons highlighted above: natural prey competes with livestock over scarce water and fodder, poaching of prey by herders can be substantial, and guarding dogs to protect livestock from wolf (*Canis lupus*) attacks frequently kill cheetahs (Farhadinia et al., 2017). There are also anecdotal reports of the removal and smuggling of cheetah cubs by herders (Farhadinia et al., 2017). Livestock husbandry in this region is thus the most serious threat to Asiatic cheetahs, which are now on the brink of extinction (Durant et al., 2022).

Given this imminent global extinction, reducing pressure from livestock pastoralism on cheetahs is urgently needed. However, many established practices are not useful in this landscape (van Eeden et al., 2018): funding for long-term compensation schemes is currently not available, reducing herding dogs could result in higher livestock losses due to wolf attacks, and separating carnivores from livestock through fencing pastures is not feasible, due to low pasture productivity that requires very large grazing areas and (semi-)nomadic pastoralism. Grazing right buyouts remain as a potentially useful conservation strategy to reduce the spatial overlap between livestock pastures and cheetah habitat. Previous buyout campaigns, initiated in 2017 in Touran Biosphere Reserve resulted in the buyout of eight pastures (7,700 livestock heads, representing 18% of the total livestock; covering 332 km^2, which is less than home range of an Asiatic cheetah; Cheraghi et al., 2018) for a total of US$309,225 as of June 2020. Given that livestock depredation by Asiatic cheetah is rare (Khalatbari et al., 2022), it is unlikely that the removal of livestock negatively affects their survival. Encouragingly, since then no cheetah mortality due to dogs and herders has been reported in the areas that were subject to previous buyout campaign, while the cheetah population has remained stable in these areas (Iranian Department of Environment, unpublished data). These initial results highlight both the feasibility and potential benefits of buyouts. However, optimally identifying priority areas for ramping up buyouts to maximize conservation benefits to Asiatic cheetahs is lacking.

In this study, our overall aim was to use spatial prioritization to identify where grazing right buyouts should take place to better protect Asiatic cheetahs. Specifically, our objectives were:

1. Identifying suitable Asiatic cheetah habitat within and around Touran Biosphere Reserve and Miandasht Wildlife Refuge using species distribution modeling.
2. Develop a spatial prioritization framework to systematically identify priority pastures for grazing right buyouts using a range of budget scenarios.

2 | MATERIALS AND METHODS

2.1 | Study area

Our study area (84,544 km^2) in northeastern Iran (Figure 1) is centered around two protected areas: Miandasht Wildlife Refuge (IUCN category IV, with a section recently upgraded to national park IUCN category II; 844 km^2) and Touran Biosphere Reserve (comprising of a national park, a wildlife refuge and a protected area, with IUCN categories II, IV, and V, respectively; 14,000 km^2). The study area hosts the northern subpopulation of the Asiatic cheetah, which is the only population with recent evidence of reproduction (Khalatbari et al., 2017). We defined the study area to include the surrounding of the protected areas, applying a buffer of 45 km. Although Asiatic cheetahs are known to have long-distance dispersals (Farhadinia et al., 2016), we were interested in predicting cheetah's key habitat in vicinity of these protected areas and therefore, followed the area mapped as suitable habitat by Ahmadi et al. (2017), excluding higher elevations of the Alborz Mountains as well as the Hyrcanian forests, both of which are outside the Asiatic cheetah's natural range (Khalatbari et al., 2017).

The region is characterized by a semiarid to arid climate with annual precipitation between 140 and 400 mm, increasing from south to north and primarily falling from November to May (Fick & Hijmans, 2017). Annual mean temperature varies from 2 to 20°C, with extreme temperatures reaching 40°C in summer and −15°C in winter (Fick & Hijmans, 2017). Elevation in the study area varies from ∼400 to >3700 m a.s.l. and the landscape includes arid plains, hilly terrain, salt deserts,

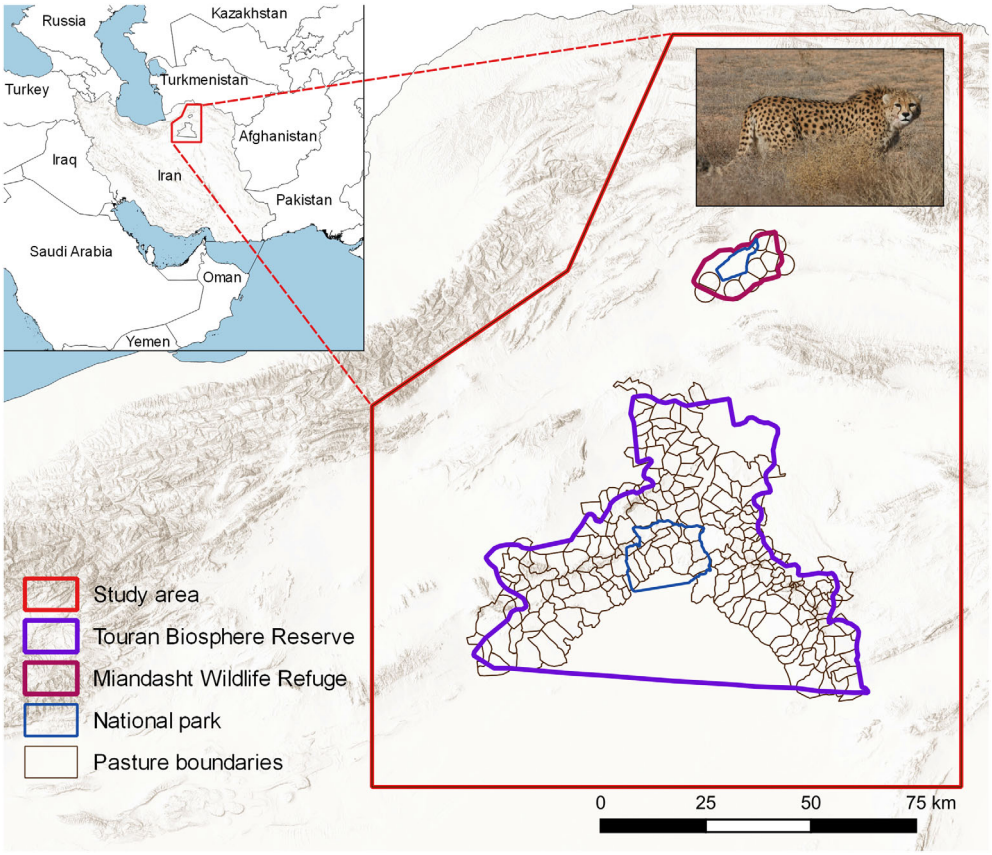

FIGURE 1 Study area in northeastern Iran encompassing the Touran Biosphere Reserve and the Miandasht Wildlife Refuge with pasture boundaries (Photo: Arash Ghoddousi)

sand dunes, dry river beds, and a saline river system (Heshmati, 2007). Natural vegetation is sparse and primarily consists of dwarf scrub vegetations (Heshmati, 2007). Besides the Asiatic cheetah, other large carnivores in the study region include the Persian leopard (*Panthera pardus saxicolor*), striped hyaena (*Hyaena hyaena*), and gray wolf. Wild ungulates include goitered gazelle (*Gazella subgutturosa*), chinkara (*G. bennettii*), urial (*Ovis vignei*), and bezoar goat (*Capra aegagrus*), all of which are the cheetah's prey (Farhadinia et al., 2018).

Livestock pastoralism is a widespread traditional land use in both protected areas and the wider region, with high numbers of livestock (here: sheep and goats) and herding dogs. Touran Biosphere Reserve permits up to 76,000 domestic sheep and goats to graze annually (Khalatbari et al., 2017). In 2017, nearly 40,000 sheep and goats grazed in the Touran Biosphere Reserve together with about 5,000 animals in the Miandasht Wildlife Refuge, accompanied by about 450 and 80 dogs, respectively (Table 1; Abangah Consulting Engineer Company, 2017). While in Miandasht Wildlife Refuge livestock husbandry is performed by the local pastoralists, Touran Biosphere Reserve is predominantly used as winter pasture (November to April) by nomadic livestock owners from other regions (Abangah Consulting Engineer Company, 2015). Pastures are on public land and leased through licenses or long-term contracts. Thus, removing livestock from certain pastures and relocating them does not require evicting people from their own land (Abangah Consulting Engineer Company, 2015).

2.2 | Asiatic cheetah habitat suitability

We identified suitable Asiatic cheetah habitat within the study area using species distribution modeling (Elith & Leathwick, 2009). Previous studies have already assessed the regional (Nazeri et al., 2015) and broad-scale distribution of the Asiatic cheetah habitat (Ahmadi et al., 2017; Moqanaki & Cushman, 2017) or within multispecies studies (Khosravi et al., 2018). Building on these works, we here use species distribution modeling to predict cheetah habitat as a basis for our spatial prioritization exercise. The goal of this exercise was to identify priority pastures for relocation of pastoralists from the key habitat of the Asiatic cheetah. We used a cheetah occurrence dataset comprised of 144 points in and near the Touran Biosphere Reserve (100 points) and the Miandasht Wildlife Refuge (44 points) in a time frame between 2011 and 2017. The data were collected by the Conservation of Asiatic Cheetah Project (CACP) and the Persian Wildlife

TABLE 1 Livestock (in our study region mainly domestic sheep and goats) and herder presence on pastures in the Touran Biosphere Reserve and the Miandasht Wildlife Refuge before the first buyout campaign in 2017, with minimum, maximum, mean, median, and standard deviation (SD) of pasture area in km^2, number of livestock, herders, and dogs (values rounded to two decimal places)

	Number of corrals	Number of pastures	Pastures in km^2	Number of livestock	Number of herders	Numbers of dogs
Touran	89	71	4,350.42	42,313	233	446
Miandasht	23	9	597.46	4,640	38	76
Min			16.51	150	2	2
Max			357.73	1,350	8	16
Mean			61.85	586.91	3.39	6.53
Median			54.51	500	3	5.50
SD			43.94	278.02	1.55	3.52
Total	112	80	4,947.88	46,953	271	522

Heritage Foundation through direct observations and camera traps. We applied spatial filtering of occurrence points to reduce sampling bias (Kramer-Schadt et al., 2013) and chose a resolution of 1×1 km^2, representing a trade-off between large home range size of Asiatic cheetah and retaining enough occurrence points (Ahmadi et al., 2017; Hemami et al., 2018). After the filtering, 56 points remained and served as presence data for the modeling (49 in Touran Biosphere Reserve and seven in the Miandasht Wildlife Refuge; 38 were recorded by camera traps and 18 by direct observation). Based on the ecology of the species, we established predictor variables, including the categories topography, land cover, and anthropogenic pressures (see Table 2 for more detail). We prepared all variables to match our 1×1 km^2 resolution, transformed them to an Albers equal-area projection and applied bilinear interpolation for resampling. Finally, we tested all predictor variables for multicollinearity and, in cases of high correlations $|r| > 0.7$, only one predictor was chosen (Dormann et al., 2017), resulting in six predictor variables: roughness, distance to small rivers, fraction of green vegetation cover, human population density, road density, and distance to ranger stations.

We used the maximum entropy (Maxent) algorithm (Phillips et al., 2006), which outperforms other algorithms for small occurrence datasets and shows a high model performance (Wisz et al., 2008). Maxent can model species distribution with presence-only data by contrasting them with background data (Phillips et al., 2006). We parameterized the model with a maximum of 2,500 iterations using default settings for the convergence threshold (0.00001) and regularization (1.0). We omitted product, threshold, and linear features and only used quadratic and hinge features to avoid overfitting (Phillips & Dudík, 2008). Background points should reflect the range of predictor variable values available to the species (Phillips et al., 2009). Even though cheetahs may have remarkably long dispersals, detected up to 217 km (Farhadinia et al., 2016), such distances do not necessarily indicate the extent of their habitat. As we wanted to predict the habitat of the species within a region that could be considered as one (sub-)population, we used the median movement rate of the cheetahs and checked the sensitivity of this choice in comparison to similar thresholds. Therefore, we restricted the background points to a radius of 50 km around occurrence points, using the median dispersal distance of Asiatic cheetahs (Ahmadi et al., 2017). We allowed one background point per cell, which resulted in a total of 7,464 background points. To examine the sensitivity of the models to the sampling radius, we additionally applied 40 and 60 km radii and compared model performance. Based on 10-fold cross-validation, we assessed model performance using the area under the receiver operating characteristic curve (AUC). We assessed variable importance through percent contribution (i.e., the cumulative contribution of each predictor variable to model gain during model training) and permutation importance (i.e., the decline in training AUC for each variable when their values are randomly permuted). We performed the modeling using the R package *dismo* (version 1.1.4) (Hijmans et al., 2017). We used our final model to map a habitat suitability index (HSI), using the 10-fold cross-validated model. To convert the continuous index to a binary habitat map, we used the threshold maximizing the sum of sensitivity and specificity (maxSSS). This binary habitat suitability map served as input for our spatial prioritization.

2.3 | Spatial prioritization framework

Spatial prioritization of conservation actions requires defining a set of (1) discrete spatial planning units,

TABLE 2 Predictor variables used for the Asiatic cheetah habitat modeling in northeastern Iran

Variables	Expected effect	Rationale
Topography		
Roughness[a]	±	The species favors a relatively moderate level of landscape roughness (Ahmadi et al., 2017; Hemami et al., 2018; Hunter et al., 2007; Moqanaki & Cushman, 2017).
Land cover		
Distance to small rivers (m)[b]	−	Cheraghi et al. (2018) show a preference for shorter distances to water sites.
Fraction of green vegetation cover (FCover) (%)[c]	+	Vegetation indices could act as indirect proxies for cheetah prey (Cheraghi et al., 2018).
Anthropogenic pressures		
Human population density (%)[d]	−	Asiatic cheetahs avoid human-dominated areas (Ahmadi et al., 2017; Cheraghi et al., 2018).
Road density (%)[e]	−	Road infrastructure represents a major threat to the species due to vehicle collisions (Farhadinia et al., 2017; Mohammadi & Kaboli, 2016) and inhibits habitat connectivity (Moqanaki & Cushman, 2017).
Distance to ranger stations (m)[f]	−	Ghoddousi et al. (2016) showed a positive effect of ranger station presence on the distribution of urial *Ovis vignei*, a main prey of cheetah, which could positively influence the cheetah as well.

[a]Shuttle Radar Topography Mission, 2008 (https://srtm.csi.cgiar.org/srtmdata/).
[b]Global River Classification, 2018 (https://www.hydrosheds.org/page/gloric).
[c]Copernicus Global Land Service, 2014–2016 (mean, without summer months; https://land.copernicus.eu/global/products/fcover).
[d]Socioeconomic Data and Applications Center: Gridded Population of the World, 2015 (https://sedac.ciesin.columbia.edu/data/collection/gpw-v4).
[e]Socioeconomic Data and Applications Center: Global Roads Open Access Data Set, 1980 – 2010 (ensemble; https://sedac.ciesin.columbia.edu/data/set/groads-global-roads-open-access-v1).
[f]Conservation of Asiatic Cheetah Project (CACP), 2016/2020.

(2) spatially-differentiated costs for implementing actions, and (3) spatial representation of conservation features (Beyer et al., 2016; Figure 2). In terms of planning units, we mapped individual pastures inside the protected areas that can potentially be the target of grazing right buyouts. For the Touran Biosphere Reserve, we used geospatial data delineating 223 individual pastures from the CACP. For Miandasht Wildlife Refuge, exact pasture boundaries do not exist and we, therefore, mapped pastures as areas within a 5-km distance around nine livestock corrals, based on the daily travel distance of sheep herds (Abangah Consulting Engineer Company, 2015; McGranahan et al., 2018). The remaining areas beyond these pastures form three additional planning units: (1) areas without pastures in the Touran Biosphere Reserve, (2) areas without pastures in the Miandasht Wildlife Refuge, and (3) the area outside protected areas and pastures. Overall, this resulted in 235 planning units. Out of these, we had information on livestock, herders, and dogs for 80 planning units (Table 1; 71 in Touran Biosphere Reserve and all nine in the Miandasht Wildlife Refuge). For the remaining planning units in Touran Biosphere Reserve, livestock information does not exist but many of these pastures are not in use (A. Radman, manager of Touran Biosphere Reserve, *pers. comm.*, 2020).

In terms of potential costs for grazing right buyouts (hereafter: buyout costs), we estimated compensation costs for each of the 80 pastures. Specifically, we estimated buyout costs by calculating (1) pasture value, (2) revenues, and (3) operational expenses associated with herding, using a survey report on pastures by the Abangah Consulting Engineer Company (2015) (Table S2; Figure S1). Unlike the previous campaign, which negotiated with lower prices but faster payments, for our model we used the official price of US$70 per livestock (sheep and goats) head to estimate the pasture value based on the respective livestock density. Revenues represented earnings (from selling dairy and nondairy products and excess livestock) per pasture stated by the herders. Operational expenses included payments for water, fodder, shepherds, herding dogs, disposal of carcasses, transportation, and depredation costs due to livestock losses to large carnivores (mainly wolves). For all pastures without corral

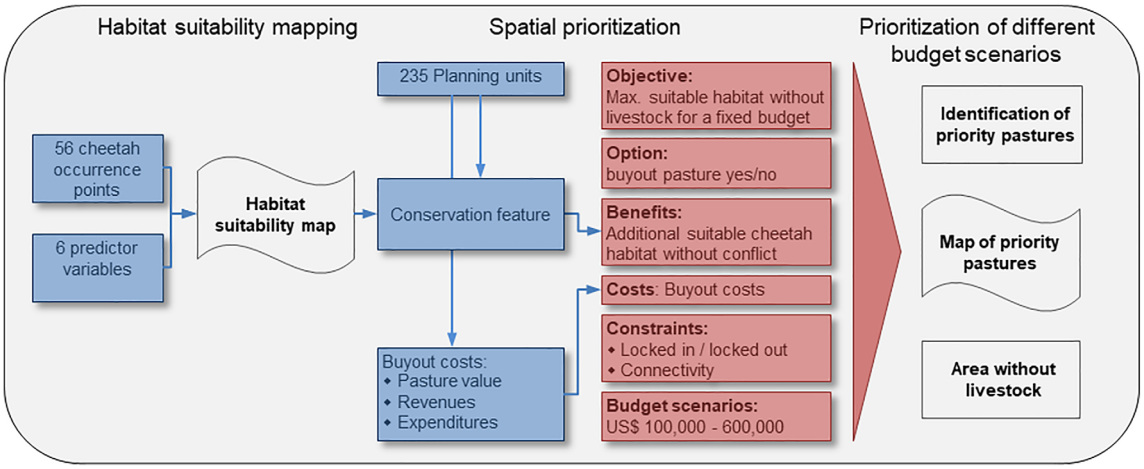

FIGURE 2 Flowchart of the overall procedure of our spatial prioritization framework. Our objective was to maximize the Asiatic cheetah habitat without livestock for a fixed budget by selecting priority pastures for the buyout. The cheetah habitat suitability values served as the conservation feature, which together with the cost data were consolidated per planning unit

data and non-pasture areas, we set the costs to zero and removed or kept the pastures according to their status (already bought pastures or non-pasture area) applying the locked in/locked out constraints (see below). Finally, in terms of conservation features, we used the share of high-quality cheetah habitat per planning unit, based on the binary habitat suitability map.

The overall goal of our prioritization framework was to lessen livestock pastoralism pressure on the Asiatic cheetah habitat by grazing rights buyouts as our conservation action, optimally allocating the buyouts. Thus, our conservation problem included the objective to maximize the Asiatic cheetah habitat without livestock husbandry for a range of fixed budgets, leaning on the assumption that only limited budgets are available for conservation measures. This means the decision in our framework was to select a pasture as a buyout pasture or not. The benefit of this action would be additional suitable cheetah habitat free of livestock disturbances. The costs represent the costs of grazing rights per pasture. For the model setup, we used the "maximum utility objective," able to generate solutions with fixed budgets. To include or exclude planning units from the solution, we used "lock in" and "lock out" constraints. We locked in pastures that already have been bought out, as well as pasture-free areas within the protected areas, as these areas were already considered to be free of livestock. We locked out areas outside the protected areas and pastures without cost data, since these areas were not available for buyout. As an additional constraint, we included connectivity, meaning the prioritization should preferentially select planning units for a buyout that border pastures already free of livestock. Thus, we penalized every exposed boundary of a solution, relative to their length using a penalty value of 0.001 and an edge factor of 0.5. We implemented the prioritization framework using the Gurobi Optimizer (version 9.0.3) (Gurobi Optimization, 2020) within the R package *prioritzR* (version 5.0.2) (Hanson et al., 2020). Within the Gurobi argument, we specified an optimality gap of zero. Thus, Gurobi finds optimal solutions based on integer linear programming, which minimizes or maximizes an objective function with respect to our constraints.

To identify areas for grazing right buyouts, we used our prioritization framework and problem formulation, and assessed six budget scenarios, ranging from US $100,000 to US$600,000, in increments of US$100,000. We provided additional detail for the US$300,000 budget scenario as this scenario roughly represents the budget of the previous buyout campaigns. We solved the conservation problem for all scenarios and accumulated how often each pasture was selected in the six model runs. We then calculated the livestock-free area gained in suitable cheetah habitat within different scenarios. Additionally, to provide insights into the relative importance of the selected buyout pastures, we created a portfolio of solutions containing 1,000 iterations allowing for 10% variation in the optimal solution and computed the selection frequencies per planning unit for the US$300,000 scenario.

3 | RESULTS

3.1 | Asiatic cheetah habitat suitability

The final Maxent model showed an AUC value of 0.90, based on a 10-fold average cross-validation. Our sensitivity analysis with different background sampling strategies demonstrated that model performance variation was

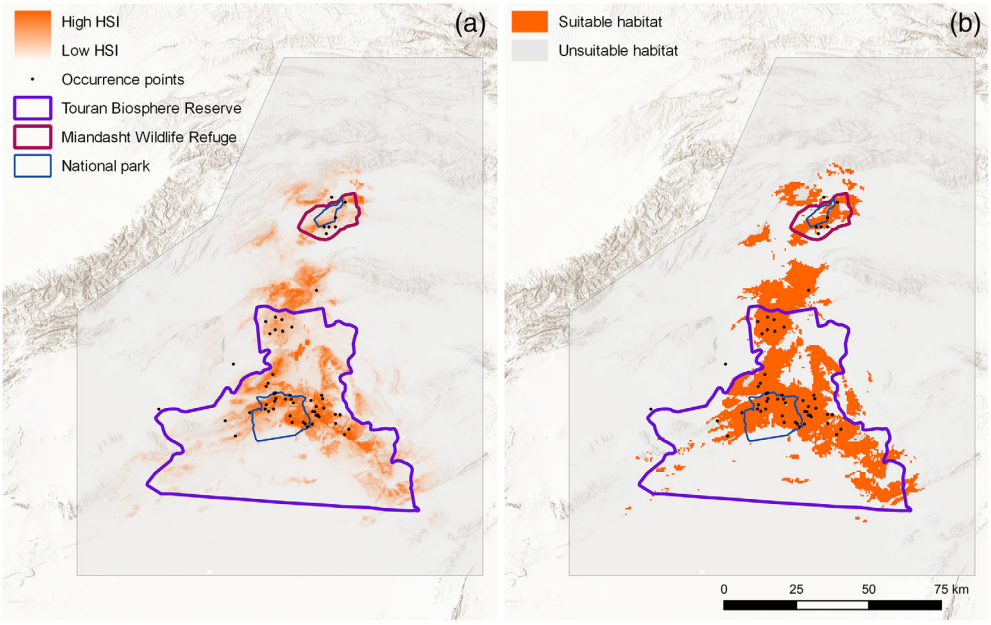

FIGURE 3 Continuous map of habitat suitability index (HSI) of Asiatic cheetah in northeastern Iran (a) and binary representation distinguishing suitable habitat (b)

minimal across these strategies (AUC = 0.88–0.91). Our final model contained six variables, with topographic roughness and distance to small rivers as the most important variables. Both had high contributions to model performance (37.3 and 25.7%, respectively), and dropping these variables caused the highest loss in training AUC (−34.3 and −24.5%, respectively). The four other variables in this model were less important in terms of relative contributions (i.e., human population density = 14%, distance to ranger stations = 11%, road density = 7%, fraction of green vegetation cover = 5%). Response curves in our final model were all in line with our a priori expectations, except for vegetation cover, our least important variable, which showed an unclear trajectory (Figure S2). Asiatic cheetahs preferred flatter, less rugged areas as well as the proximity of small rivers. Habitat suitability decreased rapidly with increasing human population density and road density, while the proximity to ranger stations had a positive effect on habitat suitability.

Predicting our final model across the study region suggested that the most suitable habitat areas were all located inside or in the close vicinity of the two protected areas (Figure 3). Using the maxSSS threshold (HSI > 0.24) showed that our study region contains 8,030 km^2 of suitable habitat for the Asiatic cheetah. Around 67% of this suitable habitat (5,368 km^2) occurred in the Touran Biosphere Reserve and only a minor portion (5% = 413 km^2) was located in the Miandasht Wildlife Refuge, with the remaining habitat outside these protected areas, particularly between them. However, not all areas inside the protected areas constitute suitable cheetah habitat (36% of the area of Touran Biosphere Reserve, 52% of the area of Miandasht Wildlife Refuge).

3.2 | Priority areas for future buyouts

Identifying priority pastures for buyouts to maximize the overall area of suitable cheetah habitat resulted in overall consistent results across our budget scenarios (Figure 4b). As the budget increased, more pastures were selected for buyouts; however, additional pastures were generally added to the existing solutions rather than the selection of an entirely new set of pastures for every scenario. Moreover, most of the additional pastures selected bordered existing buyout pastures, creating a larger contiguous area. Four pastures were selected in all solutions (two in Touran Biosphere Reserve and two in Miandasht Wildlife Refuge), and five additional pastures were selected in four of our six budget scenarios (Figure 4b). Overall, the area of suitable cheetah habitat that would become available increased relatively linearly across our budget scenarios, indicating high additivity of buyout campaigns (Figure 4a).

To exemplify the information contained in our solutions, for an available budget of US$300,000, our analysis identified ten priority pastures for buyouts at a total cost of US$299,652 (see Table S1 and Figure S3 for more details). The total area of these ten pastures was 738 km^2, with 80% of this area representing suitable cheetah habitat (see Figure S4 for details). The buyout of the selected pastures, as envisioned by this scenario, would imply the relocation of 3,742 livestock heads outside the core

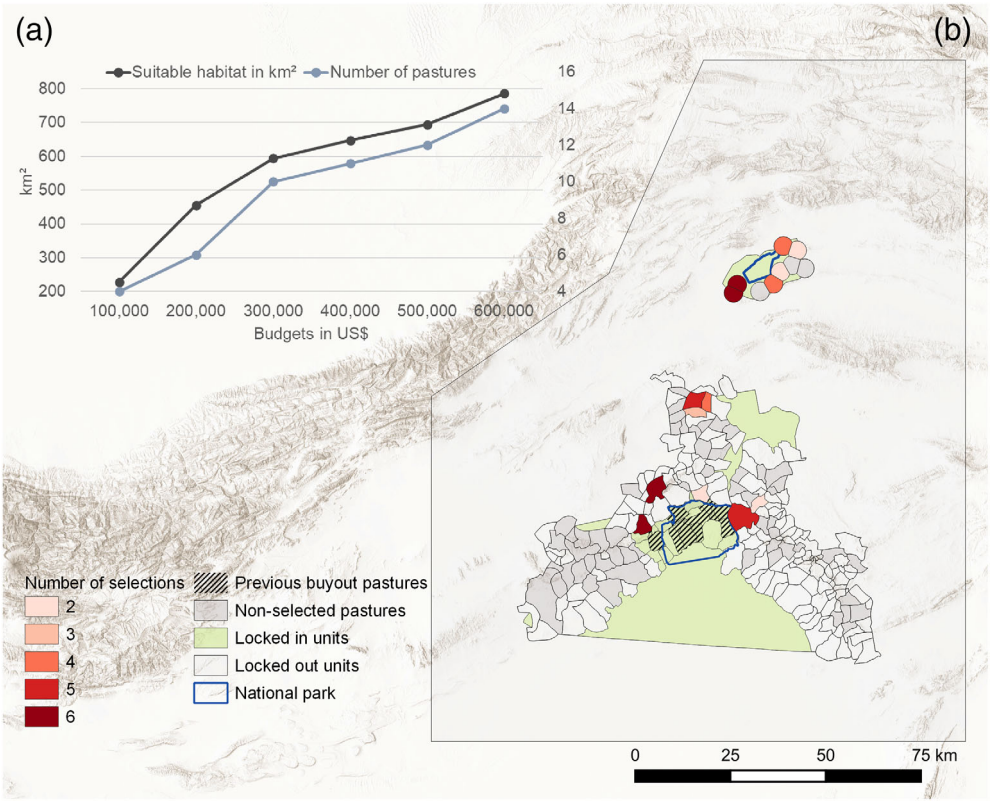

FIGURE 4 The line chart showing the suitable habitat (km^2) and the number of pastures for different budget scenarios from US$100,000 to 600,000 (a). All selected pastures and the frequency of selections within the six budget scenarios (b)

cheetah habitat where the risk for cheetah-pastoralist conflict is lower. Six pastures (2,780 livestock heads) in this solution were located in the Touran Biosphere Reserve, resulting in an additional 476 km^2 of livestock-free area, 93% of which would be suitable habitat for Asiatic cheetahs. Three of these six pastures were connected to the pastures acquired in the previous buyout campaigns, in close proximity to the national park, forming a larger livestock-free area of 1,390 km^2. The remaining four pastures were selected in the Miandasht Wildlife Refuge, covered an area of 262 km^2 and had a share of suitable cheetah habitat of 58%. Buying out these four pastures would result in the relocation of 962 livestock heads. All ten pastures selected in this budget scenario had a high irreplaceability (relative importance of planning unit selection) with a selection frequency of >98% (Figure S5).

4 | DISCUSSION

Where livestock pastures overlap with the ranges of large carnivores, achieving coexistence between people and carnivores is typically hard (Ripple et al., 2014). A range of interventions can help to minimize risks to large carnivores of conservation concern, including working with herders to better protect livestock, to reduce retaliatory killing, and to lower the poaching of carnivores' prey (Pozo et al., 2021). However, all these interventions take time to yield conservation benefits, which is problematic for critically endangered large carnivores. This is the case for the Asiatic cheetah, which faces major threats from livestock husbandry in its only remaining viable population in northeastern Iran. Reducing the spatial overlap between Asiatic cheetah habitat and livestock pastures is therefore urgently needed to prevent the global extinction of this species but where pastures should be best set aside to maximize the chances of cheetah survival is unclear. We here combined species distribution modeling with spatial prioritization to optimally allocate grazing right buyouts. To the best of our knowledge, this is a novel application of a spatial prioritization framework in the context of action planning for large carnivore conservation. Our results thus highlight a novel application of prioritization to reduce human–wildlife conflict and make a direct contribution to the conservation of one of the world's most endangered felids.

Our analyses highlight three main findings. First, we found large, contiguous areas of suitable habitat for the Asiatic cheetah in our study area where livestock is ubiquitous. Our habitat predictions are broadly in line with previous habitat assessments (Ahmadi et al., 2017; Ahmadi et al., 2020; Mohammadi et al., 2018; Nazeri et al., 2015) suggesting our habitat map was useful for our prioritization exercise to identify and prioritize livestock pastures for relocation. Our habitat analyses also

confirm the general importance of the two protected areas, as well as the area in between them. This is consistent with the analysis by Ahmadi et al. (2017), Moqanaki and Cushman (2017) and Ashrafzadeh et al. (2020) who identified this area as important to the Asiatic cheetah movement. Generally, our habitat model had highly plausible relationships between predictors and habitat suitability. The response curves (Figure S2) largely corresponded to our own a priori assumptions (Table 2), based on prior work in cheetah habitat. Roughness showed a positive effect on habitat suitability only at low levels, as confirmed by Hemami et al. (2018). Increasing habitat suitability closer to small rivers was likely related to higher hunting success, as small and even dry rivers are often associated with the presence of prey (Hunter et al., 2007). Furthermore, watercourses create more heterogeneous surfaces, possibly providing more opportunities for hiding and increasing the chance of hunting success. The beneficial effect of ranger stations we found may be due to their positive impact on prey availability, as ranger stations are known to be associated with less poaching pressure and higher ranger patrols (Ghoddousi et al., 2016). Prey availability is known as a primary predictor of the habitat use of carnivore species (Khalatbari et al., 2022; Vanak et al., 2013), and our models could be improved once accurate and recent prey data are available. The use of ensemble distribution models could have improved our predictions of cheetah habitat (Araújo & New, 2007; Elith et al., 2006), and could be considered in future habitat assessments. Most camera trap records used in our models were made within the protected areas (especially Touran Biosphere Reserve), and while we accounted for potential sampling bias through spatial filtering and sampling background data with a similar spatial structure, we cannot fully rule out remaining biases. Finally, we acknowledge that our sensitivity analysis does not cover extreme long-distance cheetah dispersals (e.g., >200 km; Farhadinia et al., 2016).

Second, while 70% of the suitable habitat occurred inside protected areas, mainly in Touran Biosphere Reserve, much of this habitat overlapped with areas used for pastoralism. This highlights the ubiquitous and high risk that livestock pastoralism represents for the Asiatic cheetah (Khalatbari et al., 2017), similar to other endangered large felids, such as for Persian leopard in the Hyrcanian forests (Soofi et al., 2018) and snow leopard (*P. uncia*) in the Himalaya (Sharma et al., 2015). Similarly, this finding underpins that while protected areas globally harbor key areas for biodiversity (Gray et al., 2016), their contribution to conservation goals may be low due to overlapping with traditional land uses (DeFries et al., 2007). For many species, such overlaps are much less problematic than for the Asiatic cheetah. For example, certain small mammals (Reid et al., 2007), reptiles (Germano et al., 2012) or insects (Alvarado et al., 2018) can thrive in traditionally grazed areas. For large carnivores, despite examples of recovering populations in human-dominated landscapes (Carter & Linnell, 2016; Chapron et al., 2014), the high overlap between carnivore habitat and livestock pastures translates to a high risk of mortality for carnivores (Chapron & López-Bao, 2016). This highlights limitations to coexistence and the importance of strictly protected areas inside the wider working landscape, particularly for species that are occurring in low densities and/or are endangered (Woodroffe et al., 2005). Thus, separating predator habitat from livestock pastures can reduce conflict, as identified by Xiao et al. (2022) for snow leopards and recommended by Kuijper et al. (2019) for wolf conservation in Europe. For the Asiatic cheetah, where every individual killed is a major step closer to global extinction, this highlights the relevance of pasture buyouts as an immediate conservation measure.

Third, our conservation prioritization exercise showed the high additionality of the solutions we found across varying budget scenarios (Figure 4b). This, and the linearly increasing conservation benefit we find with increasing budget (Figure 4a) is very encouraging, as it suggests that buyouts can start with any budget and further buyouts could be added as additional funding becomes available, without major trade-offs in terms of overall benefits (in our case: suitable cheetah habitat protected). High additionality is not a given, for example, for expansion of the current conservation network considering biodiversity and ecosystem services in the Netherlands (Remme & Schröter, 2016) the availability of full budget before implementation was recommended. In our case, we found even small budgets (e.g., US$100,000) to deliver major increases in safe suitable habitat for Asiatic cheetahs, which is encouraging as it suggests that even smaller-scale conservation initiatives can make a notable contribution to better protect this species. Even though we have expanded the cost calculation by including more variables, we acknowledge that our budget scenarios are rather indicative and there is uncertainty around cost fluctuations during and between future buyouts. On the one hand, Iran's economic situation and increase in prices could lead to higher costs of buyout but on the other hand, droughts, low willingness in traditional pastoralism in the younger generation and also awareness of the situation of Asiatic cheetah could lead to constant or lower costs. Given such levels of uncertainty, follow-up research on the willingness to participate in the buyouts and factors affecting it is necessary.

For our exemplary budget scenario of US$300,000, a budget equivalent to the previous buyout campaigns,

buyout pastures were allocated in both protected areas (six in Touran Biosphere Reserve and four in Miandasht Wildlife Refuge). All selected pastures in the optimal solution showed a high level of irreplaceability, indicating the relative importance of selecting exactly these planning units. Based on the only telemetry study on cheetahs in Iran (Cheraghi et al., 2018), we argue that the large contiguous livestock-free area of 1,390 km^2 likely reflects a safe core patch for several individuals and thus can be of substantial importance to the species. A relatively large number of buyout pastures were selected in the Miandasht Wildlife Refuge, although this reserve is much smaller than Touran Biosphere Reserve. Miandasht is an important habitat for the Asiatic cheetah with frequent observations of breeding females and past reports of cheetah cubs killed by shepherds and their guarding dogs (Iran Environment and Wildlife Watch, 2017). Still, pastures there are—at least in part—selected because of their comparatively lower costs. We caution that in the absence of accurate cost data from Miandasht these costs were only estimated based on typical movement patterns of herders. Our analyses could be updated once more accurate data becomes available. More generally, we caution that our pasture costs may not reflect the willingness of herders to actually accept the buyout offer (e.g., due to certain cultural values or ecological properties of the pasture). It is crucial to consider the ethical and socioeconomic circumstances of conservation measures (especially for segregation) and whether such measures are acceptable in the local context. Thus, excluding traditional land uses from protected areas may be ethically questionable as cultural identity can be damaged, potentially negatively affecting the ecosystem as well (Naughton-Treves et al., 2005). In our case, a survey among herders in Touran indicated that they are less dependent on pasture income and therefore more willing to accept a buyout offer to relocate (Abangah Consulting Engineer Company, 2015). Furthermore, we know from the past buyout that due to recent droughts, livestock rearing is on the decline and the people eagerly accepted the buyout offer (H.R. Mirzadeh, pers. obs., 2020). However, further in-depth assessment of people's willingness to participate in this program using targeted surveys and continued monitoring of the cheetah mortalities and human–wildlife conflict is required, as a buyout campaign would move toward implementation to make sure the problem is not moving from one location to another.

Our study provides both datasets and new insights of direct relevance to the conservation of the Asiatic cheetah. First, our results underpin the importance of the Touran Biosphere Reserve and the Miandasht Wildlife Refuge, translating into an urgency to ramp up protection level and extent in this last stronghold of the species. Given the large patch of suitable habitat between the two protected areas, an extension should be considered, as this area is an important movement corridor for Asiatic cheetahs (Ahmadi et al., 2017; Ashrafzadeh et al., 2020; Moqanaki & Cushman, 2017). Second, our study provides scientific evidence for the benefit of future buyouts, a recommended priority conservation action by Asiatic cheetah experts (Khalatbari et al., 2017). Third, we provide a spatial template for prioritizing the buyouts in both protected areas, and show that even starting with small budgets can provide major conservation benefits and could be continued once an additional budget is available. Given the higher reliance of local people on pastures in Miandasht than in Touran, alternatives to complete buyouts in Miandasht could be to reduce the numbers of livestock and dogs in order to reduce the mortality risk to cheetahs. Generally, to increase the acceptance of grazing right buyouts, further incentives than monetary compensation should be created, for example, substitute pastures that do not overlap with suitable cheetah habitat. In addition, ethical considerations and current socioeconomic conditions of pastoralists must be taken into account in future buyouts, for example through recent qualitative surveys of locals. Importantly, the future sustainability of Asiatic cheetah conservation lies in securing pathways for international financial support (in addition to domestic sources), given the current constraints caused by political sanctions (Khalatbari et al., 2018). Although we focused on livestock relocation as a conservation measure, we also acknowledge the importance of mitigating other threats. In particular, road collisions pose a major threat and have caused cheetah mortalities (Mohammadi et al., 2018). Hence, conservation funding should also be allocated to making crossings of major roads that transect cheetah habitat safer for cheetahs. Finally, human–cheetah conflict in this area should be closely monitored to adapt, if needed, buyout campaigns and to avoid funneling cheetahs to unsafe habitats (Ghoddousi et al., 2021).

Protecting large carnivores in human-dominated landscapes is one of the major challenges of our time (Ripple et al., 2014). With less than 50 individuals left (Durant et al., 2017; Khalatbari et al., 2017) the Asiatic cheetah is one of the world's most endangered large felids. Our analyses provide an evidence-based strategy for reducing the main threat to cheetah survival, livestock husbandry, via the robust and plausible identification of where pasture buyouts are feasible and beneficial to Asiatic cheetah conservation. To our knowledge, this is a novel application of prioritization to allocate grazing right buyouts for carnivore protection. Given the prevailing lack of funding in Iran due to economic

sanctions (Khalatbari et al., 2018), our approach can help to find cost-effective conservation strategies. More generally, given that many large predators around the world face similar threats, from snow leopards (Sharma et al., 2015) to lions and cheetahs (Broekhuis et al., 2017), our study highlights the potential of spatial prioritization to help avoid or reverse the local extinctions.

AUTHOR CONTRIBUTIONS

Michaela Daberger, Tobias Kuemmerle, Amirhossein Khaleghi Hamidi, Leili Khalatbari, Hamed Abolghasemi, and Arash Ghoddousi conceived the ideas and designed the study. Michaela Daberger, Amirhossein Khaleghi Hamidi, Leili Khalatbari, Hamed Abolghasemi, and Hamid Reza Mirzadeh collected and processed the raw data. Michaela Daberger implemented the data analysis. Michaela Daberger, Tobias Kuemmerle, and Arash Ghoddousi led the writing. All coauthors contributed to interpretation and manuscript drafts.

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CONFLICT OF INTEREST

The authors declare no conflict of interest.

DATA AVAILABILITY STATEMENT

The final habitat map and the priority maps with pasture boundaries used in the modeling will be provided once the paper gets accepted. Cheetah occurrence data supporting this research are sensitive and not publicly available.

ETHICS STATEMENT

This manuscript is solely the work of the authors. This study did not involve any experiments on animal or human subjects. No ethical approval was required for this research.

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REFERENCES

Abangah Consulting Engineer Company. (2015). *Planning and participatory management of livestock control in Touran Biosphere Reserve and Miandasht Wildlife Refuge* (p. 82). Conservation of Asiatic Cheetah Project (CACP).

Abangah Consulting Engineer Company. (2017). *Reconvene expanded Livestock Control Committee (LCC) in Touran and establish the LCC for Miandasht with participation of all stakeholders*. Conservation of Asiatic Cheetah Project (CACP).

Ahmadi, M., Farhadinia, M. S., Cushman, S. A., Hemami, M.-R., Nezami Balouchi, B., Jowkar, H., & Macdonald, D. W. (2020). Species and space: A combined gap analysis to guide management planning of conservation areas. *Landscape Ecology*, *35*, 1–13.

Ahmadi, M., Nezami Balouchi, B., Jowkar, H., Hemami, M. R., Fadakar, D., Malakouti-Khah, S., & Ostrowski, S. (2017). Combining landscape suitability and habitat connectivity to conserve the last surviving population of cheetah in Asia. *Diversity and Distributions*, *23*, 592–603.

Alvarado, F., Escobar, F., Williams, D. R., Arroyo-Rodríguez, V., & Escobar-Hernández, F. (2018). The role of livestock intensification and landscape structure in maintaining tropical biodiversity. *Journal of Applied Ecology*, *55*, 185–194.

Araújo, M. B., & New, M. (2007). Ensemble forecasting of species distributions. *Trends in Ecology & Evolution*, *22*, 42–47.

Ashrafzadeh, M. R., Khosravi, R., Adibi, M. A., Taktehrani, A., Wan, H. Y., & Cushman, S. A. (2020). A multi-scale, multi-species approach for assessing effectiveness of habitat and connectivity conservation for endangered felids. *Biological Conservation*, *245*, 108523.

Bautista, C., Revilla, E., Naves, J., Albrecht, J., Fernández, N., Olszańska, A., Adamec, M., Berezowska-Cnota, T., Cuicci, P., Groff, C., Härkönen, S., Huber, D., Jerina, K., Jonozovic, M., Karamanlidis, A. A., Palazon, S., Quenette, P.-Y., Rigg, R., Seijas, J., … Selva, N. (2019). Large carnivore damage in Europe: Analysis of compensation and prevention programs. *Biological Conservation*, *235*, 308–316.

Beyer, H. L., Dujardin, Y., Watts, M. E., & Possingham, H. P. (2016). Solving conservation planning problems with integer linear programming. *Ecological Modelling*, *328*, 14–22.

Boronyak, L., Jacobs, B., & Wallach, A. (2020). Transitioning towards human–large carnivore coexistence in extensive grazing systems. *Ambio*, *49*, 1982–1991.

Broekhuis, F., Cushman, S. A., & Elliot, N. B. (2017). Identification of human–carnivore conflict hotspots to prioritize mitigation efforts. *Ecology and Evolution*, *7*, 10630–10639.

Carter, N. H., & Linnell, J. D. C. (2016). Co-adaptation is key to coexisting with large carnivores. *Trends in Ecology & Evolution*, *31*, 575–578.

Carwardine, J., Wilson, K. A., Watts, M., Etter, A., Klein, C. J., & Possingham, H. P. (2008). Avoiding costly conservation mistakes: The importance of defining actions and costs in spatial priority settings. *PLoS One*, *3*, e2586.

Chadés, I., Nicol, S., van Leeuwen, S., Walters, B., Firn, J., Reeson, A., Martin, T. G., & Carwardine, J. (2015). Benefits of integrating complementarity into priority threat management. *Conservation Biology*, *29*, 525–536.

Chapron, G., Kaczensky, P., Linnell, J. D. C., Von Arx, M., Huber, D., Andrén, H., López-Bao, J. V., Adamec, M., Álvares, F., Anders, O., Balčiauskas, L., Balys, V.,Bedő, P., Bego, F., Blanco, J. C., Breitenmoser, U., Brøseth, H., Bufka, L., Bunikyte, R., ... Boitani, L. (2014). Recovery of large carnivores in Europe's modern human-dominated landscapes. *Science*, *346*, 1517–1519.

Chapron, G., & López-Bao, J. V. (2016). Coexistence with large carnivores informed by community ecology. *Trends in Ecology & Evolution*, *31*, 578–580.

Cheraghi, F., Delavar, M. R., Amiraslani, F., Alavipanah, S. K., Gurarie, E., & Fagan, W. F. (2018). Statistical analysis of Asiatic cheetah movement and its spatio-temporal drivers. *Journal of Arid Environments*, *151*, 141–145.

DeFries, R., Hansen, A., Turner, B. L., Reid, R., & Liu, J. (2007). Land use change around protected areas: Management to balance human needs and ecological function. *Ecological Applications*, *17*, 1031–1038.

Dhakal, N. P., Nelson, K. C., & Smith, D. J. L. (2011). Resident well-being in conservation resettlement: The case of padampur in the Royal Chitwan National Park, Nepal. *Society & Natural Resources*, *24*, 597–615.

Di Minin, E., Slotow, R., Hunter, L. T. B., Montesino Pouzols, F., Toivonen, T., Verburg, P. H., Leader-Williams, N., Petracca, L., & Moilanen, A. (2016). Global priorities for national carnivore conservation under land use change. *Scientific Reports*, *6*, 1–9.

Dormann, C. F., Elith, J., Bacher, S., Buchmann, C., Carl, G., Carré, G., Marquéz, J.R.G., Gruber, B., Lafourcade, B., Leitão, P.J., Münkemüller, T., Mcclean, C., Osborne, P.E., Reineking, B., Schröder, B., Skidmore, A.K., Zurell, D., & Lautenbach, S. (2017). Collinearity: A review of methods to deal with it and a simulation study evaluating their performance. *Ecography*, *36*, 27–46.

Durant, S. M., Groom, R., Ipavec, A., Mitchell, N., & Khalatbari, L. (2022). *Acinonyx jubatus*. The IUCN Red List of Threatened Species 2022: e.T219A124366642.

Durant, S. M., Mitchell, N., Groom, R., Pettorelli, N., Ipavec, A., Jacobson, A. P., Woodroffe, R., Böhm, M., Hunter, L.T.B., Becker, M.S., Broekhuis, F., Bashir, S., Andresen, L., Aschenborn, O., Beddiaf, M., Belbachir, F., Belbachir-Bazi, A., Berbash, A., De Matos Machado, I.B., ... Young-Overton, K. (2017). The global decline of cheetah *Acinonyx jubatus* and what it means for conservation. *Proceedings of the National Academy of Sciences of the United States of America*, *114*, 528–533.

Eklund, A., López-Bao, J. V., Tourani, M., Chapron, G., & Frank, J. (2017). Limited evidence on the effectiveness of interventions to reduce livestock predation by large carnivores. *Scientific Reports*, *7*, 1–9.

Elith, J., Graham, C. H., Anderson, R. P., Dudík, M., Ferrier, S., Guisan, A., Hijmans, R., Huettmann, F., R. Leathwick, J., Lehmann, A., Li, J., G. Lohmann, L., A. Loiselle, B., Manion, G., Moritz, C., Nakamura, M., Nakazawa, Y., McC. M. Overton, J., Townsend Peterson, A., ... Zimmermann, N. E. (2006). Novel methods improve prediction of species' distributions from occurrence data. *Ecography*, *29*, 129–151.

Elith, J., & Leathwick, J. (2009). Species distribution models: Ecological explanation and prediction across space and time. *Annual Review of Ecology, Evolution, and Systematics*, *40*, 677–697.

Estes, J. A., Terborgh, J., Brashares, J. S., Power, M. E., Berger, J., Bond, W. J., Carpenter, S.R., Essington, T.E., Holt, R.D., Jackson, J.B.C., Marquis, R.J., Oksanen, L., Oksanen, T., Paine, R. T., Pikitch, E.K., Ripple, W.J., Sandin, S.A., Scheffer, M., Schoener, ... Wardle, D. A. (2011). Trophic downgrading of planet earth. *Science*, *333*, 301–306.

Farhadinia, M. S., Gholikhani, N., Behnoud, P., Hobeali, K., Taktehrani, A., Hosseini-Zavarei, F., Eslami, M., & Hunter, L. T. B. (2016). Wandering the barren deserts of Iran: Illuminating high mobility of the Asiatic cheetah with sparse data. *Journal of Arid Environments*, *134*, 145–149.

Farhadinia, M. S., Hunter, L. T. B., Jourabchian, A., Hosseini-Zavarei, F., Akbari, H., Ziaie, H., & Jowkar, H. (2017). The critically endangered Asiatic cheetah *Acinonyx jubatus venaticus* in Iran: A review of recent distribution, and conservation status. *Biodiversity and Conservation*, *26*, 1027–1046.

Farhadinia, M. S., Hunter, L. T. B., Jowkar, H., Schaller, G. B., & Ostrowski, S. (2018). Asiatic cheetahs in Iran: Decline, current status and threats. In P. J. Nyhus, L. Marker, L. K. Boast, & A. Schmidt (Eds.), *Cheetahs: Biology and conservation: Biodiversity of the world: Conservation from genes to landscapes* (pp. 55–69). Academic Press.

Feng, J., Sun, Y., Li, H., Xiao, Y., Zhang, D., Smith, J. L. D., Ge, J., & Wang, T. (2021). Assessing mammal species richness and occupancy in a Northeast Asian temperate forest shared by cattle. *Diversity and Distributions*, *27*, 857–872.

Fick, S. E., & Hijmans, R. J. (2017). WorldClim 2: New 1-km spatial resolution climate surfaces for global land areas. *International Journal of Climatology*, *37*, 4302–4315.

Germano, D. J., Rathbun, G. B., & Saslaw, L. R. (2012). Effects of grazing and invasive grasses on desert vertebrates in California. *The Journal of Wildlife Management*, *76*, 670–682.

Ghoddousi, A., Buchholtz, E. K., Dietsch, A. M., Williamson, M. A., Sharma, S., Balkenhol, N., ... Dutta, T. (2021). Anthropogenic resistance: Accounting for human behavior in wildlife connectivity planning. *One Earth*, *4*, 39–48.

Ghoddousi, A., Khaleghi Hamidi, A. H., Soofi, M., Khorozyan, I., Kiabi, B. H., & Waltert, M. (2016). Effects of ranger stations on predator and prey distribution and abundance in an Iranian steppe landscape. *Animal Conservation*, *19*, 273–280.

Giuliano, W. M., & Homyack, J. D. (2004). Short-term grazing exclusion effects on riparian small mammal communities. *Journal of Range Management*, *57*, 346–350.

Gray, C. L., Hill, S. L. L., Newbold, T., Hudson, L. N., Boïrger, L., Contu, S., Hoskins, A.J., Ferrier, S., Purvis, A., & Scharlemann, J. P. W. (2016). Local biodiversity is higher inside than outside terrestrial protected areas worldwide. *Nature Communications*, *7*, 1–7.

Gurobi Optimization LLC. (2020). Gurobi optimizer reference manual.

Hanson, J. O., Schuster, R., Morrell, N., Strimas-Mackey, M., Watts, M. E., Arcese, P., & Possingham, H. P. (2020). prioritizr: Systematic conservation prioritization in R. R package version 5.0.2. https://CRAN.R-project.org/package=prioritizr.

Hemami, M. R., Esmaeili, S., Brito, J. C., Ahmadi, M., Omidi, M., & Martínez-Freiría, F. (2018). Using ecological models to explore niche partitioning within a guild of desert felids. *Hystrix*, *29*, 216–222.

Heshmati, G. A. (2007). Vegetation characteristics of four ecological zones of Iran. *International Journal of Plant Production*, *1*, 215–224.

Hijmans, R. J., Phillips, S., Leathwick, J., & Elith, J. (2017). Package 'dismo'. *Circles*, *9*, 1–68.

Hoeks, S., Huijbregts, M. A. J., Busana, M., Harfoot, M. B. J., Svenning, J. C., & Santini, L. (2020). Mechanistic insights into the role of large carnivores for ecosystem structure and functioning. *Ecography*, *43*, 1752–1763.

Hughes, J., & Macdonald, D. W. (2013). A review of the interactions between free-roaming domestic dogs and wildlife. *Biological Conservation*, *157*, 341–351.

Hunter, L., Jowkar, H., Ziaie, H., Schaller, G., Balme, G., Walzer, C., Ostrowski, S., Zahler, P., Robert-Charrue, N., & Kashiri, K. (2007). Conserving the Asiatic cheetah in Iran: Launching the first radio-telemetry study. *Cat News*, *46*, 8–11.

Iran Environment and Wildlife Watch. (2017). The female cheetah died and her cubs were attacked by herding dogs. http://www.iew.ir/1396/06/23/54477.

Karanth, K. K. (2007). Making resettlement work: The case of India's Bhadra Wildlife Sanctuary. *Biological Conservation*, *139*, 315–324.

Karanth, K. U., & Karanth, K. K. (2007). Free to move: Conservation and voluntary resettlements in the Western Ghats of Karnataka, India. In E. Fearn & K. H. Redford (Eds.), *Protected areas and human displacement: A conservation perspective* Working Paper 29 (pp. 48–59). Wildlife Conservation Society.

Khalatbari, L., Brito, J. C., Ghoddousi, A., Abolghasemi, H., Breitenmoser, U., Breitenmoser-Würsten, C., Yusefi, G.H., Ostrowski, S., & Durant, S. M. (2018). Sanctioning to extinction in Iran. *Science*, *362*, 1255.

Khalatbari, L., Egeter, B., Abolghasemi, H., Hakimi, E., Ghadirian, T., Khaleghi Hamidi, A. H., Jowkar, H., Breitenmoser, U., & Brito, J. C. (2022). Assessing Asiatic cheetah's individual diet using metabarcoding and its implication for conservation. *Scientific Reports*, *12*, 1–12.

Khalatbari, L., Jowkar, H., Yusefi, G. H., Brito, J. C., & Ostrowski, S. (2017). The current status of Asiatic cheetah in Iran. *Cat News*, *66*, 10–13.

Khosravi, R., Hemami, M. R., & Cushman, S. A. (2018). Multispecies assessment of core areas and connectivity of desert carnivores in central Iran. *Diversity and Distributions*, *24*, 193–207.

Kramer-Schadt, S., Niedballa, J., Pilgrim, J. D., Schröder, B., Lindenborn, J., Reinfelder, V., Stillfried, M., Heckmann, I., Scharf, A.K., Augeri, D.M., Cheyne, S.M., Hearn, A.J., Ross, J., Macdonald, D.W., Mathai, J., Eaton, J., Marshall, A.J., Semiadi, G., Rustam, R., ... Wilting, A. (2013). The importance of correcting for sampling bias in MaxEnt species distribution models. *Diversity and Distributions*, *19*, 1366–1379.

Kuijper, D. P. J., Churski, M., Trouwborst, A., Heurich, M., Smit, C., Kerley, G. I. H., & Cromsigt, J. P. G. M. (2019). Keep the wolf from the door: How to conserve wolves in Europe's human-dominated landscapes? *Biological Conservation*, *235*, 102–111.

Kukkala, A. S., & Moilanen, A. (2013). Core concepts of spatial prioritisation in systematic conservation planning. *Biological Reviews*, *88*, 443–464.

Li, J., Weckworth, B. V., McCarthy, T. M., Liang, X., Liu, Y., Xing, R., Li, D., Zhang, Y., Xue, Y., Jackson, R., Xiao, L., Cheng, C., Li, S., Xu, F., Ma, M., Yang, X., Diao, K., Gao, Y., Song, D., ... Beissinger, S. R. R. (2020). Defining priorities for global snow leopard conservation landscapes. *Biological Conservation*, *241*, 108387.

López-Bao, J. V., Bruskotter, J., & Chapron, G. (2017). Finding space for large carnivores. *Nature Ecology and Evolution*, *1*, 140.

McGranahan, D. A., Geaumont, B., & Spiess, J. W. (2018). Assessment of a livestock GPS collar based on an open-source datalogger informs best practices for logging intensity. *Ecology and Evolution*, *8*, 5649–5660.

Mohammadi, A., Almasieh, K., Clevenger, A. P., Fatemizadeh, F., Rezaei, A., Jowkar, H., & Kaboli, M. (2018). Road expansion: A challenge to conservation of mammals, with particular emphasis on the endangered Asiatic cheetah in Iran. *Journal for Nature Conservation*, *43*, 8–18.

Mohammadi, A., & Kaboli, M. (2016). Evaluating wildlife-vehicle collision hotspots using kernel-based estimation: A focus on the endangered Asiatic cheetah in central Iran. *Human–Wildlife Interactions*, *10*, 103–109.

Molnár, Z. (2012). Classification of pasture habitats by Hungarian herders in a steppe landscape (Hungary). *Journal of Ethnobiology and Ethnomedicine*, *8*, 28.

Moqanaki, E. M., & Cushman, S. A. (2017). All roads lead to Iran: Predicting landscape connectivity of the last stronghold for the critically endangered Asiatic cheetah. *Animal Conservation*, *20*, 29–41.

Morehouse, A. T., & Boyce, M. S. (2017). Troublemaking carnivores: conflicts with humans in a diverse assemblage of large carnivores. *Ecology and Society*, *22*. https://doi.org/10.5751/es-09415-220304

Naughton-Treves, L., Holland, M. B., & Brandon, K. (2005). The role of protected areas in conserving biodiversity and sustaining local livelihoods. *Annual Review of Environment and Resources*, *30*, 219–252.

Nayeri, D., Mohammadi, A., Qashqaei, A. T., Vanak, A. T., & Gompper, M. E. (2021). Free-ranging dogs as a potential threat to Iranian mammals. *Oryx*, 56, 383-389.

Nazeri, M., Madani, N., Kumar, L., Salman Mahiny, A., & Kiabi, B. H. (2015). A geo-statistical approach to model Asiatic cheetah, onager, gazelle and wild sheep shared niche and distribution in Turan biosphere reserve-Iran. *Ecological Informatics*, *29*, 25–32.

Newbold, T., Bentley, L. F., Hill, S. L. L., Edgar, M. J., Horton, M., Su, G., Şekercioğlu, Ç.H., Collen, B., & Purvis, A. (2020). Global effects of land use on biodiversity differ among functional groups. *Functional Ecology*, *34*, 684–693.

Packer, C., Loveridge, A., Canney, S., Caro, T., Garnett, S. T., Pfeifer, M., ander, K.K., Swanson, A., MacNulty, D., Balme, G., Bauer, H., Begg, C.M., Begg, K.S., Bhalla, S., Bissett, C., Bodasing, T., Brink, H., Burger, A., Burton, A.C., Polasky, S. (2013). Conserving large carnivores: Dollars and fence. *Ecology Letters*, *16*, 635–641.

Phillips, S. J., Anderson, R. P., & Schapire, R. E. (2006). Maximum entropy modeling of species geographic distributions. *Ecological Modelling*, *190*, 231–259.

Phillips, S. J., & Dudík, M. (2008). Modeling of species distributions with Maxent: New extensions and a comprehensive evaluation. *Ecography*, *31*, 161–175.

Phillips, S. J., Dudík, M., Elith, J., Graham, C. H., Lehmann, A., Leathwick, J., & Ferrier, S. (2009). Sample selection bias and presence-only distribution models: Implications for background and pseudo-absence data. *Ecological Applications*, *19*, 181–197.

Pozo, R. A., Cusack, J. J., Acebes, P., Malo, J. E., Traba, J., Iranzo, E. C., Morris-Trainor, Z., Minderman, J., Bunnefeld, N., Radic-Schilling, S., Moraga, C.A., Arriagada, R., & Corti, P. (2021). Reconciling livestock production and wild herbivore conservation: Challenges and opportunities. *Trends in Ecology & Evolution*, *36*, 750–761.

Pudyatmoko, S. (2017). Free-ranging livestock influence species richness, occupancy, and daily behaviour of wild mammalian species in Baluran National Park, Indonesia. *Mammalian Biology*, *86*, 33–41.

Reid, N., McDonald, R. A., & Montgomery, W. I. (2007). Mammals and agri-environment schemes: Hare haven or pest paradise? *Journal of Applied Ecology*, *44*, 1200–1208.

Remme, R. P., & Schröter, M. (2016). Effects of budget constraints on conservation network design for biodiversity and ecosystem services. *Ecological Complexity*, *26*, 45–56.

Ripple, W. J., Estes, J. A., Beschta, R. L., Wilmers, C. C., Ritchie, E. G., Hebblewhite, M., Berger, J., Elmhagen, B., Letnic, M., Nelson, M.P., Schmitz, O.J., Smith, D.W., Wallach, A. D., & Wirsing, A. J. (2014). Status and ecological effects of the world's largest carnivores. *Science*, *343*, 1241484.

Ripple, W. J., Newsome, T. M., Wolf, C., Dirzo, R., Everatt, K. T., Galetti, M., Hayward, M.W., Kerley, G.I.H., Levi, T., Lindsey, P. A., Macdonald, D.W., Malhi, Y., Painter, L.E., Sandom, C.J., Terborgh, J., & van Valkenburgh, B. (2015). Collapse of the world's largest herbivores. *Science Advances*, *1*, e1400103.

Schieltz, J. M., & Rubenstein, D. I. (2016). Evidence based review: Positive versus negative effects of livestock grazing on wildlife. What do we really know? *Environmental Research Letters*, *11*, 113003.

Sharma, R. K., Bhatnagar, Y. V., & Mishra, C. (2015). Does livestock benefit or harm snow leopards? *Biological Conservation*, *190*, 8–13.

Soofi, M., Ghoddousi, A., Zeppenfeld, T., Shokri, S., Soufi, M., Jafari, A., Ahmadpour, M., Qashqaei, A.T., Egli, L., Ghadirian, T., Chahartaghi, N.R., Zehzad, B., Kiabi, B.H., Khorozyan, I., Balkenhol, & Waltert, M. (2018). Livestock grazing in protected areas and its effects on large mammals in the Hyrcanian forest, Iran. *Biological Conservation*, *217*, 377–382.

Steinfeld, H., Gerber, P., Wassenaar, T. D., Castel, V., Rosales, M., Rosales, M., & de Haan, C. (2006). *Livestock's long shadow: Environmental issues and options*. FAO.

Torre, I., Díaz, M., Martínez-Padilla, J., Bonal, R., Viñuela, J., & Fargallo, J. A. (2007). Cattle grazing, raptor abundance and small mammal communities in Mediterranean grasslands. *Basic and Applied Ecology*, *8*, 565–575.

Torri, M. C. (2011). Conservation, relocation and the social consequences of conservation policies in protected areas: Case study of the Sariska Tiger Reserve, India. *Conservation and Society*, *9*, 54–64.

Treves, A., & Karanth, K. U. (2003). Human-carnivore conflict and perspectives on carnivore management worldwide. *Conservation Biology*, *17*, 1491–1499.

Treves, A., Wallace, R. B., & White, S. (2009). Participatory planning of interventions to mitigate human-wildlife conflicts. *Conservation Biology*, *23*, 1577–1587.

van Eeden, L. M., Crowther, M. S., Dickman, C. R., Macdonald, D. W., Ripple, W. J., Ritchie, E. G., & Newsome, T. M. (2018). Managing conflict between large carnivores and livestock. *Conservation Biology*, *32*, 26–34.

Vanak, A. T., Fortin, D., Thaker, M., Ogden, M., Owen, C., Greatwood, S., & Slotow, R. (2013). Moving to stay in place: Behavioral mechanisms for coexistence of African large carnivores. *Ecology*, *94*, 2619–2631.

Wilson, K. A., Cabeza, M., & Klein, C. J. (2009). Fundamental concepts of spatial conservation prioritization. In A. Moilanen, K. A. Wilson, & H. P. Possingham (Eds.), *Spatial conservation prioritization: Quantitative methods and computational tools* (pp. 16–27). Oxford University Press.

Wilson, K. A., Underwood, E. C., Morrison, S. A., Klausmeyer, K. R., Murdoch, W. W., Reyers, B., ... Possingham, H. P. (2007). Conserving biodiversity efficiently: What to do, where, and when. *PLoS Biology*, *5*, 1850–1861.

Wisz, M. S., Hijmans, R. J., Li, J., Peterson, A. T., Graham, C. H., & Guisan, A. (2008). Effects of sample size on the performance of species distribution models. *Diversity and Distributions*, *14*, 763–773.

Woodroffe, R., Thirgood, S., & Rabinowitz, A. (2005). *People and wildlife, conflict or co-existence?* Cambridge University Press.

Xiao, L., Hua, F., Knops, J. M. H., Zhao, X., Mishra, C., Lovari, S., Alexander, J.S., Weckworth, B., & Lu, Z. (2022). Spatial separation of prey from livestock facilitates coexistence of a specialized large carnivore with human land use. *Animal Conservation*. https://doi.org/10.1111/acv.12769

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