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EFFECTS OF ANTHROPOGENIC PRESSURE ON LARGE MAMMAL SPECIES IN THE HYRCANIAN FOREST, IRAN

Dissertation

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رنج راحت شد، چو مطلب شد بزرگ

گرد گله توتیای چشم گرگ شد

Towards great deeds, do not fear hardships

The herd's dust is kohl for an eager wolf's eyes

Sheikh Baha'i (1548-1621)

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A typical montane area of the Hyrcanian forest (Dodangeh Wildlife Refuge). *Photo by M. Soofi.*

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Summary

Anthropogenic activities, such as overexploitation (poaching, logging) and farming (livestock grazing), are the most serious threats to biodiversity and ecosystem services. The effects of these drivers may be synergistic and variable across different species. Many terrestrial large-bodied mammals experience range shrinkage and face extinction risks and population declines across the world. By these activities, humans either directly (prey poaching) or indirectly (logging and livestock grazing) affect the survival rates of large mammal species. Protected areas (PAs) have been the most effective tool to preserve native species. However, the effectiveness of PAs in relation to large mammal distribution or conservation in temperate forests has rarely been assessed, particularly at a large landscape scale. In this study, I assessed the effects of threats to seven native mammal species in the Hyrcanian forest of Iran, namely the Persian Leopard (*Panthera pardus saxicolor*), grey wolf (*Canis lupus*), brown bear (*Ursus acrtos*), bezoar goat (*Capra aegagrus*), Caspian red deer (*Cervus elaphus maral*), roe deer (*Capreolus capreolus*), and wild boar (*Sus scrofa*). In addition, I assessed whether there are direct effects of poaching on livestock depredation by large carnivores.

I used a novel approach to survey mammal species occupancy over a large landscape (18 protected and non-protected areas) and walked 1204 km distributed randomly over 93 16-km^2 cells. Field surveys resulted in 2876 animal signs of the above-mentioned species over three discrete surveys. I used single-season Bayesian occupancy modeling and estimated the occupancy and detection probability rates for each target species across the study sites. The results explicitly showed that grazing had negative and significant impact on the occupancy of the very patchily distributed Persian leopard (β = -1.65, Credibility Interval CI - 2.85 to -0.65), Caspian red deer (β = -1.36, CI -2.34 to -0.45) and roe deer (β = -1.61, CI - 2.96 to -0.58) while logging negatively affected red deer (β = -0.82, CI -1.69 to -0.03). The intensities of grazing and logging were correlated (r = 0.59), followed by logging and poaching (r = 0.39), grazing and poaching (r = 0.37) (Chapter2).

I estimated the population density of the main wild ungulate species hunted by poachers and Persian leopards using random encounter modelling from camera traps (2777 camera days), fecal standing crop sampling (38 km), direct line transect sampling (186 km) and double-observer point-counts (64 scans) (Chapters 3 and 4). The results suggest that, due to poaching, population densities of the Caspian red deer, bezoar goat and urial have decline by 66-89% compared to the 1970ies. However, wild boar abundance estimates have increased by 58% during the same period.

Using wild prey encounter rates (1204 km) and interview data (n = 201), I estimated the effects of forest cover, IUCN category of reserves, distance to villages and livestock encounter rates on livestock depredation rates by Persian leopard and grey wolf. Prey poaching was the most influential predictor of livestock depredation, as an increase in poaching occurrence by one sign/km significantly increased depredation up to three times depending on the combination of livestock and carnivore species. The results also showed that the level of poaching was significantly lower in national parks (cat. II) than in other reserves and non-protected areas, though poaching signs were frequently found in the majority of surveyed cells (58%). The occurrences of Caspian red deer and roe deer were significantly inversely associated with poaching and these species seem to be locally extinct in some of the surveyed sites.

Furthermore, using household interview data (n = 162) in 45 villages and wild prey richness, I assessed the species-specific patterns of human-wildlife conflicts in the Hyrcanian forest. Based on multivariate analyses, grey wolf and wild boar were the major conflict species. Crop loss due to wild boars was reported by the majority of the households and was mainly triggered by crop variety. Wolf conflicts were mainly related to depredation on sheep (81%) compared to goats (11%) and cattle (8%). These attacks were positively associated with highlands, villages located in vicinity or inside PAs and lower prey richness.

In conclusion, conservation authorities should consider upgrading parts of protected areas and wildlife refuges retaining natural habitats to the category II. PAs require priority actions in assessment of grazing capacities, allocation and enforcement of grazing quotas. Moreover, better cross-sectoral coordination is needed among conservation authorities to avoid further depletion of the mammal community in the Hyrcanian forest and to address sustainable livelihoods near PAs. The poachers' incentives need to be clearly understood and be subject to focused follow-up studies. Mitigating livestock depredation requires a combination of strict law enforcement of anti-poaching measures, upgrading the status of reserves and wild prey recovery plans. The persistence of protection-reliant species depends on their existence outside and inside PAs, ecological requirements and law enforcement measures. Thus, for sustainable wildlife conservation a holistic participatory approach is essential that involves local communities.

Chapter 1

General introduction

Habitat degradation in the Hyrcanian forest landscape. *Photo by M.Soofi.*

1.1. Anthropogenic threats to biodiversity

Nowadays, the world confronts an unprecedented loss of biodiversity that is referred to as a human-induced mass extinction (Karanth et al. 2010; Yackulic et al. 2011; Ceballos et al. 2015; Ripple et al. 2015). Many large-bodied terrestrial mammals experience range shrinkage and face extinction due to habitat destruction and overexploitation across the world (Ripple et al. 2014). Of all animal taxonomic groups that have gone extinct since AD 1500, 75% were threatened by overexploitation (i.e. poaching, logging) or agricultural activities (i.e. livestock grazing, crop farming), which are the biggest drivers of biodiversity decline (Maxwell et al. 2016). The effects of these drivers may be synergistic across different species (Brodie et al. 2015). For example, logging and livestock grazing affect habitat size and quality, but they also facilitate poaching on large mammals (Laurance et al. 2008).

Livestock continues to encroach natural habitats rapidly and has multiplicative effects on large mammal distribution (Ripple et al. 2015; Maxwell et al. 2016). Such a pervasive pressure on ecosystems can have extensive impacts that can result in cascading effects, where large predators increasingly hunt smaller prey species or even switch to other prey, ultimately altering ecosystem functions and services (Brook et al. 2008; Ripple et al. 2016; Gurdon et al. 2017).

Habitat degradation and loss often accompany poaching and result in depletion of wild ungulates (Brodie et al. 2015). As a consequence, large carnivores may be forced to shift to livestock depredation (Khorozyan et al. 2015; Suryawanshi et al. 2013, 2017). Logging also simplifies the complexity of forest systems and reduces the quality of habitats for biodiversity in general (Müller et al. 2015). Some studies have demonstrated that the distribution of many endemic species in tropical forests is severely threatened by logging concessions (Laurance et al. 2008; Brodie et al. 2015). In addition, logging has multiple effects on large mammal distribution, which often correlates with overgrazing and poaching (Brodie et al. 2015). In turn, poaching drives many species, especially larger ones, towards extinction through the reduction in abundance and distribution (Milner-Gulland et al. 2003; Rauset et al. 2016). Several studies report local extirpations caused by logging, overgrazing and poaching in tropical regions (Brodie et al. 2015; Benitez-Lopez et al. 2017), but the effects of these threats in temperate ecosystems remain heavily understudied given the critical role of large mammal species in ecosystem functioning.

1.2. Impacts of overexploitation on large mammals

The effects of exploitation on mammal species vary among different geographic regions (Ripple et al. 2015). Globally, of the 301 mammal species threatened by overhunting, 61 species occur in Asia (20%) where the majority of these species exist in rural parts of the countries and poverty is common (Milner-Gulland et al. 2003; Ripple et al. 2016).

Due to intrinsic biological traits, such as large body size, large home range and limited reproduction rate, large-bodied mammals are subject to various anthropogenic threats that require conservation at the landscape level (Ripple et al. 2014; Chapron & Lopez-Bao 2016). This is especially true for heavily hunted large herbivores. Consequently, this affects large carnivores which strongly depend on their prey resources (Chapron et al. 2008; Ghoddousi et al. 2017a). Therefore, vulnerability to extermination makes large carnivores priority species for conservation (Brook et al. 2007; Suryawanshi et al. 2017). However, prey depletion occurs cryptically and it is rarely detectable, especially in densely forested habitats (Laurance et al. 2008; Brodie et al. 2015). Furthermore, when wild prey species are depleted livestock become dominant in wildlife habitats and potentially compete with wild herbivores (Ripple et al. 2014).

As such, when wild prey resources are scarce, large carnivores can shift to prey more upon domestic animals (Ghoddousi et al. 2016). In such conditions, 'problem' individuals can be killed by rural people in retaliation to livestock depredation which brings additional conservation challenges (Chapron et al. 2008; Chapron & Lopez-Bao 2016). Thus, knowing the causal relationships between human pressures and large mammal communities is essential for long-term conservation.

1.3. Conservation status of terrestrial large mammal species in Iran

So far, about 200 mammal species have been identified across Iran, of which 60 (30.5%) species live in the Hyrcanian forest (Karami et al. 2017). The Hyrcanian forest has a diverse community of large mammals, including the endangered Persian leopard (*Panthera pardus saxicolor*), brown bear (*Ursus arctos*), grey wolf (*Canis lupus*), endangered Caspian red deer (*Cervus elaphus maral*), endangered roe deer (*Capreolus capreolus*), vulnerable bezoar goat (*Capra aegagrus*), vulnerable urial (*Ovis vignei*) and wild boar (*Sus scrofa*). In 1958, this forest has lost one of its top predators, the globally extinct Caspian tiger (*Panthera tigris virgata*), due to poaching and habitat loss (Firouz 2005). According to the Iranian Department of Environment (hereafter referred to as DoE), the Caspian red deer, roe deer and Persian leopard are nationally listed as ‘endangered’, and the Persian leopard is ‘endangered’ also globally (Khorozyan 2008). The bezoar goat is considered as globally ‘vulnerable’ (IUCN 2008). The brown bear and grey wolf are common in Iran but are being persecuted because of conflicts over livestock depredation (Qashqaei et al. 2014); the same applies also to wild boar for crop raiding.

Approximately four million livestock (i.e. cattle, sheep and goat) are roaming in the Hyrcanian forest (Sagheb-Talebi et al. 2014), potentially out-competing wild herbivores and increasingly simplifying the forest system (Kiabi et al. 2004). As a consequence, large carnivores are increasingly threatened due to habitat loss and retaliatory killings resulting from livestock depredation (Kiabi et al. 2002; Ghoddousi et al. 2016; Khorozyan et al. 2017). Despite of severe population decline of most ungulate species due to poaching, livestock grazing and logging (Kiabi et al. 2002; Ghoddousi et al. 2016; Ghoddousi et al. 2017), these threats have never been assessed at the community level in the Hyrcanian forest.

1.4. Status of protected areas in Iran

Iran covers 1,648,000 km^2 and is known to be the main biodiversity “bridge” between Central Asia, Arabia, India and Mesopotamia (Firouz 2005). From a biogeographical standpoint, it is a crossroad between Asia and Europe and is home to a great diversity of large mammalian species with contrasting life histories and habitat use patterns (Firouz 2005; Khalilzdeh et al. 2016; Karami et al. 2017; Ghoddousi et al. 2017). Despite Iran being blessed with a remarkable diversity of landscapes, fauna and flora, it has largely been overlooked by conservation scientists in the past decades (Firouz 2005; Darvishsefat 2006; Kolahi et al. 2012).

The hunting law in Iran was designated in 1956 with the establishment of the Game Council of Iran, later renamed to the Department of Environment (DoE) in 1974, and the creation of the first network of protected areas (Firouz 2005; Ghoddousi et al. 2017). Since then, DoE had continued to increase the number of protected areas (PAs), which now cover 10.34% (n = 274 PAs) of the country’s land surface (Makhdoum 2008; Kolahi et al. 2012; DoE 2016; Ghoddousi et al. 2017). However, earlier studies dealing with the prioritization of PAs in Mazandaran Province confirmed that their effectiveness is inadequate to achieve conservation goals (Kolahi et al. 2012; Mehri et al. 2014).

According to Iranian legislation, protection regime changes from most strict to least strict along the categories “national park”, “wildlife refuge” and “protected area”. Additionally, no-hunting areas have been established for a short period of time (usually 5 years) to allow recovery of threatened populations (Darvishsefat 2006; Kolahi et al. 2012). The Hyrcanian forest comprises three national parks, two wildlife refuges, 16 protected areas and 16 no-hunting areas (Zehzad et al. 2002; Darvishsefat 2006).

1.5. The Hyrcanian forest

The Hyrcanian forest, also called the Caspian forest, represents 15% of the total Iranian forest and is the last relict of primary temperate broadleaf forests globally (Sagheb-Talebi et al. 2014; Figure 1). This forest covers the southeastern part of the Caucasus Biodiversity Hotspot and is of great biogeographic and conservation importance (Olson and Dinnerstein 2002; Breitenmoser et al. 2017). The forest is a green bow fringing the southern coastline of the Caspian Sea and stretching from the Talysh Mountains in Azerbaijan eastwards over the northern slopes of the Alborz Ridge to Golestan National Park in Iran (Akhani et al. 2010). The area is almost 800 km long and 110 km wide, covers 18,500 km^2 and the elevation ranges from -28 m to 2800 m above sea level. The mean annual precipitation ranges within 530-1350 mm. The mean air temperature of the warmest and coldest months varies from 28-35°C to 1.5-4°C, respectively (Sagheb-Talebi et al. 2014).

The Hyrcanian forest is dominated mainly by relic species in lowlands: Caucasian elm (*Zelcova carpinifolia*), Persian Ironwood (*Parrotia persica*), Caspian locust (*Gleditsia caspica*) and *Pterocaria fraxinifolia*. The main species at higher elevations are the chestnut-leaved oak (*Quercus castaneifolia*) and hornbeam (*Carpinus betulus*). The Oriental beech (*Fagus orientalis*) occurs at 400-1500 m and the Persian oak (*Quercus macranthera*), field maple (*Acer campestre*), Montpellier maple (*Acer monspessulanum*) and Oriental hornbeam (*Carpinus orientalis*) communities dominate at 1500-2800 m. The transitional scrub-line, which is covered mainly by barberry (*Berberis integerrima*), Boissier pear (*Pyrus boissieriana*), buckthorn (*Rhamnus cathartica*) and Oriental apple (*Malus orientalis*), is present from ~ 200 m to 3000 m (Akhani et al. 2010).

Human population of the three Iranian provinces, which encompass the Hyrcanian forest (Gilan, Mazandaran and Golestan), is nearly 7.7 million or 10% of the Iranian population, and has doubled since 1976 (Statistical Center of Iran 2016). At the same time, the Hyrcanian forest is among the most severely threatened ecosystems in Iran due to its fertile soils, mild climate, timber productivity and attractiveness for new settlements (Akhani et al. 2010). This region is affected by intense human use, e.g. poaching, logging, overgrazing of livestock, fire, drought, rapid urbanization and industrialization and infrastructure development (e.g. road and dam construction) (Akhani et al. 2010; Sagheb-Talebi et al. 2014; Ghoddousi et al. 2017). As a result, it lost ~ 45% of its forest during 1942-2005 (Sagheb-Talebi et al. 2014).

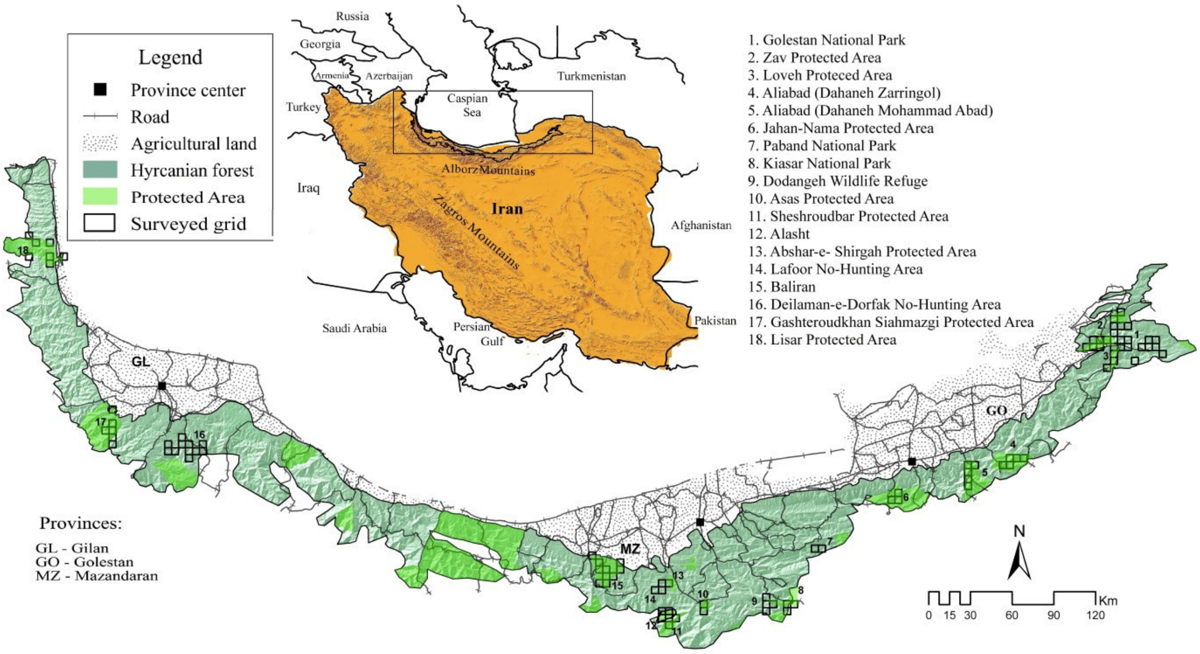

Figure 1. Map of the Hyrcanian forest, presenting the selected study sites in numerous protected areas of three provinces in northern Iran.

The law of nationalization of Iran's forests was passed in 1963 and the Forest, Rangeland and Watersheds Organization (FRWO) was authorized for the protection of plant biodiversity. Later in 1982, DoE in concert with FRWO adopted a new law, banning livestock grazing inside the core areas which covered only 20% of protected areas and wildlife refuges (Makhdoum 2008). With the law of 1989, FRWO attempted to exclude livestock grazing from the entire Hyrcanian forest by providing arable land to herders outside the forest. This initiative failed as local herders tended to continue their traditional approach of forest grazing (Makhdoum 2008). Livestock continued to graze even in sites of FRWO forest recovery projects, thus depressing forest regeneration (Sagheb-Talebi et al. 2014). Because of simultaneous trampling and overgrazing, the Hyrcanian forest cannot supply sufficient fodder for livestock as well as for wild ungulates and its current economic use is unsustainable (Noack et al. 2013).

Moreover, in the past three decades the Hyrcanian forest was adversely affected by large-scale shelterwood cutting. Most recently, in 2013 FRWO has enacted a law that permitted the removal of deadwood from the Hyrcanian forest (Müller et al. 2015; Müller et al. 2017). This practice has been recently proven by Müller et al. (2015) to negatively affect the diversity of endemic invertebrates (beetles) and ecosystem stability (Müller et al. 2015). This may also severely affect the occurrence of wide-ranging mammals, especially threatened species and habitat specialists (Laurance et al. 2008; Brodie et al. 2015; Ripple et al. 2015). For this reason, overgrazing and logging imply various conservation challenges and the scientific research of the effects of these threats on species ecology and distribution is of high priority for science and conservation.

Hunting has an age-old history in Iran, but since 1956 when a hunting ban was introduced it is permitted only under special licenses (Firouz 2005; Ghoddousi et al. 2017a). Nevertheless, a recent study has attributed to hunting severe population declines of ungulates in the oldest protected area of Iran, estimated at 69-89% since the 1970s (Ghoddousi et al. 2017a). Especially in this context, the impacts of poaching and overgrazing on large native mammalian species in the whole region of the Hyrcanian forest are of major importance.

Since the status of large mammals in this region still remains largely overlooked by scientists and conservationists, I have chosen the Hyrcanian forest for this research project. Its relict evolutionary history and potential stronghold for a diverse ungulate and carnivore community under the pressure of poaching, livestock grazing and logging, makes it both interesting from a management as well as research standpoint.

1.6. Thesis objectives and achievements

The aim of this research was to assess the effects of anthropogenic threats (i.e. poaching, livestock grazing and logging) on the distribution of large mammal species (i.e. Persian leopard, grey wolf, Caspian red deer, roe deer, bezoar goat and wild boar) in the Hyrcanian forest, Iran. Due to the potential synergistic effects of hypothesized threats, the multidisciplinary nature of conservation research and the lack of baseline information, there was a dire need to conduct basic in-depth studies in this region. In order to meet the objectives, I quantified the effects of threats on the distribution of the large mammal community (Chapter 2), population ecology of wild prey in regard to poachers' attitudes and incentives (Chapters 3 and 4) and the effects of human agricultural strategies (e.g. crop variety) and the state of wild ungulate populations on human-wildlife conflicts (Chapter 6). The aims and achievements of this study are as follows:

(i) ***Assessment of the principal human-induced threats to the occurrence of large mammal species across the Hyrcanian forest*** (Chapter 2: Livestock grazing in protected areas and its effects on large mammals in the Hyrcanian forest, Iran)

To reach the overall aim of this study, in Chapter 2 I estimated the probabilities of site (grid cell) occupancy and detection for all targeted mammal species (Persian leopard, grey wolf, brown bear, Caspian red deer, roe deer, bezoar goat and wild boar) throughout the 18 protected and non-protected areas. I used the novel Bayesian single-season occupancy modelling and made inferences on covariates to account for uncertainty in model parameters. These inferences are valid irrespective of sample size because they do not depend on asymptotic approximations. I also measured the effectiveness of protected areas in regard to the reduction of grazing pressure and quantified local extinction of threatened species across the Hyrcanian forest.

I used a novel approach, the Bayesian single-season occupancy model (MacKenzie et al. 2006; Kery & Royle 2016) to quantify the magnitude of anthropogenic pressures (i.e. poaching, livestock grazing and logging) on the occurrence of targeted large mammal species across 18 protected and non-protected areas in the Hyrcanian forest (Chapter 2). By application of the occupancy approach, I estimated the status of native threatened species and the key conservation areas at the large landscape scale which has rarely been tested, particularly in rugged terrain where robust information is often lacking. However, I was not able to illustrate the real magnitude of poaching pressure on the distribution of targeted species. Probably, poaching tends to act in areas where animals are available, resulting in a positive correlation between poaching and species population.

I also assessed the effectiveness of protected areas in regards to the reduction of livestock grazing pressure. The results showed that grazing in national parks and wildlife refuges was notably lower compared to protected areas, non-hunting areas and non-protected areas. Moreover, the fragmented distribution of sensitive species, such as leopard, red deer and roe deer, may reflect systemic failures of management and law enforcement due to budget constraints. Even common species may need stronger conservation actions as wolves and wild boar have been intensively persecuted for livestock and crop damage, respectively.

(ii) ***Estimation of wild prey abundance and assessment of poaching pressure*** (Chapter 3: The precision and reliability of indirect population assessments for the Caspian red deer (*Cervus elaphus maral*) and Chapter 4: The decline of ungulate population in Iranian protected areas calls for urgent actions against poaching)

Quantification of poachers and leopard's hunting/prey preferences requires information on abundance of wild prey and also depends on reliable and accurate monitoring techniques. In Chapters 3 and 4, I applied various techniques (random encounter model, line-transects, faecal standing crop and point counts) to estimate the population densities of Caspian red deer, roe deer, bezoar goat and wild boar. I also investigated poachers' incentives using semi-structured interviews in order to understand the causal effects of poaching among local communities.

In Chapters 3 and 4, I estimated the abundance of four main prey species including the Caspian red deer, bezoar goat and wild boar in Golestan National Park (GNP), which are the target species of poachers and leopards. I estimated the abundance of rare Caspian red deer using a novel cost-effective method, the random encounter model based on camera trap data (91-423 individuals) and dung counts based on the faecal standing crop method (103-258 individuals). The results showed that the Caspian red deer population has dropped by 78-96% compared to its prior population size in the 1970s. The bezoar goat population was estimated using double-observer point counts (201-807 individuals) and showed a sharp decline by 79-96% compared to the 1970s. The urial abundance (2117-8632 individuals) was estimated by distance sampling using line transects. These estimates showed a 14-86% drop compared to records from the 1970s. Wild boar was the only prey species showing a population increase by 2-75% compared to the earlier estimates.

Logging is not an issue in GNP and livestock grazing is prevalent only along the park boundaries. Therefore, poaching was identified as the main culprit for a sharp decline of ungulates in GNP.

(iii) ***Assessment of poaching impacts on human-wildlife conflicts*** (Chapter 5: Assessing the relationship between poaching intensity, wild prey occurrence and livestock depredation rate by large carnivores and Chapter 6: Crop variety and wild prey richness affect special patterns of human-wildlife conflict in Iran's Hyrcanian forest)

In Chapter 5, I estimated the relative occurrence of the main wild prey species, namely the Caspian red deer, roe deer, bezoar goat and wild boar, as well as of domestic prey (i.e. cattle and sheep/goats), and concurrently documented the relative occurrence of poaching signs by walking 1204 km along the trails in designated grid cells. I conducted 201 interview surveys with herders encountered randomly inside each cell and documented livestock losses over three discrete surveys across the Hyrcanian forest landscape. Eventually, I quantified the effects of prey poaching on livestock depredation by Persian leopard and grey wolf in the Hyrcanian forest.

Lastly, using interviews from 162 households (45 villages) among local communities of three provinces (Golestan, Mazandaran and Gilan), I documented livestock and crop losses by large mammals to quantify the human-wildlife conflicts (Chapter 5).

In Chapter 5, I assessed the effects of prey poaching on livestock depredation by large carnivores (Persian leopard and grey wolf). There were negative correlations between the occurrence of signs of prey poaching and the abundance of wild prey species, but that was significant only for the threatened Caspian red deer and roe deer. Livestock abundance exceeded that of wild prey. Leopards were more likely to prey on cattle (79%) than on sheep/goat (21%) in proximity to villages. Contrarily, grey wolves depredated more on sheep/goat (96%), but rarely preyed upon cattle (4%). Avoidance of forest by wolves is probably related to the ecology of this predator which is mainly adapted to the steppe areas. The differences in predatory behaviour of these carnivores might also be driven by herding practices, as sheep/goats are much more abundant on plateaus above the tree-line, but cattle are more available in forested areas. Presumably, leopards function as the apex predator and may suppress wolves to prey more on small stock.

Assessing the effectiveness of PAs I found out that national parks (IUCN category II) may have significantly lower levels of poaching than other reserves. However, the NPs cover only ~0.01% of the total size of reserves in the Hyrcanian forest. By contrast, protected areas of category V represent 66% of reserves in the region. The combination of results from questionnaire surveys on livestock and crop losses and the assessed prey richness has shown that the frequency of wild boar conflicts is related to high abundance of this species, which may be due to religious beliefs that prohibit pork consumption (Chapter 4). Intensive human-wild boar conflict was due to a greater crop variety and larger crop fields. Wolf predation on sheep/goat was not influenced by small stock numbers. Instead, livestock depredation by wolves was likely to increase with increasing elevation, deficiency of wild prey richness, and in areas located closer to, or inside, PAs.

Chapter 2

Livestock grazing in protected areas and its effects on large mammals in the Hyrcanian forest, Iran

Chapter 2: Livestock grazing in protected areas and its effects on large mammals in the Hyrcanian forest, Iran

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Livestock grazing in the Hyrcanian forest (Baliran). *Photo M. Soofi*

Abstract

Protected areas are the most important tool to safeguard large mammals from overexploitation, but their effectiveness is insufficiently studied in temperate ecosystems. The Hyrcanian forest is one of the oldest and most threatened temperate forests globally. Anthropogenic activities are widespread and negatively affect wildlife species in the Hyrcanian forest. We conducted surveys in ~22% of the Hyrcanian forest by walking 1204 km in 93 16-km^2 cells distributed randomly in 18 protected and non-protected study sites. We used Bayesian occupancy modeling to measure the effects of livestock grazing, logging and poaching on distribution of six large mammal species. Our results explicitly show that grazing had negative and significant impact (β = -1.65, Credibility Interval - 2.85 to -0.65) on the occupancy of very patchily distributed Persian leopard, Caspian red deer (β = -1.36, CI -2.34 to -0.45) and roe deer (β = -1.61, CI -2.96 to -0.58) while logging did so for red deer (β = -0.82, CI -1.69 to -0.03). Poaching could not be determined due to low detectability of poaching signs. Grazing intensity was high in protected areas (IUCN category V), no-hunting and non-protected areas and much lower in national parks (II) and wildlife refuges (IV). Representing 66% of total reserves in the Hyrcanian forest, category V protected areas urgently require priority actions in assessment of grazing capacities, allocation and enforcement of grazing quotas, and better coordination between governmental conservation and natural resource management organizations to avoid further depletion of the large mammal community in the Hyrcanian Forest.

Keywords: Bayesian occupancy, Caspian, law enforcement, logging, poaching, protected areas

2.1. Introduction

Protected areas are the cornerstone of conservation, but many of them lose rare and ecologically sensitive large mammals at alarming rates due to insufficient size and poor protection from overexploitation and other threats (Watson et al., 2014; Maxwell et al., 2016). Albeit many studies reporting local species extirpations from logging, grazing and poaching in tropical regions, the effects of these threats on temperate ecosystems remain understudied (Brodie et al., 2015) since most temperate forests have already lost many large species.

Livestock grazing, logging and poaching are among the main drivers of biodiversity loss but their effects can be both synergistic and contrasting across different species (Brodie et al., 2015; Maxwell et al., 2016). For example, logging and grazing may improve food supply for predators but also provoke human-predator conflicts and poaching (Laurance et al., 2008).

Livestock grazing inflicts intense landscape degradation and has multiple effects on large mammal distributions (Karanth et al., 2011; Ripple et al., 2014, 2015). Livestock causes large-scale changes in vegetation structure and adversely affects native herbivores via trophic competition (Maxwell et al., 2016; Gordon et al., 2017). Logging simplifies the complexity of forest ecosystems and reduces habitat quality (Müller et al., 2016). In addition, logging and grazing contribute to road development which increases habitat accessibility to poachers, thus exerting substantial effects on the survival of large mammals (Laurance et al. 2008; Brodie et al., 2015; Maxwell et al., 2016).

The Hyrcanian forest (hereafter, HF) located in Iran and Azerbaijan is a Tertiary relict temperate forest and of high conservation value due to the exceptional diversity of landscapes and species converging between Asia, Europe and Africa (Fig. 1). It is part of the Caucasus Biodiversity Hotspot and harbors a diverse community of large mammals, such as the Persian leopard (*Panthera pardus saxicolor* Pocock, 1927), brown bear (*Ursus arctos* Linnaeus, 1758), grey wolf (*Canis lupus* Linnaeus, 1758), Caspian red deer (*Cervus elaphus maral* Ogilby, 1840), roe deer (*Capreolus capreolus* Linnaeus, 1758) and wild boar (*Sus scrofa* Linnaeus, 1758) (Olson & Dinerstein 1998; Firouz 2005). The last Caspian tiger (*Panthera tigris virgata*) was killed in 1953 in the Hyrcanian forest (Firouz, 2005). Sixty percent of the HF is under legal protection and natural resource use is managed by the government (Zehzad et al., 2002; Firouz, 2005; Makhdoum, 2008; Dabiri et al., 2010; Müller et al., 2017).

Several laws to protect plant biodiversity in Iran’s forests have been implemented, such as the forest nationalization law (1963), the law banning livestock grazing inside core zones of protected areas and wildlife refuges (1982) and the law on livestock exclusion from all HF (1989). Since 1956, hunting inside protected areas is permitted only under special licenses (Firouz, 2005). Despite these legislative acts, human activities such as grazing, logging, poaching and wood collection are widespread and unorganized in the HF (Firouz, 2005; Makhdoum, 2008; Sagheb-Talebi et al., 2014; Ghoddousi et al., 2017a; Müller et al., 2017). Due to overexploitation, the forest cover of Iran has halved during the past five decades (Ghoddousi et al., 2017a). Nowadays, about 4 million livestock are roaming across the HF, leading to overgrazing (Sagheb-Talebi et al., 2014), deterioration of forest regeneration and forest recessions, especially in lowlands (Akhani et al., 2010). The Hyrcanian forest cannot supply sufficient fodder for livestock and its current economic use is unsustainable (Noack et al., 2010). In Golestan National Park, Iran's oldest reserve, the red deer population has declined by 89% since the 1970s due to poaching motivated by subsistence, leisure and hostility toward park staff and conservation laws (Ghoddousi et al., 2017b).

Whilst understanding of the effects of human threats on the distribution of large mammals is among the top conservation priorities in this region, it largely remains overlooked by scientists and conservationists. The paucity of information and conservation guidance is particularly evident at large scales, which is critical considering the spatial requirements of populations of these species (Ripple et al., 2015). In this study, we combined intensive field surveys and Bayesian occupancy modeling to document the effects of overgrazing, logging and poaching on the distribution of six large mammal species throughout the HF. We also assessed the efficiency of protected area categories in preservation of large mammals. Further, we discuss the management actions required to address declines of large mammals in the Hyrcanian forest.

2.2. Material and Methods

Study area

The Hyrcanian forest forms a green arc along the Caspian Sea. It expands from the Talysh Mountains in Azerbaijan through the northern slopes of the Alborz Mountains to Gollidagh in eastern Iran with elevations ranging from -28 to 2800 m. The mean annual precipitation ranges from 530 to 1350 mm, occasionally reaching up to 2000 mm in the western parts. The mean air temperature of the warmest and coldest months varies from 28-35°C to 1.5-4°C, respectively. The lowland forests are dominated by *Zelcova carpinifolia, Gleditsia caspica* and *Pterocarya fraxinifolia* with regular presence of *Parrotia persica*. In montane areas, tree dominance shifts to *Quercus castaneifolia*, *Carpinus betulus*, *Fagus orientalis* and *Quercus macranthera* depending on temperature regimes. The forest understory is covered mainly by *Ruscus hyrcanus*, *Ilex spinigera*, *Buxus hyrcana* and ferns (Sagheb-Talebi et al., 2014).

Study design

We assessed the impact of anthropogenic threats on the Persian leopard, brown bear, grey wolf, Caspian red deer, roe deer, and wild boar. We selected 18 study areas, covering 4015.60 km^2 and including three national parks (NP), eight protected areas (PA), one wildlife refuge (WR), two no-hunting areas (NHA) and four non-protected areas (NPA) throughout the HF (Fig. 1). We placed a regular grid of 4x4 km cells over the study areas using the Hawth's Tools in ArcGIS 10.2 (ESRI Co., USA). Cell size was based on an approximate average home range size of all target species (Yackulic et al., 2011; Kiffner et al., 2013). For surveys, we randomly selected ~45% of the total number of cells in each study area.

The single-season occupancy framework assumes that the occupancy state of the species does not change in a site within a season (MacKenzie et al., 2006) and we assumed that our survey periods were short enough to comply with this assumption. We considered surveyed cells as sites and the entire period of surveys as a season. During three survey periods (August-October 2015, February-April 2016 and August-October 2016), we surveyed most cells by a team of 2-3 people led by an experienced ranger or a local guide who could unambiguously identify signs of target species and anthropogenic threats. We walked along random trails of 2-13 km inside each selected cell and recorded the presence of fresh signs (tracks, scratches, scrapes, feeding and resting places, and wallows) and direct observations (sightings and sounds) of species at 200 m intervals (Karanth et al., 2011). Concurrently, we recorded the occurrence of anthropogenic threats such as the signs of poaching (encounters with poachers, gun shells, gunshots), logging (cut trees, logging activities), and livestock grazing (cattle, sheep, goats and domestic dogs). Each survey team took photographs of animal and threat signs for final identification. To minimize the observer bias, we rotated team members between study areas and sites (MacKenzie et al., 2006).

Analysis

We used the presence (1) and absence (0) data on each species across cells as the response variables. The intensities of logging, poaching and livestock grazing represented the predictors. These intensities were quantified as the proportions of the number of 200-m trail segments with signs to sampling effort (km of trails walked per cell and survey). Additionally, we considered sampling effort as a predictor of detection probability (MacKenzie et al., 2006). We calculated Spearman's *rho* for rank correlation among predictors and used Mann-Whitney test to compare grazing intensities among the study areas with different protection levels. We took the IUCN categories of study areas from Protected Planet (www.protectedplanet.net). For each species, we quantified the effects of threats on their occupancy probability ψ while simultaneously accounting for imperfect detection and sampling efforts. Specifically, ψ of each species in cell *i* was described as:

$$\text{logit}(\psi_i) = \alpha_\psi + \beta_{livestock} X_{livestock\,i} + \beta_{\log ging} X_{\log ging_i} + \beta_{poaching} X_{poaching_i}$$

To assess ψ by the observed presence-absence data for each species, we modeled the probability of true occurrence *z* of each species in cell *i* as a random variable derived from the Bernoulli distribution with probability ψ:

$$z_i \sim Bernoulli(\psi)$$

Occupancy models treat the observed presence (or absence) of a species at survey *j* as an outcome of a detection process, i.e. a random Bernoulli variable defined by *z* and the sign detection probability *p*:

$$y_{ij} \sim Bernoulli(z_i \times p_{ij})$$

The quantification of detection probability *p* allows including possible impacts of bias arising from variability in sampling effort:

$$\text{logit}(p_{ij}) = \alpha_p + \beta_{effort} y_{ij}$$

We used the Bayesian occupancy modeling in R2JAGAS package of R (Plummer, 2003; Su and Yajima, 2015; R Core Team 2016; see models in Appendix 1). Apart from adaptability to low sample sizes, the Bayesian framework offers flexibility in regard to missing observations (Kéry, 2010; Dorazio and Rodríguez., 2012). Threat effects on species occupancy were assessed from the posterior distributions of the intercept *α* and slope *β*. The direction of threat effects was determined from positive or negative estimates of *β*. The significance of difference of threat effects from 0 (no effect) was assessed from the overlap of the credibility interval (CI) with 0. The CI ranges between 2.5 and 97.5 percentile of the posterior distribution. We ran three chains with 100 000 iterations to assess the posterior distribution of the coefficients from the estimation of their prior distribution. We chose a vague prior from the uniform distribution with the boundary estimates of α and β from -10 to 10 (Kéry, 2010). The first 20000 iterations were discarded. Chains were thinned to every 40th value of the iteration to avoid autocorrelation. Convergence of three chains was assured by Gelman-Rubin statistics (Gelman et al., 2014) and achieving a minimum effective posterior sample size of 100 (Kéry, 2010).

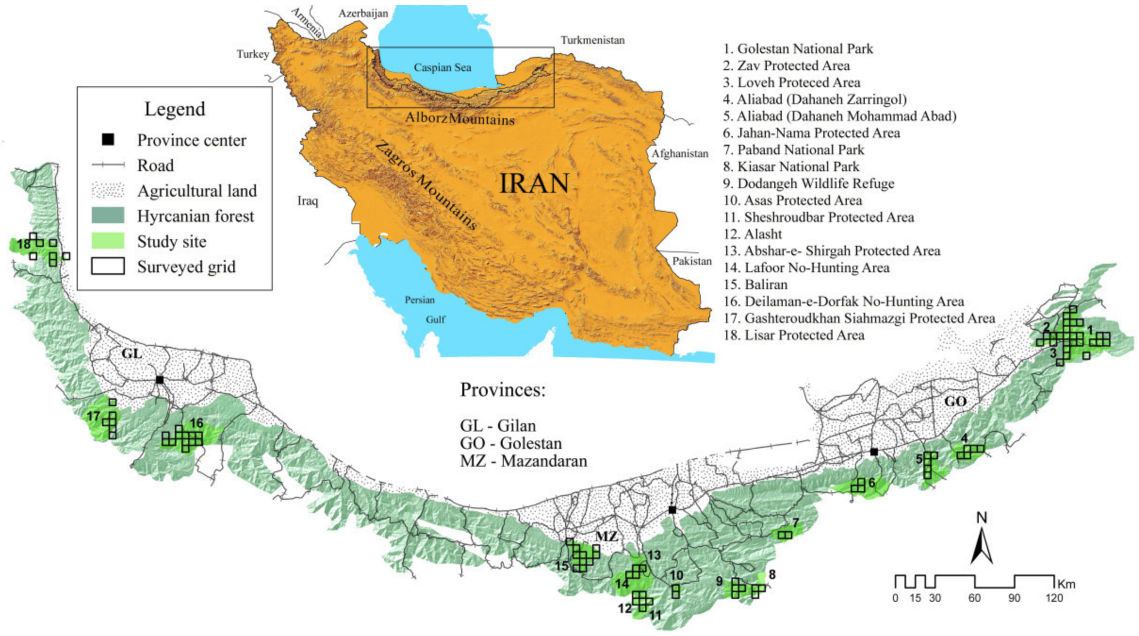

Fig. 1. The map of the study areas across the Hyrcanian forest, northern Iran.

2.3. Results

We walked 1204 km of trails during 147 field days and recorded 2876 signs of six mammal species (Appendix 1). Overall, we surveyed 93 cells, of which 45 cells were surveyed three times, 21 twice and 27 once for logistical reasons (Table 1). The intensities of grazing and logging were most correlated (r = 0.59), followed by logging and poaching (0.39), and grazing and poaching (0.37).

Signs of both roe deer and red deer were absent in Zav PA, Lisar PA and Lafoor NHA (Fig. 1). The roe deer was absent in Alasht. The Persian leopard was absent in Paband NP and Lisar PA. The grey wolf and red deer were absent in Abshar-e-Shirgah PA. Wild boar and brown bear were present in all sites. Grazing had the highest intensity (0.92, CI 0.78 to 1.05), logging had intermediate (0.52, CI 0.42 to 0.62) and poaching had the lowest (0.14, CI 0.11 to 0.18).

The leopard had a moderate detection probability (p = 0.70, CI 0.61 to 0.77), but fragmented distribution (ψ = 0.88, CI 0.27 to 0.99). Leopard occupancy was negatively affected by grazing (β = -1.65, CI - 2.85 to -0.65) (Fig. 2). The gray wolf had the lowest detection probability regardless of effort (p = 0.25, CI 0.18 to 0.34), but it was present in all study areas (ψ = 1, CI 0.81 to 1). The brown bear was present in all study areas (ψ = 0.99, CI 0.51 to 1) and had a moderate detection probability (p = 0.62, CI 0.54 to 0.71), which increased with effort (β = 0.38, CI 0.04 to 0.75; Fig. 2). The red deer had very fragmented distribution (ψ = 0.71, CI 0.13 to 0.97), but a moderate detection probability (p = 0.78, CI 0.70 to 0.86). Red deer occupancy strongly decreased with grazing (β = -1.36, CI -2.34 to -0.45) and logging (β = -0.82, CI -1.69 to -0.03) (Fig. 2). Compared to other studied species, roe deer had the most limited and highly fragmented distribution (ψ = 0.67, CI 0.10 to 0.97), with low detection probability (p = 0.55, CI 0.43 to 0.67). Roe deer occupancy was negatively affected by grazing (β = -1.61, CI -2.96 to -0.58). Wild boar was the most widespread and highly detectable species (ψ = 1, CI 0.80 to 1; p = 0.95, CI 0.91 to 0.98) and its detection probability significantly increased with effort (β = 1.47, CI 0.55 to 2.55 (for detailed models and data see Appendix 2). Grazing intensity was significantly higher in PA vs. NP (F-value = 6.18, $p < 0.001$), NPA vs. NP (F-value = 5.70, $p < 0.0018$), and NHA vs. NP (F-value = -2.99, p = 0.04) (Fig. 3).

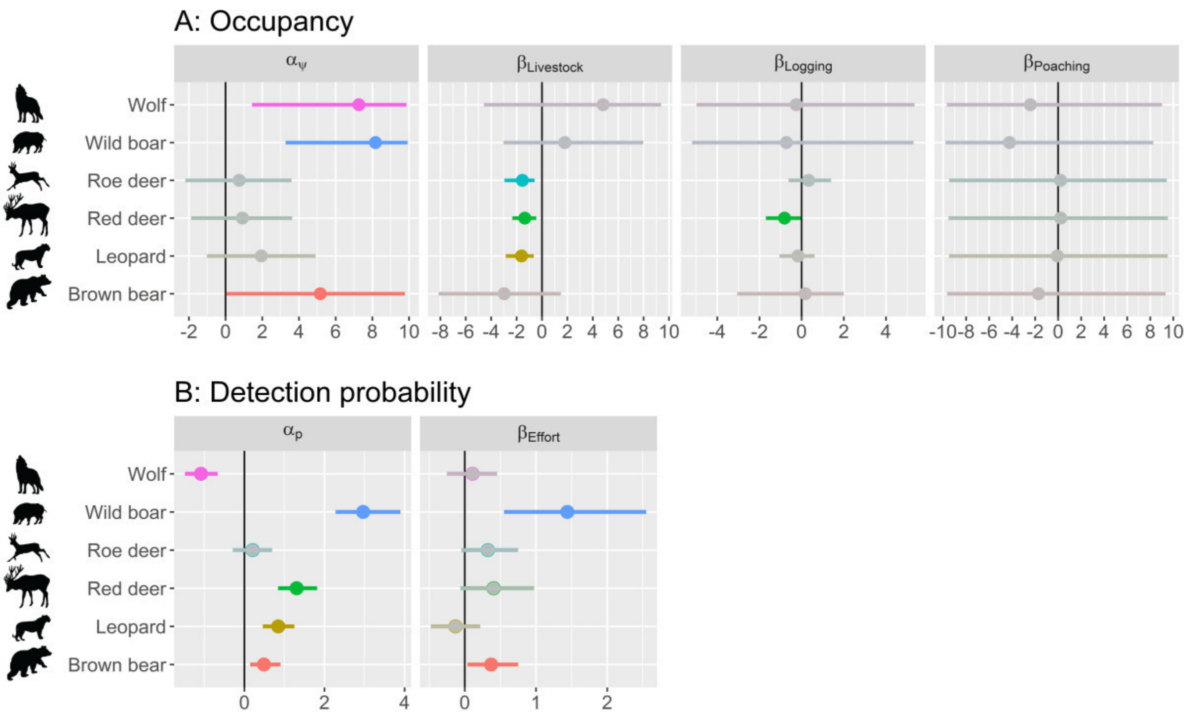

Fig. 2. The alpha (intercept) and beta (slope) coefficients of Bayesian single-season occupancy models and their 95% credibility intervals estimated for six large mammal species in the Hyrcanian forest. The credibility intervals intersecting with zero are shaded.

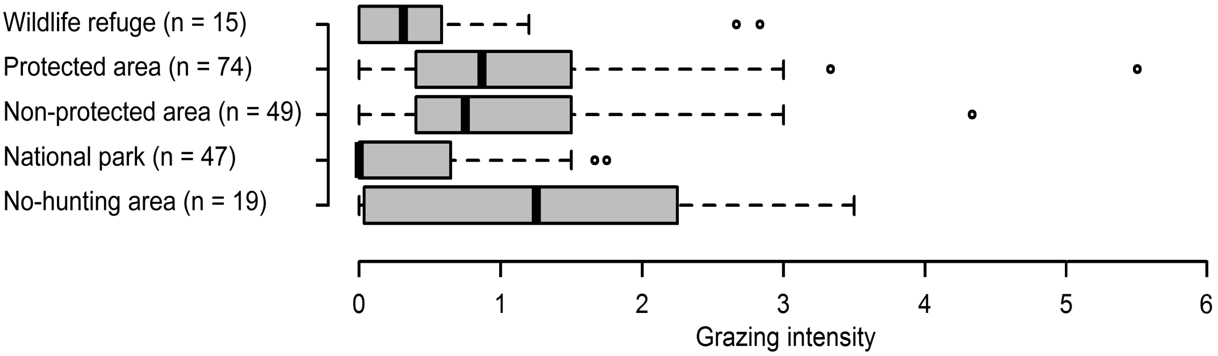

Fig. 3. Comparison of livestock grazing intensities across 18 study areas in the Hyrcanian forest. Circles indicate the outliers of the grazing intensity from individual field surveys. Abbreviations: NHA – no-hunting area (n = 19 grid cells), NP – national park (n = 47), NPA – non-protected area (n = 49), PA – protected area (n = 74), WR – wildlife refuge (n = 15).

Table 1. The distribution of anthropogenic threats in study areas throughout the Hyrcanian forest

Abbreviations: IUCN – International Union for Nature Conservation, NHA – no-hunting area, NP – national park,

Area	Size (km2)	No. cells	IUCN category	Proportion of grid cells with presence of anthropogenic threats		
				Grazing	Logging	Poaching
Golestan NP	874.02	14	II	0.37	0	0.12
Zav (A & B) PA	143.23	8	NR	1	0.57	0.70
Loveh PA	33.49	3	NR	0.89	0.78	0.33
Aliabad (Dahane Zarringol) NPA	121.67	5	-	0.89	0.78	0.11
Aliabad (Dahaneh Mohamm Adabad) NPA	82.94	5	-	1	0.80	0.20
JahanNama PA	317.47	3	V	0.89	0.22	0
Paband NP	181.45	2	NR	1	0.50	0.50
Kiasar NP	92.65	2	-	1	0.50	0.50
Dodangeh WR	169.04	5	IV	0.60	0.53	0.53
Asas PA	29.97	2	V	1	1	1
Sheshroudbar PA	79.22	2	NR	1	0	0.50
Abshar-e-Shirgah PA	36.39	1	V	1	0.50	0.50
Lafoor NHA	363.52	3	-	1	1	1
Alasht NPA	129.11	3	-	1	0	0.33
Baliran NPA	206	10	-	0.96	0.93	0.52
Deilaman-e-Dorfak NHA	448.86	10	-	0.64	0.64	0.57
Gashteroudkhan-Siahmazgy PA	395.14	8	V	1	0.17	0.50
Lisar PA	311.42	7	V	1	0.79	0.63
Total	4015.60	93	-	0.90	0.54	0.47

NPA – non-protected area, NR – not reported, PA – protected area, WR – wildlife refuge.

2.4. Discussion

Our results show that livestock grazing strongly and negatively affects the distribution of the Persian leopard, Caspian red deer and roe deer in the HF. These species are threatened either globally or nationally, and have very patchy distributions in this region. The two deer species and the leopard appear to be locally extinct in some study areas. The fact that most of these areas are officially protected and located within the distribution areas of these species raises a serious concern regarding the effectiveness of conservation efforts in the HF. Our study region is believed to be the stronghold for survival of the globally endangered Persian leopard in the Middle East (Kiabi et al., 2002; Farhadinia et al., 2015), but our results suggest a high degree of fragmentation of its population. Additionally, our results suggest that the red deer is under persistent pressure from logging which may facilitate access of poachers to core zones and lead to increased poaching (Laurance et al., 2008; Brodie et al., 2015). We did not find significant effects of poaching on red deer or other species, possibly due to low detectability of poaching signs (Brodie et al., 2015; Rauset et al., 2016). In contrast, fine-scale studies demonstrate drastic declines of red deer due to poaching, e.g. in Golestan National Park by 89% from 2096 individuals in 1976-1977 to 194-257 individuals in 2015-2016 (Kiabi et al., 2004; Ghoddousi et al., 2017b; Soofi et al., 2017). Possibly, the count of poaching signs is an inappropriate metric of poaching pressure because poachers tend to act in areas where animals are available, resulting in a positive correlation between poaching and prey populations (Brodie et al., 2015). Moreover, poaching can go undetected in forests due to dense vegetation, litter and secretive trails (Laurance et al., 2008).

We demonstrate that livestock grazing is the main threat affecting large mammal distribution in the HF. Therefore, it should be effectively managed through the assessment of the carrying capacity of pastures, allocation of grazing quotas and their enforcement. Local people still strongly depend on forest for pastures during the snow-free seasons. Since 1982, grazing has been permitted in 80% of the territories of protected areas (IUCN category V) and wildlife refuges (IUCN category IV), putting these reserves under serious pressure of overgrazing. We confirmed high levels of grazing in protected areas, but not in the wildlife refuge. Category V protected areas represent about 66% of the total coverage of reserves in the HF compared to only 0.01% of wildlife refuges and 0.10% of national parks.

Herders hold official permits with specified sizes of pastures and grazing periods, but often overuse pasture lands and penetrate deep into the core zones under non-existing land allotments and inefficient governmental control. Such large-scale encroachment makes large mammals retreat into non-protected lands and clash with rural people (Farhadinia et al., 2015; Khorozyan et al., 2015).

Grazing control is impossible without the enforcement of better coordination between the Iranian governmental organizations responsible for conservation (Department of Environment, DoE) and natural resource management (Forest, Rangeland and Watersheds Organization, FRWO). Traditionally, DoE is responsible for the control of non-compliance activities inside reserves, but the enforcement of logging and grazing control inside and outside reserves is under the credentials of FRWO (Makhdoum, 2008; Dabiri et al., 2010; Kolahi et al., 2012). However, interests and management strategies of the two agencies often collide in protected areas and wildlife refuges. There is no clear separation of responsibilities of DoE and FRWO in these areas, where grazing is occurring on 80% of lands and prohibited in core zones covering only 20% (Makhdoum, 2008). The same situation is in national parks where DoE and FRWO lack cooperation and coordination in managing illegal grazing and logging. Poaching control is the responsibility of DoE alone. Thus, there is much uncertainty in mechanisms of cooperation between these two organizations and the development of inter-agency policy is a priority need. Inadequate cooperation between DoE and FRWO can be illustrated by the example of adverse effects of logging on red deer. Red deer is the only studied large mammal strongly preferring mixed forests with dense shrubs (Kiabi et al., 2004), but its populations suffer from habitat deterioration caused by the even-aged tree management system and removal of fallen or dead woods (Sagheb-Talebi et al., 2014; Müller et al., 2017).

Conclusions

We conclude that the existing governmental actions are insufficient to alleviate the pressure of human activities on large mammals in the Hyrcanian Forest. Fragmented distribution of such sensitive species as the leopard, red deer and roe deer may reflect systemic failures of management, law enforcement and budget constraints (Watson et al., 2014; Rauset et al., 2016) while the satisfactory status of grey wolf, brown bear and wild boar is achievable due to their high tolerance to humans. However, even these common species may need stronger conservation action as wolves and wild boars have been intensively persecuted for livestock and crop damage, respectively (Ripple et al., 2014). We emphasize the need for stricter law enforcement regarding overgrazing and poaching under the consideration of improvements of rural livelihoods. Furthermore, clear land use zoning of reserves should be developed and stringently managed (Kolahi et al., 2012). All these efforts should be participatory to minimize conflicts with local communities (Rauset et al., 2016) and coordinated by DoE and FRWO.

2.5. Supplementary material

Appendix Table 1

The number of signs recorded for six large mammal species in protected and non-protected areas in the Hyrcanian forest, Iran.

Area	Leopard	Brown bear	Red deer	Roe deer	Wild boar	Wolf
Golestan NP	78	156	150	70	442	6
Zav PA (A & B)	20	25	0	2	78	4
Loveh PA	10	26	40	12	93	4
Aliabad (Z & M)	23	9	19	21	34	5
JahanNama PA	30	10	126	3	26	3
Paband NP	0	4	9	0	23	6
Kiasar NP	11	23	40	12	37	5
Dodangeh WR	58	32	147	24	112	7
Asas PA	6	4	5	3	8	2
Sheshroudbar PA	1	1	2	1	1	0
Abshar-e-Shirgah PA	6	6	0	1	1	1
Lafoor NHA	7	2	0	0	17	4
Alasht	20	10	2	0	7	11
Baliran	63	29	26	22	155	8
Deilaman-e-Dorfak NHA	47	15	41	21	97	6
Gashteroudkhan Siahmazgy PA	7	9	3	1	12	5
Lisar PA	0	8	0	0	82	9
Total	387	369	610	193	1231	86

Appendix Table 2

The parameter estimates and quality measures of Bayesian single-season occupancy models for large mammal species in the Hyrcanian forest. Rhat and n.eff provide information on model convergence (Rhat = 1) and the effective size of the posterior distribution (n.eff).

Species	Parameters	Mean (SD)	Percentiles of posterior distribution						
			2.50 %	25 %	50 %	75 %	97.50	Rhat	n.eff
Persian leopard *Panthera pardus saxicolor*	α_{ψ}	1.95 (1.73)	-1.02	0.50	1.95	3.38	4.91	1.00	6000
	α_{p}	0.85 (0.20)	0.46	0.71	0.85	0.98	1.25	1.00	6000
	$\beta_{livestock}$	-1.65 (0.57)	-2.85	-1.98	-1.61	-1.27	-0.65	1.00	6000
	$\beta_{logging}$	-0.18 (0.42)	-1.05	-0.44	-0.17	0.10	0.62	1.00	6000
	$\beta_{poaching}$	-0.05 (5.83)	-9.49	-5.19	-0.07	5.01	9.54	1.00	6000
	β_{effort}	-0.13 (0.18)	-0.48	-0.25	-0.13	-0.02	0.22	1.00	6000
Grey wolf	α_{ψ}	6.81 (2.23)	1.44	5.35	7.27	8.69	9.87	1.00	3300
Canis lupus	α_{p}	-1.08 (0.21)	-1.48	-1.22	-1.08	-0.94	-0.66	1.00	4000
	$\beta_{livestock}$	4.34 (3.39)	-4.57	2.73	4.81	6.71	9.38	1.00	6000
	$\beta_{logging}$	-0.16 (2.60)	-4.99	-1.89	-0.26	1.41	5.35	1.00	6000
	$\beta_{poaching}$	-1.72 (5.54)	-9.68	-6.53	-2.41	2.57	9.03	1.00	5100
Brown bear *Ursus arctos*	α_{ψ}	0.11 (0.18)	-0.25	-0.01	0.11	0.23	0.45	1.00	2600
	α_{p}	5.23 (2.84)	0.04	3.00	5.17	7.73	9.79	1.00	3800
	$\beta_{livestock}$	0.50 (0.20)	0.15	0.37	0.49	0.63	0.90	1.00	1200
	$\beta_{logging}$	-3.37 (2.53)	-8.13	-5.13	-2.98	-1.77	1.49	1.00	4100
	$\beta_{poaching}$	0.08 (1.26)	-3.05	-0.32	0.17	0.65	2.00	1.01	2700
	β_{effort}	-1.15 (5.77)	-9.66	-6.22	-1.72	3.59	9.34	1.00	3400
	α_{ψ}	0.38 (0.18)	0.04	0.25	0.37	0.49	0.75	1.00	4800

Appendix Table 2 continued.

Species	Parameters	Median (SD)	Percentiles of posterior distribution						
			2.50 %	25 %	50 %	75%	97.50 %	Rhat	n.eff
Wild boar	α_{ψ}	7.73 (1.82)	3.27	6.68	8.18	9.21	9.93	1.00	6000
Sus scrofa	α_{p}	3.00 (0.42)	2.28	2.70	2.96	3.26	3.90	1.00	3900
	$\beta_{livestock}$	2.10 (2.96)	-3.03	-0.15	1.80	4.30	7.97	1.00	6000
	$\beta_{logging}$	-0.54 (2.70)	-5.19	-2.53	-0.72	1.16	5.31	1.00	6000
	$\beta_{poaching}$	-3.11 (5.20)	-9.79	-7.54	-4.22	0.56	8.27	1.00	6000
	β_{effort}	1.47 (0.51)	0.55	1.11	1.44	1.80	2.55	1.00	4000
Red deer	α_{ψ}	0.89 (1.63)	-1.88	-0.49	0.92	2.26	3.63	1.00	6000
Cervus elaphus maral	α_{p}	1.31 (0.25)	0.84	1.14	1.31	1.48	1.82	1.00	3700
	$\beta_{livestock}$	-1.36 (0.48)	-2.34	-1.67	-1.35	-1.04	-0.45	1.00	4800
	$\beta_{logging}$	- 0.82 (0.42)	-1.69	-1.09	-0.81	-0.55	-0.03	1.00	6000
	$\beta_{poaching}$	0.11 (5.77)	-9.53	-4.89	0.22	5.02	9.54	1.00	5100
	β_{effort}	0.42 (0.26)	-0.06	0.24	0.40	0.59	0.97	1.00	6000
Roe deer	α_{ψ}	0.70 (1.68)	-2.20	-0.68	0.74	2.06	3.60	1.00	4400
Capreolus capreolus	α_{p}	0.21 (0.25)	-0.29	0.04	0.21	0.38	0.69	1.00	6000
	$\beta_{livestock}$	-1.61 (0.63)	-2.96	-1.95	-1.54	-1.19	-0.58	1.00	5500
	$\beta_{logging}$	0.35 (0.52)	-0.62	0.02	0.33	0.66	1.40	1.00	6000
	$\beta_{poaching}$	0.04 (5.73)	-9.48	-4.91	0.21	4.90	9.45	1.00	3700
	β_{effort}	0.33 (0.20)	-0.05	0.19	0.32	0.46	0.75	1.00	6000

Abbreviations: for each parameter Rhat is the potential scale reduction factor (at convergence, Rhat = 1) and n.eff is a crude measure of the effective sample size. α_{ψ} is the intercept of occupancy models, α_p *is the intercept of* detection probability models and β is the slope of predictors in all models.

Appendix 2 Statistical outputs and data

Modelling of Bayesian single-season occupancy applied for assessing effect sizes of anthropogenic threats (i.e. livestock grazing, logging, poaching) on six large mammals species in Hyrcanian forest.

```
#--|-----------------------------------------------------------------------------------------------|-------#

source("fitJags_occ.R("

spec <- c("Leopard","Brown.bear","Red.deer","Roe.deer","Wild.boar","Wolf("

pritab <- c("dunif(-10,10("(

xp <- c(1)  # No Covariates for p

xpsi <- c("all")#,"logg","poach","live") # Covariates for occupancy

 vartab <- expand.grid(species=spec,prior=pritab,xp=xp,xpsi=xpsi,stringsAsFactors = F (

vartab$ini.a <- -5

vartab$ini.b <- 5

#vartab[vartab$prior==pritab[2],"ini.a"] <- 0

#vartab[vartab$prior==pritab[2],"ini.b"] <- 5

erg <- list()

for (i in 1:nrow(vartab}((

  set.seed(1234(

  cat(i,". Processing",vartab$species[i],"\n("

  tmp <- dat %>% select(FID,visit,species=matches(vartab$species[i]))  %>% arrange(FID(

  tmp <- tmp %>% spread(visit,species(

  y <- as.matrix(tmp[,2:4([
```

```
  rm(tmp(
  xp <- matrix(0,ncol=3,nrow=93,byrow=T(
  if (!is.na(vartab$xp[i} (([
    tmp <- dat %>% select(FID,visit,effort) %>% arrange(FID(
    tmp <- tmp %>% spread(visit,effort(
    xp <- as.matrix(tmp[,2:4([
    rm(tmp(
    xp <- apply(xp,2,scale(
    xp[is.na(xp)] <- 0
{
  xpsi <- rep(0,nrow(y((
  if (!is.na(vartab$xpsi[i]) & vartab$xpsi[i] != "all} ("
    tmp <- dat %>% select(FID,visit,threat=matches(vartab$xpsi[i])) %>% arrange(FID(
    tmp <- tmp %>% spread(visit,threat(
    xpsi <- as.matrix(tmp[,2:4([
    rm(tmp(
    xpsi <- apply(xpsi,1,max,na.rm=T)   ####### Manipulate threat-intensity here
    xpsi <- as.vector(scale(xpsi((
{
  if (vartab$xpsi[i] == "all}("
    xpsi <- matrix(rep(xpsi,3),ncol=3(
    tmp <- dat %>% select(FID,visit,live) %>% arrange(FID(
```

```
    tmp <- tmp %>% spread(visit,live(

    xpsi[,1] <- apply(tmp[,2:4],1,max,na.rm=T)   ####### Manipulate threat-intensity here

    tmp <- dat %>% select(FID,visit,logg) %>% arrange(FID(

    tmp <- tmp %>% spread(visit,logg(

    xpsi[,2] <- apply(tmp[,2:4],1,max,na.rm=T)   ####### Manipulate threat-intensity here

    tmp <- dat %>% select(FID,visit,poach) %>% arrange(FID(

    tmp <- tmp %>% spread(visit,poach(

    xpsi[,3] <- apply(tmp[,2:4],1,max,na.rm=T)   ####### Manipulate threat-intensity here

    xpsi <- apply(xpsi,2,scale(

{

   zst <- apply(y,1,max,na.rm=T(

  n <- 1

  if(vartab$xpsi[i] == "all") n <- 3

  if (substring(vartab$prior[i],1,5) == "dnorm}("

    inits <- function(){list(z=zst,

    alpha.occ=rnorm(1,vartab$ini.a[i],vartab$ini.b[i,([

     alpha.p=rnorm(1,vartab$ini.a[i],vartab$ini.b[i,([

     beta.occ=rnorm(n,vartab$ini.a[i],vartab$ini.b[i,([

     beta.p=rnorm(1,vartab$ini.a[i],vartab$ini.b[i{(([

 {  if (substring(vartab$prior[i],1,5) == "dunif}("

   inits <- function(){list(z=zst,
```

```
    alpha.occ=runif(1,vartab$ini.a[i],vartab$ini.b[i,([
  alpha.p=runif(1,vartab$ini.a[i],vartab$ini.b[i,([
   beta.occ=runif(n,vartab$ini.a[i],vartab$ini.b[i,([
   beta.p=runif(1,vartab$ini.a[i],vartab$ini.b[i{(([
{
  if(vartab$xpsi[i] != "all}("
    erg[[i]] <- try(fitJags(y = y, xp = xp ,
                           xpsi = xpsi ,
                           prior = vartab$prior[i,[
                           ini=list(inits(),inits(),inits(()
((
{  else}
    erg[[i]] <- try(fitJagsAll(y = y, xp = xp ,
                              xpsi = xpsi ,
                              prior = vartab$prior[i,[
                              ini=list(inits(),inits(),inits(()
((
{
{
lapply(erg,function(x) gelman.diag(as.mcmc(x(((
```

```
#--|-----------------------------------------------------|---#
```

Model formulation codes

```
fitJags <- function(y,
                    xp,
                    xpsi,
                    prior="dnorm(0,0.000001,"(
                    ini,
                    filename="model.txt}("
 ###  Write model text file
  sink(filename(
  cat")
    model}
 #      Priors
        alpha.occ ~ ", prior",
        alpha.p ~ ", prior",
        beta.occ ~ ", prior",
        beta.p ~ ", prior",
#       Likelihood
      for (i in 1:R} (
 #      true occup. state
        z[i] ~ dbern(psi[i([
```

```
        logit(psi[i]) <- alpha.occ + beta.occ * xpsi[i    [
      for (j in 1:T} (
     y[i,j] ~ dbern(eff.p[i,j([
    eff.p[i,j] <- z[i] * p[i,j[
    logit(p[i,j]) <- alpha.p + beta.p * xp[i,j[
     Presi[i,j] <- abs(y[i,j] - p[i,j([
   y.new[i,j] ~ dbern(eff.p[i,j([
   Presi.new[i,j] <- abs(y.new[i,j]-p[i,j([
{
{
   fit <- sum(Presi([,]
   fit.new <- sum(Presi.new([,]


   occ.fs <- sum(z([]
   {
 ,"     fill=TRUE(
sink()
data <- list(y=y, R = dim(y)[1], T =dim(y)[2], xp = xp, xpsi=xpsi (
params <- c("alpha.occ","alpha.p","beta.occ","beta.p","occ.fs","fit","fit.new("
nc <- 3
nb <- 20000
ni <- 100000
```

```
nt <- 40

#--|------------------------------------------------------------------------------------------------------------|-#

require(R2jags(

out <- jags(data = data, inits = ini, params, model.file=filename,n.chains=nc, n.iter=ni,
n.burn=nb,

  n.thin=nt(

{

fitJagsAll <- function(y,

 xp,

 xpsi,

 prior="dnorm(0,0.000001,"(

 ini,

 filename="model.txt}("

 ### Write model text file

  sink(filename(

 cat")

 model}

 # Priors

  alpha.occ ~ ", prior",

  alpha.p ~ ", prior",

  for (i in 1:3(

 }beta.occ[i] ~ ", prior",
```

```
{
 beta.p ~ ", prior",
#   Likelihood
   for (i in 1:R} (
 #  true occup. state
 z[i] ~ dbern(psi[i([
 logit(psi[i]) <- alpha.occ + beta.occ[1] * xpsi[i,1] +beta.occ[2] * xpsi[i,2] + beta.occ[3] *
xpsi[2,1 [
  for (j in 1:T} (
  y[i,j] ~ dbern(eff.p[i,j([
  eff.p[i,j] <- z[i] * p[i,j[
  logit(p[i,j]) <- alpha.p + beta.p * xp[i,j[
  Presi[i,j] <- abs(y[i,j] - p[i,j([
 y.new[i,j] ~ dbern(eff.p[i,j([
 Presi.new[i,j] <- abs(y.new[i,j]-p[i,j([
{
{
  fit <- sum(Presi([,]
  fit.new <- sum(Presi.new([,]
  occ.fs <- sum(z([]
  {
,"      fill=TRUE(
```

```
sink()

data <- list(y=y, R = dim(y)[1], T =dim(y)[2], xp = xp, xpsi=xpsi (

params <- c("alpha.occ","alpha.p","beta.occ","beta.p","occ.fs","fit","fit.new("

nc <- 3

nb <- 20000

ni <- 100000

nt <- 40

require(R2jags(

out <- jags(data = data, inits = ini, params, model.file=filename,n.chains=nc, n.iter=ni,
n.burn=nb
```

Table 1. Data used for the analysis in our modelling

poach	live	logg	Eff	Wolf	Rodeer	Boar	Rdeer	Leop	Bear	visit	N
1.00	3.00	1.50	2	0	0	0	0	0	0	1	1
0.17	0.83	1.00	6	1	0	1	0	0	1	1	2
0.57	0.43	0.43	7	0	0	1	0	0	0	2	3
0.11	1.00	0.00	9	1	0	1	0	1	1	3	4
0.33	1.00	0.67	3	1	0	1	0	0	1	1	5
1.00	3.33	3.67	3	0	0	1	0	0	0	2	6
0.17	0.00	0.00	6	0	0	1	0	0	1	3	7
0.00	0.33	0.50	6	0	0	1	0	0	0	1	8
0.00	1.67	1.67	3	0	0	1	0	0	0	2	9
0.40	1.40	1.20	5	1	0	1	0	0	0	3	10
0.00	0.67	0.56	9	0	0	1	0	0	0	1	11
0.00	0.33	0.67	3	0	0	1	0	0	0	2	12
0.00	2.50	1.00	2	0	0	0	0	1	0	3	13
0.00	1.20	1.20	10	0	0	1	0	0	1	1	14
0.67	2.33	2.00	3	0	0	1	0	0	0	2	15
0.00	2.67	0.00	6	0	0	1	0	0	0	3	16
0.50	2.50	1.25	4	0	0	1	0	0	0	1	17
1.33	1.33	2.00	3	0	0	0	0	0	0	2	18
0.17	1.00	0.00	6	0	0	1	0	0	0	3	19
0.00	0.67	0.00	3	1	0	1	1	1	1	3	20
0.00	0.67	0.00	3	1	0	1	1	1	1	3	21
0.00	1.50	0.00	4	1	1	1	1	1	1	3	22
0.50	1.25	0.00	4	1	0	1	0	1	1	3	23
0.25	2.75	0.00	4	0	0	1	0	0	0	3	24

0.67	3.33	0.00	3	0	0	1	0	0	0	3	25
0.00	2.00	0.14	7	0	0	1	0	0	1	2	26
0.60	3.20	2.60	5	1	0	1	0	0	1	2	27
0.00	0.38	0.38	8	1	1	1	0	0	1	2	28
0.00	1.25	2.00	4	0	0	1	0	0	0	2	29
0.25	2.00	1.00	4	0	1	1	0	0	0	2	30
0.20	0.60	0.00	5	0	1	1	1	0	1	2	31
0.25	0.00	0.25	4	0	1	0	1	1	0	2	32
0.17	1.83	1.67	6	0	0	1	1	1	1	2	33
0.00	0.00	0.00	4	0	1	0	1	1	0	2	34
0.07	0.07	0.00	14	0	1	1	1	1	1	3	35
0.67	0.00	1.33	3	0	0	1	1	1	0	2	36
0.25	0.50	0.00	4	1	0	1	1	1	0	3	37
0.00	0.00	0.08	13	0	1	1	1	1	1	2	38
0.00	0.00	0.00	8	1	1	1	1	1	1	3	39
0.00	0.75	0.38	8	0	0	1	0	0	0	1	40
0.00	1.67	0.67	3	0	0	1	0	0	0	2	41
0.00	3.00	1.33	3	0	0	1	0	0	0	3	42
0.20	0.00	0.40	5	0	1	1	1	1	0	2	43
0.00	0.22	0.22	9	0	1	1	1	1	1	3	44
0.00	1.25	2.00	4	0	1	1	0	0	1	1	45
0.38	0.75	0.38	8	0	0	1	0	1	0	2	46
0.14	1.14	0.00	7	0	0	1	0	0	0	3	47
0.00	1.00	0.50	4	0	1	1	0	0	0	1	48
0.33	2.67	1.00	3	0	0	1	0	1	1	2	49
0.20	0.70	0.50	10	1	1	1	1	1	1	3	50
0.20	0.40	0.80	5	0	1	1	1	1	1	2	51
0.17	0.50	0.83	6	1	0	1	1	1	0	3	52
0.00	0.60	0.80	5	0	1	1	1	1	0	1	53
0.14	0.00	0.29	7	0	0	1	0	1	1	2	54
0.00	0.33	0.83	6	0	1	1	1	1	1	3	55
0.50	1.75	1.50	4	0	1	1	1	1	1	2	56
0.33	1.67	0.00	6	1	0	0	0	1	1	3	57
0.00	1.86	0.29	7	0	0	1	0	0	0	1	58
0.67	3.00	3.00	3	0	0	1	0	0	0	2	59
0.00	1.50	1.25	4	0	1	0	1	1	1	3	60
0.00	1.73	0.55	11	0	0	1	0	0	0	1	61
0.33	4.33	1.00	3	0	0	1	0	0	0	2	62
0.00	2.00	0.80	5	0	0	0	0	0	0	3	63
0.00	1.33	0.33	3	0	0	1	0	0	0	1	64
0.33	1.33	0.67	3	0	0	1	0	0	0	2	65
0.25	2.25	0.75	4	0	0	1	0	0	0	3	66
0.50	2.75	1.50	4	0	0	1	0	1	0	2	67
0.80	1.80	2.40	5	0	0	1	0	0	0	3	68
0.00	0.40	0.00	5	1	0	1	1	1	1	3	69
0.00	0.57	0.00	7	1	0	1	0	1	1	3	70
0.50	3.50	1.00	4	1	0	1	0	0	0	3	71
0.25	2.50	0.50	4	1	0	1	0	1	0	2	72
0.25	3.50	2.50	4	1	0	1	0	1	1	3	73
0.00	1.00	0.00	3	1	0	0	1	1	1	3	74

0.25	0.75	0.00	4	1	0	1	1	1	0	3	75
0.75	5.50	4.25	4	0	1	1	0	1	1	2	76
0.00	2.60	0.40	5	0	0	1	0	1	1	3	77
0.00	1.00	0.00	4	0	1	1	1	0	0	3	78
0.25	1.00	0.75	4	0	0	1	0	0	0	2	79
0.67	2.33	2.67	3	0	0	1	0	0	0	3	80
0.25	1.50	1.00	4	0	0	1	1	1	1	2	81
0.20	0.40	0.20	5	1	1	1	1	1	1	3	82
0.00	0.00	0.00	8	0	0	1	1	1	1	1	83
0.07	0.40	0.47	15	0	1	1	1	0	1	2	84
0.00	0.00	0.00	8	0	1	1	1	1	1	3	85
0.00	0.33	0.17	6	0	0	1	1	1	1	1	86
0.14	0.00	0.00	7	0	1	1	1	1	0	2	87
0.00	0.00	0.00	8	0	0	1	1	1	0	3	88
0.33	2.67	1.00	3	0	1	1	1	1	1	1	89
0.20	0.00	0.00	5	0	1	1	1	1	1	2	90
0.00	0.67	0.00	9	0	0	1	1	0	0	3	91
0.20	1.20	1.80	5	0	1	1	1	1	1	1	92
0.13	0.25	0.00	8	1	1	1	1	1	1	2	93
0.50	2.83	0.33	6	1	1	1	1	1	1	3	94
0.00	0.50	0.67	6	1	0	1	1	1	1	1	95
0.33	0.17	0.17	12	1	1	1	1	1	0	2	96
0.00	0.31	0.19	16	0	1	1	1	1	1	3	97
0.00	1.75	1.25	4	1	0	1	1	0	1	1	98
0.10	1.10	0.50	10	1	1	1	1	1	1	2	99
0.07	0.79	0.00	14	0	1	1	1	0	1	3	100
0.00	1.50	0.00	4	0	0	1	0	1	0	3	101
0.00	0.75	0.00	8	1	0	1	1	0	0	3	102
0.11	0.68	0.37	19	1	0	1	1	0	1	3	103
0.00	0.30	0.20	10	0	0	1	1	1	1	1	104
0.00	0.17	0.33	6	1	1	1	1	1	1	2	105
0.00	0.80	0.00	5	0	0	1	1	1	0	3	106
0.00	0.20	0.00	5	0	0	1	1	1	1	1	107
0.00	0.00	0.00	5	0	0	1	1	1	1	2	108
0.00	0.00	0.00	4	0	0	0	1	1	0	3	109
0.00	0.20	0.00	5	0	0	1	1	1	1	1	110
0.00	0.00	0.00	6	0	0	1	1	1	1	2	111
0.00	0.67	0.00	3	1	0	1	1	0	0	3	112
0.00	1.50	1.50	2	0	1	0	0	1	1	2	113
0.00	0.20	0.20	5	1	1	0	1	1	1	3	114
0.00	0.80	0.40	5	1	0	1	1	1	0	2	115
0.00	0.25	0.00	4	1	0	1	1	1	0	3	116
0.00	0.83	0.67	6	0	0	1	1	1	0	2	117
0.00	0.83	0.00	6	0	0	1	0	1	0	3	118
0.00	0.50	1.00	2	0	0	0	0	1	1	2	119
0.00	1.00	0.00	5	0	0	0	1	0	0	3	120
0.25	0.25	0.25	4	1	0	1	1	0	0	2	121
0.33	1.67	0.67	3	1	0	1	0	0	1	3	122
0.00	0.25	0.50	4	0	0	1	0	0	0	2	123
0.00	0.33	0.00	6	0	1	1	1	1	1	3	124

0.00	1.00	0.50	2	0	0	1	0	1	0	2	125
0.00	0.33	0.33	3	0	1	1	0	1	1	3	126
0.00	0.75	0.50	4	0	1	1	0	1	0	2	127
0.20	0.60	1.60	5	0	1	1	0	1	0	3	128
0.00	0.67	0.67	3	0	1	1	0	0	0	2	129
0.00	0.00	0.50	4	0	1	1	1	1	1	3	130
0.00	0.25	0.00	4	0	1	1	1	1	0	3	131
0.00	3.00	0.50	2	0	0	1	0	1	1	1	132
0.33	0.67	0.67	3	0	0	1	0	1	1	2	133
0.11	0.44	0.33	9	0	1	1	0	1	1	3	134
0.20	0.90	0.50	10	1	0	1	0	1	1	1	135
0.29	0.14	0.43	7	0	0	1	0	0	1	2	136
0.00	0.56	0.44	9	0	0	1	0	1	1	1	137
0.00	1.40	0.60	5	0	0	1	0	0	0	2	138
0.00	1.20	0.00	5	0	0	1	0	0	0	3	139
0.75	1.50	1.00	4	0	0	1	0	0	0	1	140
0.20	0.20	0.40	5	0	0	1	0	0	1	2	141
0.40	0.40	0.00	5	0	0	1	0	0	1	3	142
0.00	2.00	2.25	4	1	0	1	0	0	1	1	143
0.00	0.00	0.25	4	0	0	1	1	0	1	2	144
0.00	0.50	0.83	6	0	0	1	1	1	1	3	145
0.10	1.70	0.60	10	0	0	1	0	0	0	1	146
0.40	0.20	0.40	5	0	0	1	0	0	1	2	147
0.20	1.60	0.00	5	0	0	0	0	0	0	3	148
0.00	0.55	0.45	11	0	0	1	0	0	0	1	149
0.09	0.45	0.36	11	0	1	1	1	1	1	2	150
0.14	0.29	0.14	7	1	1	1	1	1	1	3	151
0.00	0.75	0.17	12	0	1	1	0	0	0	1	152
0.38	0.75	1.13	8	1	1	1	1	1	1	2	153
0.00	0.20	0.60	10	0	0	1	0	0	0	3	154
0.57	1.29	0.43	7	0	0	1	0	0	1	1	155
0.20	0.20	0.40	5	1	0	0	0	0	1	2	156
0.00	0.63	0.00	8	0	0	0	0	0	0	3	157
0.00	0.67	0.33	9	0	1	1	1	1	1	1	158
0.00	0.67	0.67	6	0	1	1	1	1	1	2	159
0.00	0.43	0.00	7	0	0	1	0	0	0	3	160
0.00	1.25	0.25	4	0	0	1	0	0	0	1	161
0.00	0.36	0.36	11	0	1	1	1	1	0	2	162
0.00	0.33	0.00	9	0	0	1	0	0	0	3	163
0.00	0.75	0.75	4	0	0	1	0	0	1	1	164
0.33	1.00	1.17	6	0	0	1	0	0	0	2	165
0.11	0.78	0.00	9	0	0	1	0	0	1	3	166
0.00	0.00	0.00	8	0	0	1	1	0	1	1	167
0.00	0.63	0.00	8	1	0	1	0	1	1	2	168
0.00	0.00	0.00	6	0	0	1	1	1	0	3	169
0.00	0.00	0.00	8	0	0	1	1	0	1	1	170
0.00	0.13	0.13	8	0	0	1	1	1	1	2	171
0.00	0.00	0.00	8	0	1	1	1	0	1	3	172
0.00	0.50	0.33	6	0	0	1	0	1	1	1	173
0.00	0.00	0.00	3	0	0	1	0	0	1	2	174

0.00	0.00	0.00	5	0	0	1	0	0	1	3	175
1.00	2.00	1.00	4	0	0	1	0	0	0	1	176
0.50	1.00	1.50	8	0	0	1	0	0	0	2	177
0.00	1.60	0.00	5	0	0	0	0	0	1	3	178
0.00	0.00	0.00	4	0	0	1	1	1	1	1	179
0.00	0.00	0.00	4	0	1	1	0	1	1	2	180
0.00	0.00	0.00	12	0	1	1	1	1	1	3	181
0.00	0.00	0.00	3	0	1	1	1	0	0	1	182
0.20	0.20	0.00	5	0	1	1	1	1	1	2	183
0.00	0.00	0.00	7	0	1	1	1	1	1	3	184
0.00	1.67	1.00	3	0	0	1	1	1	1	1	185
0.00	0.50	0.00	4	0	0	1	0	0	1	2	186
0.33	0.00	0.00	9	1	1	1	1	1	1	3	187
0.00	0.67	0.33	3	0	1	1	0	1	1	1	188
0.17	1.50	1.50	6	0	1	1	1	1	1	2	189
0.17	0.00	0.00	6	0	0	1	1	0	1	3	190
0.00	0.00	0.00	7	0	1	1	1	0	1	1	191
0.00	0.00	0.00	4	0	1	1	1	1	1	2	192
0.00	0.00	0.00	6	0	1	1	1	1	0	3	193
0.00	0.00	0.00	6	0	1	1	1	0	0	1	194
0.00	0.00	0.00	3	0	1	1	1	1	0	2	195
0.00	0.00	0.00	4	0	1	1	1	1	1	3	196
0.00	0.00	0.00	4	0	1	1	1	1	1	1	197
0.00	0.00	0.00	5	0	0	1	1	1	1	2	198
0.00	0.00	0.00	11	0	1	1	1	0	1	3	199
0.00	0.00	0.00	6	0	0	1	1	1	1	1	200
0.00	0.00	0.00	7	0	1	1	1	1	1	2	201
0.00	0.00	0.00	15	0	0	1	1	1	1	3	202
0.13	0.00	0.00	8	1	1	1	1	1	1	2	203
0.00	0.00	0.00	5	0	1	1	1	1	0	3	204

Abbreviations: N – number of cells, Bear – brown bear, Leop – leopard, Rdeer – red deer, Rodeer – roe deer, Eff – effort, log – logging, live – livestock, poach – poaching

Chapter 3

Precision and reliability of indirect population assessments for the Caspian red deer (*Cervus elaphus maral*)

Chapter 3: Precision and reliability of indirect population assessments for the Caspian red deer (*Cervus elaphus maral*)

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A female Caspian red deer in Golestan National Park. *Photo Jalil HasanZadeh*

Abstract

The elusive Caspian red deer (*Cervus elaphus maral*) lives at low densities in rugged forest habitats of the Caucasus and the South Caspian region and its declining population requires urgent attention. We here address the precision and reliability of dung counts (Fecal Standing Crop approach FSC) and camera trapping (Random Encounter Model REM) for estimating its population size. We surveyed 36 km of strip transects arranged in systematic random design and applied 1585 camera trap nights of effort in the mountainous forest habitats of Golestan National Park, Iran. We also conducted a dung decay analysis of 80 samples. Dung decay rates were not habitat-specific and the mean time to decay was 141.8 ± 15.1 days, only ca. 52% of the most reliable estimate available for red deer dung. Estimated deer population size and density from dung counts was lower (194 ± 46 individuals, 0.46 ± 0.11 individuals/km^2, 2012-2013) than from REM (257 ± 84 individuals, 0.61 ± 0.20 individuals/km^2, 2011), but this difference was insignificant. Both these estimates confirm a sharp decline of the population from an estimated 2096 animals in the 1970's. Density estimates reached a stable level and were most precise at a sampling effort of 15 transects (FSC) and 1345 camera trap-days (REM). Our results confirm that FSC and REM are both reliable for assessing Cervidae populations.

3.1. Introduction

Knowledge of population size is crucial for the development of effective wildlife management strategies (Laing et al. 2003). In many cases, traditional total counts of large ungulates do not provide reliable and precise density estimates because of logistical constraints, unrealistic assumptions or poor theoretical background (Buckland et al. 2001). Several techniques of population estimation are well developed, but the selection of monitoring methods is often an intricate process depending on available logistics and time, species biology, budget, purposes and habitat physiognomies (Campbell et al. 2004, Waltert et al. 2008, Alves et al. 2013). In case of threatened cryptic species living in mosaics of open and closed vegetation such as deer (*Cervus* spp.), specific survey approaches are required (Buckland et al. 2001).

Direct counts such as distance sampling are among the most popular techniques for density estimation of large herbivores (Buckland et al. 2001). For deer, these counts are applicable in open and plain areas where animals are most visible (Smart et al. 2004). However, in closed habitats such as forests and dense scrublands direct counts are difficult to apply and in most cases the method fails to produce reliable results (Marques et al. 2001). Counts become even more problematic in small populations and/or naturally cryptic species, in which detection probabilities are low (Zero et al. 2013). Under these circumstances, indirect monitoring techniques such as dung counts and camera trapping can provide valuable alternatives (Burton et al. 2015).

In contrast to direct counts, indirect dung counts estimate an average abundance over several months and not only for the day of the survey, thus leading to higher accuracy (Marques et al. 2001, Tsaparis et al. 2009). As a disadvantage, they cannot account for the sex/age structure of the population (Buckland et al. 2001). Furthermore, dung counts on strip transects may underestimate density because of litter and grass cover affecting detection probability and encounter rate (Hemami and Dolman 2005).

Dung count techniques are well described and are among the most preferable survey methods for deer monitoring (Buckland et al. 2001, Tsaparis et al. 2009). Two approaches to dung counts have been developed: (1) Fecal Standing Crop (FSC) estimation of animal density as a function of the number of recorded dung samples, decay rate as probability of dung presence/absence and the defecation rate and (2) the Fecal Accumulation Rate (FAR) method, which is based on clearance counts and substituting decay rate by the period of time between two visits (Hemami and Dolman 2005). The latter has a substantial advantage over FSC by not requiring an estimate of decay rate. Thus, abundance can be estimated relatively quickly, without the need to monitor dung piles over a lengthy period of time (Alves et al. 2013). However, FAR also has several limitations, such as highly variable decay rates which demand for quite short periods between visits during which dung piles do not decay and can be sampled (Laing et al. 2003). FSC has recently been shown to be more precise than FAR (Alves et al. 2013) and it can be conducted using both line and strip transects The length of the decay experiment is chosen so that the number of dung samples persisting between the subsequent visits would be no less than six pellets in a dung sample (Laing et al. 2003). Like FAR, FSC depends on the knowledge of defecation rates, which can be estimated in controlled (semi)captive conditions (Hemami and Dolman 2005, Buckland et al. 2001). Overall, the most cost-efficient and effective method for estimating deer number via dung counts is FSC, particularly in small populations (Laing et al. 2003, Alves et al. 2013). Camera trapping is another important technique to estimate animal population size and density (Burton et al. 2015). The use of camera traps is especially encouraging in studies of activity patterns and behavior of elusive species in forests (Cusack et al. 2015b).

The respective analytical approach of capture-recapture relies on individual recognition of camera-trapped animals, but this is difficult to do in most ungulates which do not have natural markings (Foster and Harmsen 2012, Anile et al. 2014). To overcome this limitation, Rowcliffe et al. (2008) have suggested an alternative approach, the Random Encounter Model (hereafter, REM), which is based on the ideal gas model. This model, developed originally in physics to describe the collision between gas molecules, was adapted by ecologists to characterize encounters between animals and observers (Hutchinson and Waser 2007). Thereafter, Rowcliffe et al. (2008) modified that theory in REM which similarly describes contact rates between camera traps and animals. The REM technique has been already successfully applied to rare ungulate species (Rovero and Marshall 2009, Zero et al. 2013). As REM is still relatively new, its application is still uncommon and not tested in many regions, including the Middle East.

In this study, we estimate density of the Caspian red deer (*C. elaphus maral*) by analyzing data from FSC and REM and discuss the precision and reliability in relation to application of these two techniques for population estimation in the montane forests of Iran. The main threats to Caspian red deer are poaching, competition with livestock, and habitat loss. While livestock encroachment to natural habitats is certainly on the rise , poaching for meat and trophy antlers appear to be the most important cause of the rapid decline of this large herbivore (Kiabi et al. 2004).

3.2. Materials and Methods

Study area

We conducted this research in Golestan National Park (GNP) located in northeastern Iran (Fig. 1). GNP is the first Iranian protected area, which was designated as a national park in 1957 and became a UNESCO biosphere reserve in 1977 (Zehzad et al. 2002). The park is located in the mountainous terrain and represents a transitional zone between humid Caspian deciduous forest and dry steppe, with mean annual precipitation of 142 and 866 mm in the east and west, respectively. The GNP comprises a total area of 874 km^2 with an elevation range of 450 to 2411 m a.s.l. The forested part, where this study was conducted, covers 422 km^2 in the western part of GNP (Fig. 1, Akhani 2005).

GNP has been considered as one of the last refuges for large mammals in Iran (Ghoddousi et al. 2015 in press, 2016). The red deer shares its habitat with the Persian leopard (*Panthera pardus saxicolor*), urial (*Ovis vignei*), bezoar goat (*Capra aegagrus*), wild boar (*Sus scrofa*), roe deer (*Capreolus capreolus*), brown bear (*Ursus arctos*), gray wolf (*Canis lupus*), jungle cat (*Felis chaus*) and wild cat (*Felis silvestris*) (Kiabi et al. 2004).

In the past decades, the maral distribution in Caspian forest was significantly reduced and many populations were locally extirpated, mainly due to poaching; now, several surviving populations are restricted to protected areas (Kiabi et al. 2004). In GNP, red deer suffers from intensive poaching and ineffective law enforcement (Kiabi et al. 2004, Hamidi et al. 2014). The red deer is officially listed as an endangered species in Iran (Kiabi et al. 2004).

Dung decay surveys

One of the key components in the sampling design is to ensure adequate and spatially standardized sampling to cover the study area (Sutherland 2006). The deer range in GNP was stratified into closed forest (CF) (58% of the range), closed scrubland (CS) (24%) and open scrubland (OS) (18%) using ArcGIS 9.1 (ESRI Inc. USA) and Google Earth 7.1.5 (Fig. 1, Supplementary material Appendix 1 Table 1). Mean time to decay was estimated by monitoring the status of fresh dung samples (n = 80) recorded from deer resting places between December 2011 and November 2012. Each dung sample represented a group of pellets produced in a single act of defecation. Samples were identified as being fresh based on their size, moisture content, texture, shiny and wet surface, smell and lack of decomposition signs (Laing et al. 2003). Any dung piles, which could be confused with co-existing roe deer, were excluded. Thus, all dung samples used in this study were assumed to be fresh and 0-2 days old. For better visibility during subsequent visits, each recorded sample was marked by staining trees around it. Dung samples that had been covered by leaves, were scattered as a result of animal trampling, washed away by precipitation or removed by invertebrates were all considered as decayed (Laing et al. 2003, Tsaparis et al. 2009).

Twelve visits were undertaken across three habitats, roughly once every four weeks, in order to record fresh dung samples for decay experiment (Tsaparis et al. 2009). During each visit, at least two samples per habitat were recorded, given very low deer population density and detection probability of fresh deer dung.. Decay was recorded if ≤ 6 pellets were re-found in a dung sample during subsequent visits (Laing et al. 2003).

Defecation rate survey

The defecation survey was conducted in December 2015 in a 0.02 km^2 enclosure with 20 deer individuals. Overall, 1676 dung piles were counted over an 8 day period. The area was cleared of any existing dung and deer were fed with native plant species.

Dung count surveys

Originally, 26 strip transects were randomly selected using a 2x2 km grid (systematic random sampling) using in ArcGIS 9.1 (ESRI Inc. USA) and Hawth’s Tools (Beyer 2004), of which eight were inaccessible. Of the 18 remaining transects, eight were located randomly in CF, four in CS and six in OS. For logistical reasons, we used strip transect sampling which is accurate and comparable with the line transect method (Alves et al. 2013). The survey was conducted in January-February 2013. All transects were oriented in the south-north direction, as it complies with the general density gradient from the core of the park to the boundaries (Buckland et al. 2001, Fig. 1). Each strip transect of 2-km length and 2-m width was surveyed by two observers, one on either side of the strip as recommended by Buckland et al. (2001). These observers were sufficiently skilled to recognize red deer dung based on size and general appearance. Any dung piles of doubtful origin were discarded from the study. To investigate the optimum number of transects and dung samples, we used the standard error (SE) of dung samples as a measure of precision (Alves et al. 2013).

Camera trapping

We used the camera trapping database of the Persian leopard population assessment project in GNP, which was conducted during May-December 2011 (Hamidi et al. 2014). In the forested area, we mounted, 53 passive 35-mm film camera traps (Deercam TM, Park Falls WI, USA) at a height of ~ 40 cm for 42.83 (SE ± 0.02) consecutive camera days in each station along the trails and ridge tops of the park, whenever leopard signs were present (Hamidi et al. 2014, Ghoddousi et al. 2015, In press). The minimum distance between camera traps was 2 km (Ghoddousi et al. 2016). Since 16 camera traps were stolen during the study period (Hamidi et al. 2014, Fig. 1), we used data from 37 camera traps. Although camera traps were set up for leopard captures, we assume that they were placed randomly in relation to red deer movements (Rowcliffe et al. 2013, Ghoddousi et al. 2016). Camera traps were programmed to take pictures at 1 min delay, operate 24 h/day and stamp date and time on pictures.

Data analysis

Decay rate

To estimate dung decay, each i-th marked dung sample (i = 1, …, n = 80) was assigned Y_i= 1 if present during subsequent visits or 0 if absent (decayed). The period between the time of the dung defecation and the time of the visit was denoted as age (t, days) and the habitat was denoted as H. Binary logistic regression was used to analyze the effect of t and H on dung presence or absence during re-visits (Laing et al. 2003, Tsaparis et al. 2009, Amos et al. 2014).

We tested four candidate models (Table 1): model f_0 was the null model without covariates, model f_1 incorporated dung age (t), f_2 described the additive effect of the covariates habitat (H) and age (t) and f_3 addressed the interaction between H and t (Laing et al. 2003, Tsaparis et al. 2009). The Akaike Information Criterion corrected for small sample size (AIC_c), AIC_c weights (w_i) and delta Δ_i (difference between a given model's AIC_c and the best model's minimum AIC_c) were used for model selection. The best models were defined as those with $\Delta_i < 2$ (Burnham and Anderson 2002). The mean time to decay (T_{decay}) was computed from dung age t using equation 1 (Laing et al. 2003):

$$(1) \qquad T_{decay} = \int_0^\infty \frac{-\beta_1 t[1+\exp(-\beta_0)]\exp[-(\beta_0+\beta_1 t)]}{[1+\exp(-(\beta_0+\beta_1 t))]^2}\,dt$$

Where β_0 and β_1 (intercept and coefficient of age t) were obtained from logistic regression. (Laing et al. 2003). The SE and the 95% confidence interval (95% CI) of T_{decay} were estimated by means of bias-corrected and accelerated bootstrapping with 10 000 iterations (Efron and Tibshirani 1993).

The predictive power of the logistic model was tested with the Area under the curve (AUC) of Receiver Operating Characteristic (ROC). A score AUC = 0.5 means that the model has no discriminatory ability and AUC = 1 means that models are perfectly discriminated (Stephanie et al. 2001). Chi-square test was performed to test for difference of density estimates between habitats (Hemami & Dolman, 2005).

Defecation rate

It was not possible to estimate the SE because the defecation rates of individual deer were unknown. From these samples, we estimated defecation rate as 10.48 dung piles per individual and day using Eq. 2 from Buckland et al. (2001):

$$(2) \qquad P = \frac{N_{dung}}{N_{deer} \times N_{days}}$$

where N_{dung} is number of dung piles counted, N_{deer} is the number of deer in the enclosure and N_{days} is the number of days during which dung had been accumulated. The precision of N_{dung} and D_{FSC} was expressed as the coefficient of variation CV% = (SE/mean)*100 (Plumptre 2000).

Table 1. The logistic models of red deer dung decay rates in Golestan National Park, Iran. AICc = Akaike information criterion corrected for small sample size, DF = degree of freedom, H = habitat, t = dung age (days), w_i = AIC weigh$_t$, and Δ_i = delta of the model.

Model	Covariates	Residual deviance	*df*	AIC$_c$	Δ_i	w_i
f_1	*t*	59.46	78	63.62	0.00	0.77
f_2	*H+t*	58.17	76	66.70	3.09	0.16
f_3	*H*t*	55.40	74	68.56	4.94	0.07
f_0	Null	85.30	79	87.36	23.74	0.00

Density estimation: FSC

Deer density (individuals/km^2) in GNP was calculated as in equation 3 (Laing et al. 2003):

$$D_{FSC} = \frac{N_{dung}}{T_{decay} \times P} \times A \quad (3)$$

where D_{FSC} is the estimated deer density (individuals/km^2), N_{dung} is the estimated dung density (dung samples/km^2), T_{decay} is the estimated mean time to decay (days), *P* is the estimated defecation rate (dung samples/individual and day) and *A* is the study area (km^2). The precision of FSC was measured by the delta method of equation 4 proposed by Laing et al. (2003):

$$[CV_{total}(D_{FSC})]^2 \approx [cv(N_{dung})]^2 + [cv(T_{decay})]^2 + [(p)]^2 \quad (4\ )$$

where CV_{total} is the total coefficient of variation.

Density estimation: REM

REM was used to estimate deer density from photo-captures as in equation 5 (Rowcliffe et al. 2008):

$$D_{REM} = \frac{y}{t}\frac{\pi}{vr(2+\theta)} \times g \quad (5)$$

where D_{REM} is the animal group density (groups/km^2), y is the number of independent photo-captures per camera station, *t* is the sampling effort (camera days) per camera station, *v* is the animal daily distance walked (km/day), *r* is the camera trap detection distance (meters), ϑ is the camera trap angle and *g* is the average animal group size (individuals/group) (Rowcliffe et al. 2008). The mean group size was estimated as 2.78 ± SE 0.26 individuals/group from n = 57 group observations by GNP rangers. These observations came from daily patrolling records which covered the camera trapping period across the red deer habitats in 2012 (Table 2). The values of ϑ (0.175 radians) and *r* (0.012 km) were taken from Rowcliffe et al. (2008) who used the same model of camera traps as we did.

We extracted the range of daily distances walked *v* from the literature on red deer radio-telemetry: 3.85, 2.78 and 3.2 km/day in France (Pepin et al. 2004, 2008, 2009) and 3.62 km/day in Portugal (Carranza et al. 1991). From these estimates, we calculated the average daily distance walked as 3.36 ± SE 0.23 km/day. This average estimate was used in REM density calculations using equation 5, whereas the extreme estimates were taken for fixed the simulation and sensitivity analysis. The overall CV of the REM density was computed using the delta method (Rowcliffe et al. 2008). A threshold value of 10 photo-captures which is a minimum number of captures to achieve from expected trap rates was used to estimate the deer density as indicated by Rowcliffe et al. (2008). The 95% CI of REM density was calculated as the mean ± 1.96*SE (Zero et al. 2013).

Table 2. The parameters required for estimating red deer density from camera-trapping rates using the Random Encounter Model. CV = coefficient of variation, SE = standard error.

Parameters	Mean ± SE	CV%	Reference
Photo-captures/camera station (y)	0.27 ± 0.09	33.34	Hamidi et al. (2014)
Camera days/camera station (t)	42.83 ± 0.02	0.04	Hamidi et al. (2014)
Daily distance walked (v, km/day)	3.36 ± 0.23	6.84	Pepin et al. (2004, 2008, 2009), Carranza et al. (1991)
Detection distance (r, km)	0.012		Rowcliffe et al. (2008)
Detection angle (θ, radians)	0.175		Rowcliffe et al. (2008)
Group size (g, individuals/group)	2.78 ± 0.26	9.35	Golestan National Park, unpublished data (2011-2012)
Total		35.30	

Sensitivity analysis

To evaluate the sensitivity of our population estimates to potential violations of the underlying methodical assumptions, we conducted two types of sensitivity analyses. First, we simulated dung encounter rates and camera trapping data to account for non-random distribution of deer and non-random placement of camera traps based on Rowcliffe et al. (2008). Random dung and trapping data was simulated for each transect and camera trap, respectively, using a negative binominal distribution, with the mean defined by the expected number of samples (dungs/photos) per sampling unit (transect/camera) from collected samples in the field (total number of samples divided by number of sampling units) and variance defined by the observed variance of encounter rate (Rowcliffe et al. 2008). We then recalculated population estimates based on the methods described above (equation 3 and 5, respectively), resampling for 10 000 times each. We extracted the 95% CI as SE*1.96 of the population estimates overall resamples. Second, we used the extreme ranges of the model parameters to account for their variability and the usage of non-local daily distances walked estimates. For FSC, we used the lower and upper 95% CI of decay rate 114.57-173.65 and recalculated population size. For REM, we used lowest and highest observed mean group sizes per season (2.42, 3.58 individuals/group) and the most extreme daily movement estimates (2.78, 3.85 km/day) and recalculated population size based on all four combinations. For both methods, we reported the full range of the population estimates.

Finally, in order to evaluate the difference of original population estimates between FSC and REM, we performed a Z-test (Buckland et al. 2001). All statistical analyses were done in R statistical software v. 3.2.3 (2015).

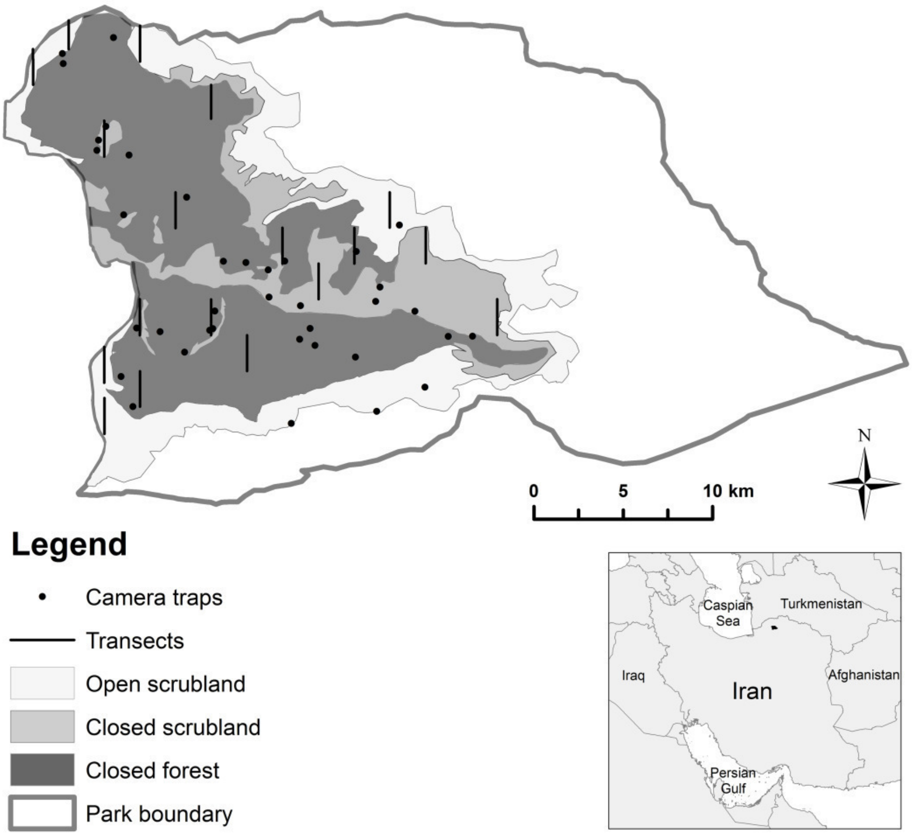

Figure 4.1. The study area, habitats and location of transects and camera trap stations in Golestan National Park, Iran.

3.3. Results

Using FSC, we estimated deer population size and density as 194 ± SE 46 individuals, overall %CV = 28.44 and 0.46 ± SE 0.11 individuals/km^2, respectively. With increasing the numbers of dung samples (N_{dung} = 50) and transects surveyed, precision increased (Figs. 2a and b), leveling off at approximately 15 transects (Fig. 2b).

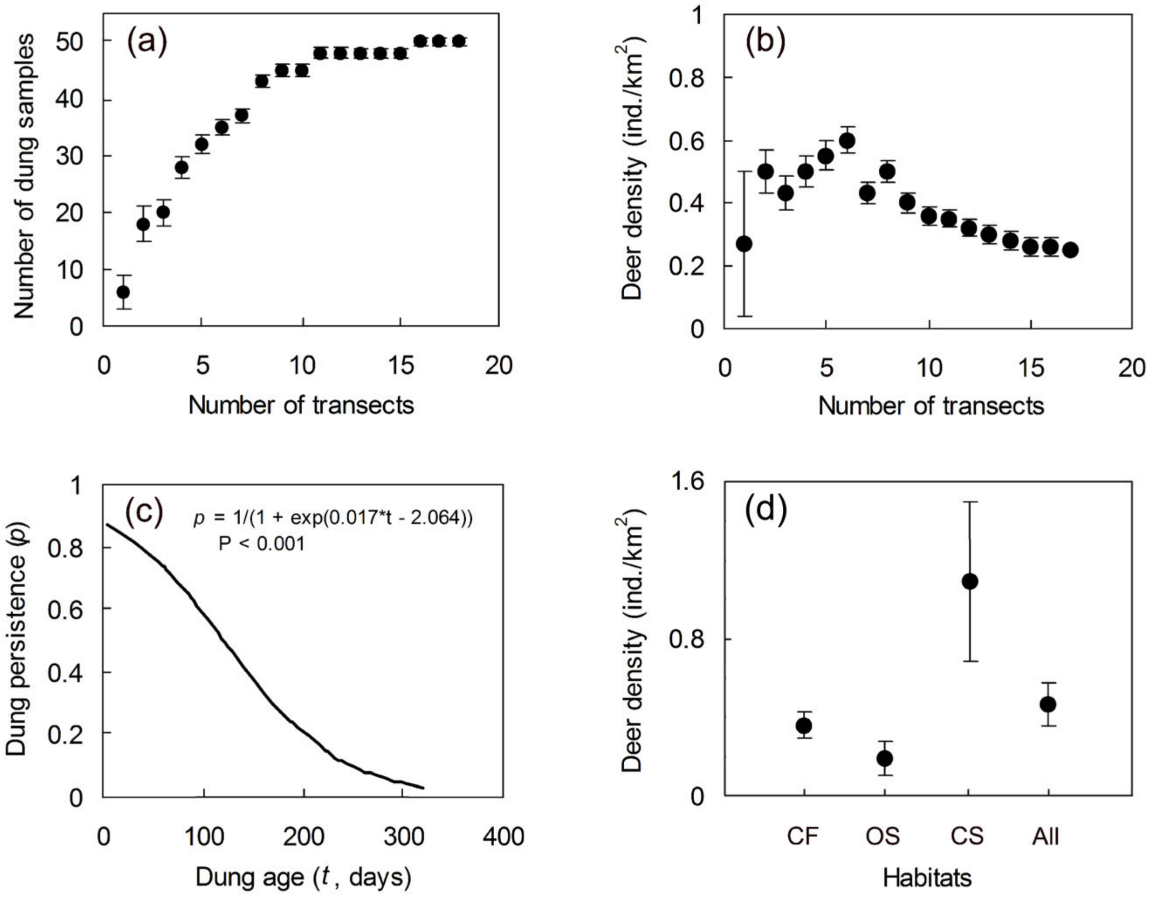

Figure 2. The number of dung samples (a) and red deer Cervus elaphus maral density estimates (b) in relation to the number of transects, the logistic regression curve of the probability of dung persistence over time (c) and the distribution of FSC deer density estimates across the closed forest CF, open scrubland OS and closed scrubland CS (d). The standard errors are displayed by bars.

We found fresh dung rather infrequently, at an interval of roughly every four weeks. Nevertheless, pooling dung observations from all three habitats provided a smooth dung decay curve (Fig. 2c). Density in closed scrubland was higher than in other habitat types (Fig. 2d, Supplementary material Appendix 2 Table 1) but using the Chi-square test, no significant difference was found between density estimates of different habitats (χ^2 = 0.875, P =0.831, df =3). For estimating the mean time to dung decay, we sampled 80 fresh dung samples, of which 28 were in CF, 26 in CS and 26 in OS. Of the logistic models of dung decay rates (Table 1), the age-based model f_1 (Fig. 2, Eq. 1) was the best with $\Delta_i < 2$. The habitat-based models received less support and the null model was not supported. The fitted logistic regression curve of the model f_1 (Fig. 2) had high predictive power (AUC= 0.87 ± SE 0.04, $P_{AUC} < 0.001$, % correct classification = 81.3%). The mean T_{decay} was estimated as 141.81 ± SE 15.07 days (95% CI = 114.57-173.65).

A total of 1585 camera days was accumulated over 37 camera trap stations, 10 photo-captures of red deer were achieved at 31 camera traps and 1345 camera days. Using camera trapping data and REM, we found the influence of effort allocation on the precision greatly depends on variation in encounter rate. Using REM, we estimated the population size and density at 257 ± SE 84 individuals with overall % CV = 35.30 and 0.61 ± SE 0.20 individuals/km^2, respectively. The deer numbers in Golestan estimated by the two methods appear to be disparate, but are not significantly different (Fig. 3, Table 3) and the confidence interval overlap. Both 95% CIs from the simulation analyses of FSC and REM revealed similar results to the CIs of the original estimates. The FSC simulation yielded a 95% CI of 135-257 individuals (compared to the original FSC 95% CI of 102-285 individuals). The REM simulation analysis estimated a 95% CI of 77-440 individuals (compared to the original REM 95% CI of 91-423 individuals). Also, the sensitivity analyses using the most values of the parameters resulted in similar population estimation estimates. They ranged from 237 to 486 individuals for REM (compared to the original estimate of 257 individuals) and from 163 to 248 individuals for FSC (compared to the original estimate of 194 individuals).

3.4. Discussion

Monitoring red deer is challenging throughout its global range (Marques et al. 2001). This research is the first attempt to fill a gap in empirical knowledge on red deer population size and density by applying two independent count techniques in a montane forest ecosystem. Although direct observation methods are methodologically well developed, they are difficult to apply where the species is rare, less habituated to human presence (Tsaparis et al. 2009), and in hardly accessible rugged landscapes (Singh and Milner-Gulland 2011).

Dung counts are often applied as an alternative method to count deer species (Alves et al. 2013, Amos et al. 2014). In this study, we successfully applied the FSC method to obtain accurate estimates of red deer population size and density in Golestan from dung counts (Table 3). In contrast to other studies (Tsaparis et al. 2009), we did not find variation in decay rates between seasons and habitats. However, the mean time to decay was 52% shorter than elsewhere (141.81 ± SE 15.07 vs. 275 ± SE 42 days; Laing et al. 2003). Laing et al. (2003) modelled dung decay by binary logistic functions and found a pattern of decay rate similar to our study. The high rate of dung decay in our study area could result from a high diversity of dung beetles, an intensive trampling rate and foraging behavior of co-existing large mammals, e.g. wild boar and brown bear, which potentially accelerate dung decomposition (Bahrami et al. 2011). Although we attempted to minimize variation of dung counts by obtaining local estimates of dung decay and defecation rates, still many transects contained no dung, which reduced the overall precision of FSC estimates of deer number and density. An accurate density estimate relies mainly on accurate estimation of dung encounter and decay rates (Alves et al. 2013). Further progress in counting deer in Golestan could be reached with the application of a stratified random survey design and species distribution modeling, for which independent historical data on population trends could serve as a reliable baseline (Kiabi et al. 2004, Hemami and Dolman 2007, Tsaparis et al. 2009, Alves et al. 2013).

Our results also suggests that estimates of deer numbers and density can be obtained using REM, a novel approach to estimate animal densities from camera-trap records without the need for their individual recognition (Rowcliffe et al. 2008). An allometric diagram of species densities and distance walked provided by Rowcliffe et al. (2008) shows that at least 1000 camera-days are required to obtain 10 photographs of rare ungulates. We reached this threshold at a similar camera trapping effort of 1345 camera days. As camera trapping rates are intuitively linked with animal abundance, encounter rates between individuals and camera traps are expected to increase with population density (Rowcliffe et al. 2008, Rovero and Marshall 2009). This relationship between camera trapping rates and population density is strong and can be linear, as in forest ungulates of Tanzania (Rovero and Marshall 2009). According to Rowcliffe et al. (2008), the number of camera trap stations and the amount of effort were adequate and could not affect deer capture rates. Therefore, low capture rates of red deer in Golestan are most likely caused by low densities of this ungulate.

Nevertheless, there are some limitations involved in the study design and application of methods, which may have influenced our results. Our camera trapping data comes from a study focused on leopards and this could have affected red deer capture rates. Although a predator-prey relationship between leopard and red deer may have affected the capture rates, it can be assumed that movement pattern of red deer is independent from leopard movement. Moreover, a recent study (Cusack et al. 2015a) reveals that herbivore capture rates are insensitive to camera trap placement scenarios. As our camera trapping was mainly conducted at the trails, we surmise that our red deer capture rate may be the result of an unbiased movement pattern of the species. Moreover, our simulation analyses revealed similar results to the original REM estimates.

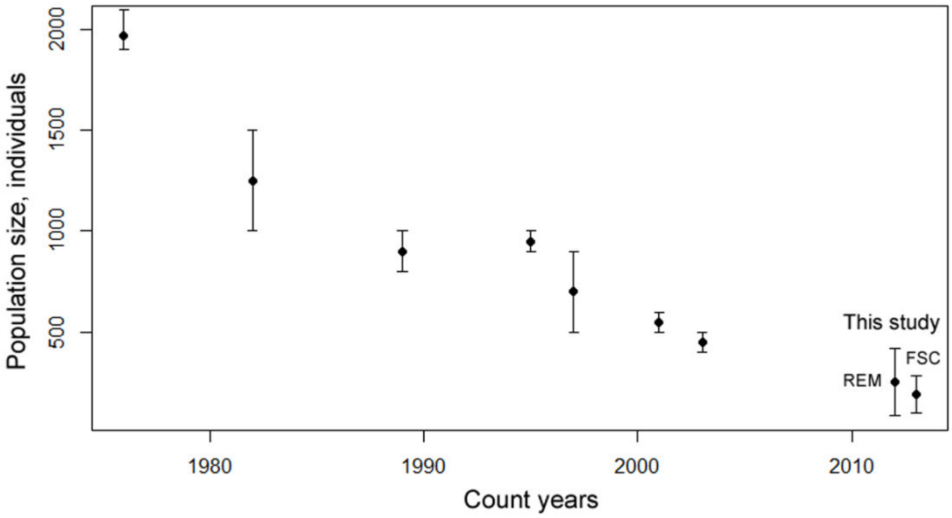

Figure 3. The estimates of red deer population size in Golestan National Park according to dung counts in 1976–1977 and total counts during the rutting season in 1978–2003 (Kiabi et al. 2004). Our REM (2011) and FSC (2012–2013) estimates are given for a comparison. The error bars show the limits of total counts (1976–2003) and 95% confidence intervals (present study).

Additional biases may arise from the application of non-local daily distance estimates, as well as seasonal variations of group size and decay rate, but sensitivity analysis showed that our results are relatively robust to substantial changes. Population estimation using REM resulted in wider a high overall coefficient of variance in comparison to FSC. The daily distances moved by red deer are unavailable for our study area and had to be borrowed from the Mediterranean region in France and Portugal (Carranza et al. 1991, Pepin et al. 2004, 2008, 2009). We strongly encourage researchers who apply REM to use local data, in particular for the most influential parameters, such as group size and animal movement for a realistic calculation of population size. Another source of uncertainty is that the defecation rate was, calculated from local captive deer this parameter may be differ between captive and free-living animals and also depending on seasons, forage intake, sex and age (Buckland et al. 2001). We attempted to minimize such potential bias by feeding animals with natural vegetation. Other studies also found that, the defecation rate shows less variation than other parameters used in the density estimation from dung counts (Neff 1968, Buckland et al. 2001, Marques et al. 2001). We also acknowledge here that our camera trapping and dung count data were collected from distinct years, which might affect results. However, we have no ground surmise that the population of red deer experienced any significance changes during this short period. Finally, our results could be affected by theft of 16 out of 53 camera traps by poachers, reducing sample size and habitat coverage (Rovero and Marshall 2009).

Both FSC and REM have clearly shown the scarcity of red deer in Golestan and through comparison with earlier counts suggest that the local deer population may have dropped by ~ 90 % from ca. 2096 (dung count, 1976-1977), 1897 (transects, 1976-1978), 900-1500 (rutting counts, 1982-1995) and 400-900 (rutting counts, 1995-2003) individuals to only 194-257 individuals now (Fig. 3, Table 3; Kiabi et al. 2004). Other, indicators of population decline are a now smaller group size (2.78 vs. 4.6 individuals/group in Kiabi et al. 2004) and heavy impact of poaching pressure on large mammals in Golestan in general (Ghoddousi et al. 2015, In press).

Most of red deer poaching occurs during the rutting season (September - October) when large stags are easily attracted by call imitation. In order to control poaching, the Iranian Department of Environment (DoE) implements constant patrolling throughout the red deer habitat in Golestan. Despite this, the scales of deer poaching are alarming. According to Kiabi et al. (2004), the main threat to red deer in Golestan is a combination of poaching and habitat degradation. This is indirectly confirmed by our study as deer were detected mainly in safe core zones rather than in ecotones along the forest verge. Thus, in line with Kiabi et al. (2004), we also emphasize the necessity to protect areas close reserve borders similar to those within the core zone of the park. Moreover, knowing better poacher's incentives may help in reversing the population decline of red deer in Golestan National Park (Kiabi et al. 2004).

Overall, we concluded that FSC and REM can serve as the practical techniques to count and monitor red deer populations living at low densities in montane forests. We strongly recommend to carefully consider opportunities and limitations of these methods and to use locally obtained variables of population size and density. . We further confirm the plight of the red deer population in Golestan and appeal for urgent, targeted and practical evidence-based conservation actions.. It is advised to conduct socio-economically oriented studies to unveil the potential reasons for poaching and decrease their incentives.

Acknowledgements

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3.5. Supplementary material

Table S4.1. Red deer (*Cervus elaphus maral*) habitats in Golestan National Park surveyed in this study (Akhani 2005).

Habitat	Description of species composition
Closed forest (CF)	Mainly *Quercus castaneifolia*, *Carpinus betulus*, *Acer velutinum*, *Cerasus avium*, *Sorbus torminalis*, *Fraxinus excelsior*, *Alnus glutinosa*, *Acer capadocium,* and *Ilex spinigera*. Understory is covered by grasses and herbs, e.g. *Danea racemosa*, *Euphorbia amygdaloides*, *Brachypodium sylvaticum*, *Parietaria officinalis*, *Carex sylvatica*, *Festuca gigantea* and *Poa nemoralis*.
Closed scrubland (CS)	Covered predominantly by invasive fern *Pteridium aquilinum* and also by *Crataegus pentagyna, Crateagus monogyna, Parrotia persica*, *Prunus divaricata*, *Paliurus spina-christi*, *Mespilus germanica*, *Thalictrum minus*, *Euonymus latifolia*, *Vicia variabilis*, *Lathyrus pratensis*, *Brachypodium sylvaticum, Dactylis glomerata*, *Rubus dolicocarpus* and *Rosa canina*.
Open scrubland (OS)	Assorted by scattered scrub layers comprising *Quercus macranthera*, *Carpinus orientalis*, *Acer monspsselanum*, *Malus orientalis*, *Crataegus microphylla*, *Pyrus boissieriana*, *Lonicera floribunda*, *Rubus sanctus*, *Prunus divaricata*, *Rahmnus pallasi*, *Cerasus microcarpa*, *Hypericum perforatum*, *Agrimonia eupatoria*, *Dactylis glomereta*, *Heteropapus altaicus*, *Botrichloa ischaemum* and *Rosa canina*.

Table 4.2. Deer density estimates and their confidence intervals in different habitats.

Habitat	Density, ind/km^2 ± SE	95% CI of density	Habitat size, km^2
Closed forest	0.35 ± 0.07	0.22 - 0.48	247.58
Open scrubland	0.19 ± 0.08	0.04 - 0.34	98.44
Closed scrubland	1.09 ± 0.41	0.29 -1.89	75.75
All deer range	0.46 ± 0.11	0.25 - 0.67	421.77

Chapter 4

The decline of ungulate populations in Iranian protected areas calls for urgent action against poaching

Chapter 4: The decline of ungulate populations in Iranian protected areas calls for urgent action against poaching

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Roe deer (*Capreolus capreolus*) killed by a poacher in a non-protected area. *Photo Unknown.*

Abstract

Poaching is cryptically but rapidly driving many species towards extinction. Knowledge of population trends of exploited species and incentives for poaching are necessary to inform appropriate conservation measures. We estimated the abundance of four ungulate species in Golestan National Park, Iran, the country's oldest protected area, where poaching of ungulates is widespread. We used line transect surveys (186 km), camera trapping (2777 camera-nights), point counts (64 scans) and dung counts (along 38 km), and compared population estimates with those from earlier records. We also investigated the incentives for poaching, using a semi-structured interview survey. Population estimates for 2011-2014 indicated a 66-89% decline in three ungulate species (bezoar goat *Capra aegagrus*, red deer *Cervus elaphus* and urial *Ovis vignei*) compared to 1970– 1978. Only wild boar *Sus scrofa* showed a population increase (of 58 %) during the same period, possibly facilitated by religious restrictions regarding the consumption of this species. The incentives for poaching were categorized (in a non-ordinal manner) as subsistence, pleasure, tradition, trade of wild meat, and conflict with conservation regulations and bodies. The decline in hunted ungulates in this Park appears to be the result of rampant poaching, and a similar trend is evident in other protected areas in the country. We suggest the adoption of participatory conservation strategies, improvement of law enforcement practices and cooperation with international experts to reduce poaching in these protected areas. Taking into account the incentives for poaching, a combination of economic and non-economic strategies should be considered.

5.1. Introduction

Poaching is a key phenomenon in the overexploitation of natural resources, driving many species towards extinction (Milner-Gulland et al., 2003). There are various incentives for poaching, which occurs at a variety of scales, from hunting by local communities for subsistence to selling wild meat in urban markets and international trafficking of wildlife or wildlife parts for income generation (Sutherland, 2000). The illegal and cryptic nature of poaching, and a lack of systematic wildlife monitoring schemes, make it difficult to detect population declines from poaching (Singh & Milner-Gulland, 2011; Nuno et al., 2013), and a lack of information on wildlife trends may hamper timely conservation responses (Milner-Gulland et al., 2003). Enforcement is probably the most widely practiced measure against poaching, and intensification has a positive influence on wildlife (Hilborn et al., 2006; Ghoddousi et al., 2016a).

However, enforcement requires social acceptability and proper sanctions to be effective (Milner-Gulland & Rowcliffe, 2007). Integrated conservation and development projects may therefore be influential in reducing poaching by targeting economic or non-economic incentives (Duffy et al., 2016), although they may fail if the wrong incentives are targeted (Duffy et al., 2016). Knowledge about incentives for poaching at the local level is required to guide the choice of appropriate conservation measures for reducing poaching pressure (Nuno et al., 2013; Challender & MacMillan, 2014). Hunting has a long history in livelihoods and culture in Iran (Firouz, 2005), but with the onset of the 20th century the availability of off-road vehicles and sophisticated firearms led to wildlife massacres across the country (Firouz, 2005). The Caspian tiger *Panthera tigris virgata* and Asiatic lion *Panthera leo persica* have gone extinct in Iran in the past century and the Persian fallow deer *Dama mesopotamica* had been considered to be extinct prior to its rediscovery in 1957 (Firouz, 2005). Wildlife populations declined significantly in most areas until the first modern hunting control was introduced in the 1950s (Moore, 1976; Firouz, 2005). Regulation of hunting in Iran began in 1956 with the establishment of the Game Council of Iran (renamed the Department of Environment in 1974) and the creation of the first network of protected areas. These efforts initiated recovery of wildlife in some areas (Moore, 1976; Firouz, 2005). Since then the Department of Environment has continued to increase the number of protected areas, and the area under its protection now comprises >10% of the country's land (Kolahi et al., 2012). However, political turbulence since 1979, lack of acceptance of conservation laws by local communities and the insufficient resources of the Department of Environment have resulted in widespread poaching in most protected areas (Tatin et al., 2003; Kiabi et al., 2004; Ghoddousi et al., 2016a). Ungulates are a major target of hunting in Iran (Firouz, 2005), with a diverse range of species, including the bezoar or wild goat *Capra aegagrus*, chinkara or jebeer gazelle *Gazella bennettii*, goitered gazelle *Gazella subgutturosa*, mouflon or wild sheep *Ovis orientalis*, urial *Ovis vignei*, *onager*

Equus hemionus onager, Persian fallow deer, red deer Cervus elaphus, roe deer *Capreolus capreolus* and wild boar *Sus scrofa*. Six of these species are threatened globally (IUCN, 2016; note synonymy of urial and mouflon on the Red List and the new classification of Rezaei et al., 2010).

Apart from the Persian fallow deer, which lives in semicaptive conditions, and the wild boar, the consumption of which is prohibited on religious grounds, all other ungulate species are threatened by poaching and have declined since the 1970s (Karami et al., 2002; Tatin et al., 2003; Kiabi et al., 2004; Shams Esfandabad et al., 2010; Ghoddousi et al., 2016a; Soofi et al., 2017). Despite widespread poaching, the scale of these declines in Iranian protected areas is unknown, as robust ungulate monitoring techniques are largely lacking. Furthermore, knowledge of the incentives for poaching remains limited. Subsistence, monetary profit, cultural values and conflict with the Department of Environment are the main incentives for poaching of ungulates in Bamu National Park (Ashayeri & Newing, 2012), but whether this is also the case in other protected areas in Iran is unclear. We measured ungulate population trends in Golestan National Park, for which ungulate abundance data are available from the 1970s (Decker & Kowalski, 1972; Kiabi, 1978; Kiabi et al., 2004). We assessed the populations of four species (bezoar goat, red deer, urial and wild boar) and compared them with their earlier status. There is no information on large-scale migrations, diseases or other environmental conditions, which might have affected the populations of these four species during this period. (Ghoddousi et al., 2016b). There is no competition with livestock in this area, as grazing is banned inside the Park and illegal grazing occurs only at a limited scale along the periphery (Ghoddousi et al., 2016b). Moreover, no major habitat destruction or development projects have reduced wildlife habitats in the Park in recent decades. However, poaching has been widely reported as being one of the main threats to ungulate species in the Park (Kiabi et al., 2004; Hamidi et al., 2014; Ghoddousi et al., 2016a; Soofi et al., 2017).

The severity of penalties, likelihood of capture by rangers, and incentives are the most important factors in poaching decisions (Milner-Gulland & Leader-Williams, 1992). Hunting is illegal in Golestan National Park and incurs fines or imprisonment. The density of rangers in the Park (c. 1 per 29.1 km2; authors, unpubl. data) is deemed sufficient to control illegal activities according to international recommendations (one ranger per 23.8 km^2; Jachmann & Billiouw, 1997). As understanding incentives may help managers find solutions to curb poaching (Milner-Gulland & Leader-Williams, 1992), we also evaluated the incentives for local poachers of ungulates in the context of existing disincentives.

Golestan National Park was established in 1957 in northeastern Iran (Fig. 1). It encompasses Hyrcanian montane forests, steppes and arid plains in an area of 874 km^2 (Akhani, 2005). From west to east, elevations span 450–2411 m, with mean annual precipitation of 866–142 mm, thus creating a variety of habitats (Akhani, 2005). The Park holds six species of ungulates, which is one of the highest numbers of ungulate species in Iranian protected areas (Ghoddousi et al., 2016b). Urial occur in steppes in the east and north of the Park, and roe deer and red deer inhabit forests in western and central parts. Wild boars are present throughout the Park (with the exception of a 25 km^2 arid plain), with higher densities in forests. A population of goitered gazelles occupies narrow plains in the east and north. Bezoar goats occur on cliffs across the Park. There are no villages within the Park but there are 15 villages, with c. 8660 inhabitants, within 2km of the Park borders. These communities rely mainly on crop and livestock farming.

5.2. Methods

Estimation of ungulate populations

We used a variety of methods to estimate population size, depending on the detection probabilities and habitat characteristics of the ungulate species (Ghoddousi et al., 2016b).

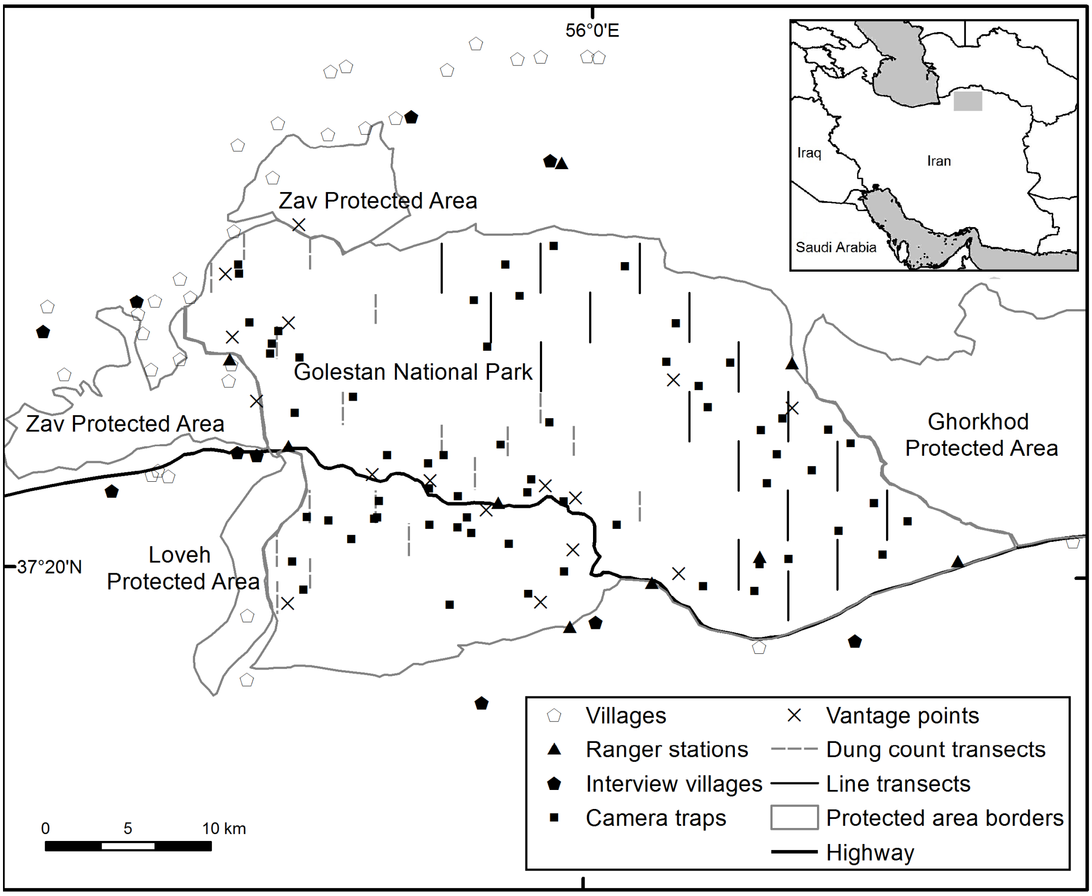

FIG. 1 Locations of line transects, dung count transects, camera traps, vantage points, ranger stations, and villages in and around Golestan National Park, Iran.

We excluded the goitered gazelle and roe deer from our study because the former has a limited distribution in the Park and there are insufficient data on the latter to facilitate estimation. Our estimates of abundance for 2011-2014 were compared to data for 1970–1978, which were based on comparable systematic monitoring methods (Decker & Kowalski, 1972; Kiabi, 1978; Kiabi et al., 2004). We are not aware of any other studies on populations of these ungulates in the Park during this time frame. When more than one estimate for a given species was available, or if a population range was given, we calculated the arithmetic mean.

Bezoar goat

We used a double-observer point count during November–December 2014 to estimate the abundance of bezoar goats (Suryawanshi et al., 2012). We identified 53.6 km^2 of rugged landscape as bezoar goat habitat and surveyed sampling points at random within this area, with a minimum distance of 3km between two points (Fig. 1). Two observers counted groups of goats, from vantage points 200–500 m away from the sampling points. The data were analyzed using DOBSERV (Nichols et al., 2000). The sampled area was calculated as the overlap of observable areas from vantage points and the identified goat habitat, using the view-shed function in ArcGIS 10.1. (ESRI, Redlands, USA). A detailed description of our sampling and modelling approaches is provided by Ghoddousi et al. (2016b). The earlier estimate of the bezoar goat population in the Park (4,000–4,500) was based on full-day observations in sample areas during (Kiabi, 1978).

Red deer

To estimate the red deer population size we used dung counts (faecal standing crop approach) and camera-trap data (randomized encounter model) in 422 km2 of forests and grasslands (Buckland et al., 2001; Rowcliffe et al., 2008). For the dung counts, we estimated red deer defecation rates by observing 20 individuals for 8 days in a 0.02 km2 enclosure with habitat comparable to that in the Park. Prior to the survey we estimated the dung decay rate by monitoring 80 fresh dung samples across red deer habitats in the Park and using binary logistic regression to assess the influence of time and habitat types on the survival of dung samples. We then surveyed 18 strip transects of 2 km length and 2 m width across red deer habitats during January–February 2013 (Fig. 1). For the randomized encounter modelling we used data from an earlier study, gathered from 37 camera traps during May–December 2011 (Fig. 1) (Hamidi et al., 2014). We calculated mean group size from observations of 57 red deer groups by park rangers during the period of camera trapping. Given the lack of red deer movement data from Golestan National Park or elsewhere in Iran, we used a mean of daily range estimates from other studies of 3.36 ± SE 0.23 km per day (Soofi et al., 2017). The camera-related parameters required by the randomized encounter model were obtained from a previous study that used a similar brand of camera trap (Deercam DC; Non Typical Inc., De Pere, USA; Rowcliffe et al., 2008). Details of the sampling and the application of both methods are provided by Soofi et al.

(2017). Red deer abundance during 1976–1978 was estimated by Hahn's census method via line transect surveys, and dung counts (Kiabi et al., 2004), as 1,897 and 2,096 individuals, respectively (Kiabi et al., 2004).

Urial

We used line transects to estimate the urial population size in 1897 km of steppes in the east and north of the Park (Buckland et al., 2001). We surveyed 17 3-km transects during January–February and August–September 2013, and February 2014 (Fig. 1), and analyzed the data using Distance 6.0 (Thomas et al., 2010). A detailed description of the methodology used to estimate the urial population is provided by Ghoddousi et al. (2016a). The urial population in 1970 was estimated by total counts in 12 sampling units and the extrapolation of recorded densities over the steppes of the Park (Decker & Kowalski, 1972). The estimated abundance was c. 15,000 individuals (Decker & Kowalski, 1972). In a separate, survey that used direct counts on line transects, there were estimated to be10,000–11,000 urial in the Park during 1976–1978 (Kiabi, 1978).

Wild boar

We estimated the abundance of wild boar using randomized encounter modelling based on camera trap surveys conducted during January–December (Rowcliffe et al., 2008; Hamidi et al., 2014). We used data from 67 camera traps installed throughout the Park, excluding a 25 km arid plain (Fig. 1) (Hamidi et al., 2014). The mean group size was estimated from observations on line transects during 2013–2014 (see above). Given the lack of information on the daily range of this species from the study site, we used an estimate of 6.8 ± SE 0.57 km per day from a study with similar habitat conditions (Podgórski et al., 2013). Details of our study design and analysis are provided by Hamidi et al. (2014) and Ghoddousi et al. (2016b). We used an earlier estimate of 2,500–3,000 wild boar from line transect surveys during 1976–1978 (Kiabi, 1978).

We assured interviewees that their data would remain anonymous and interviewees gave their verbal consent to participation in the survey. We held a focus group meeting with five poachers in December 2012 to elicit information about the incentives for poaching. We also used the findings of a social study on poachers in the Park (Ashayeri, 2014), conducting informal qualitative interviews with 15 poachers during June 2013–February 2014.

Interviews were continued until data reached the level of saturation, meaning that no further information could be extracted from new interviews (Newing, 2011). Details of the interview procedure and analytical approaches used are in Ashayeri (2014). The interviewees in both studies were all men, 29–66 years old, from 10 villages around the Park (Fig. 1). We also used seizure records for 2007–2014 to assess the frequency of poaching of various species in the Park. We calculated the poaching rate for each species as the number of hunted individuals as a proportion of their population size (2011–2014). As rangers use a sit-and-wait approach or tip-offs from local informants to detect poachers, we believe that seizure data are not biased towards a certain species, and represent the distribution of hunted species in the Park (authors, unpubl. data).

5.3. Sensitivity analysis

To assess the effects of uncertainty in population estimates we conducted sensitivity analysis using various combinations of 95% confidence intervals (2011–2014) and population ranges (1970–1978) for each species.

Interview survey and arrest records

Taking into account that poaching is a sensitive subject, we collected data on incentives for poaching by conducting semi-structured interviews with local people (Newing, 2011). We identified poachers in villages in the vicinity of the Park through a process of chain referral (Newing, 2011). Local poachers agreed to participate in the study after we collaborated with them in joint wildlife monitoring programmes (Hamidi et al., 2014; Ghoddousi et al., 2016a), built mutual trust, and explained the purpose of the study to them. Not being affiliated to any governmental organizations facilitated the process of data gathering and communicating with poachers. We asked each individual to give the main reasons for poaching ungulates in the Park (Ashayeri & Newing, 2012). We encoded and categorized incentives based on similar elements in responses (Ashayeri & Newing, 2012).

5.4. Results

Population estimation

Bezoar goat

During scans of 64 minutes each we observed 15 bezoar goats in seven groups. The model with equal detection probability between the observers estimated an abundance of 519 individuals (confidence interval CV = 31.3.%; 95% CI=201–807; Table 1). A comparison of recent estimates with the mean population size from 1976–1978 indicates an 88% decline of the bezoar goat population (Fig. 2). The results of sensitivity analysis show a decline of 79–96% during this period (Table 1).

Table 1 Abundance of four ungulate species in Golestan National Park, Iran (Fig. 1) during 1970–1978 and 2011–2014.

Species	1970–1978		2011–2014		Population change (sensitivity range)
	Abundance	Range	Abundance	95% CI	
Bezoar goat *Capra aegagrus*	4,250[1]	4000–4,500[2]	519	201–837[3]	–88% (–79 to –96%)
Red deer *Cervus elaphus*	1,997[1]	1897–2,096[4]	226[1]	91–423[5]	–89% (–78 to –96%)
Urial *Ovis vignei*	12,500[1]	10,000–15,000[2,6]	4,275	2,117–8,632[7]	–66% (–14 to –86%)
Wild boar *Sus scrofa*	2,750[1]	2,500–3,000[2]	6,478	3,050–9,906[3]	58% (2–75%)

[1] Arithmetic mean of two population estimates/ranges

[2] Kiabi (1978)

[3] Ghoddousi et al. (2016b)

[4] Kiabi et al. (2004)

[5] Soofi et al. (2017), from random encounter models

[6] Decker & Kowalski (1972)

[7] Ghoddousi et al. (2016a)

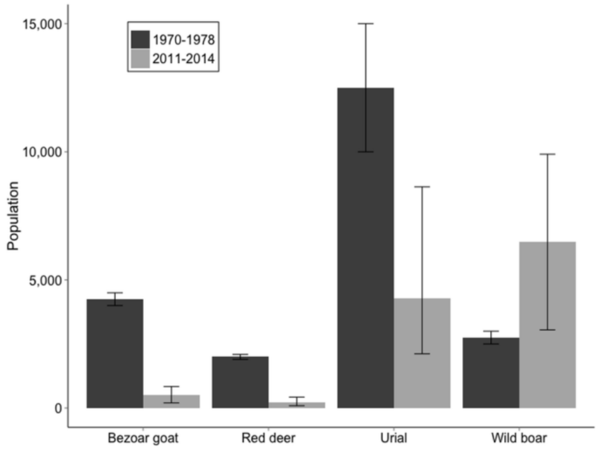

Fig. 2 Comparison of bezoar goat *Capra aegagrus*, red deer Cervus elaphus, urial Ovis vignei and wild boar *Sus scrofa* populations in Golestan National Park (Fig. 1) during 1970–1978 and 2011-2014. The error bars represent the 95% confidence intervals.

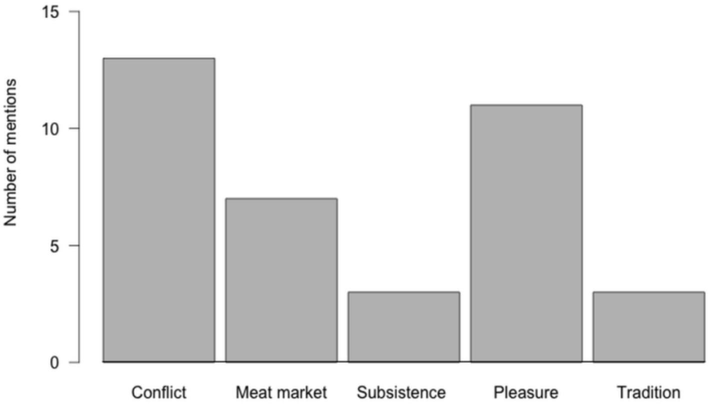

Fig. 3 The frequency with which various incentives for poaching were mentioned by interview respondents in the vicinity of Golestan National Park (Fig. 1).

The age-based model estimated a red deer dung decay rate of 142 ± SE 15 days. Fifty red deer dung samples were detected from a survey effort of 194 km. The faecal standing crop method estimated an abundance of 194 red deer (CV = 28.4%; 95% CI = 103–285; Table 1). We captured 10 photographs of red deer from 1,585 camera-nights of effort in forests and grasslands of the Park (Fig. 1). Using the randomized encounter modelling approach we estimated a red deer population of 257 individuals (CV = 35.3 %; 95% CI=91–423; Table 1). Comparison of the mean red deer populations in 2011–2013 and 1976–1978 indicates an 89% decline (Fig. 2). We used the wider 95% CI from the randomized encounter model for red deer sensitivity analysis, and the results indicated a decline rate of 78–96% compared to 1976–1978 (Table 1).

5.5. Discussion

We estimate there has been a 66–89% population decline of urial, red deer and bezoar goats in Golestan National Park since the 1970s. These were the preferred species of poachers (authors, unpubl. data) and accounted for the majority of hunted species. Such a trend in the absence of any reported migrations or mass mortalities may represent the effects of poaching in the Park. The fact that the population of wild boar, consumption of which is prohibited by Islam, has increased by 58% during the same time frame further supports our claim. Poachers avoid hunting this species in the Park (authors, unpubl. data).

The greatest declines were in bezoar goat (88%) and red deer (89%) populations. Bezoar goat habitat is restricted to patches of cliffs within the Park, where the goats are exposed and vulnerable to poaching. The Park is one of the last population strongholds of red deer in the Caspian forests (Kiabi et al., 2004) but despite dense vegetation and the elusive behaviour of red deer, this species is vulnerable to poaching, especially during the rutting season (authors, unpubl. data). In September and October each year, poachers imitate stag calls to attract deer to within shooting range. Without immediate conservation action both the bezoar goat and the red deer may go extinct in the Park in the near future. The urial population has also declined (66%), and the species is almost extirpated from some of its former range in the Park (Decker & Kowalski, 1972; Ghoddousi et al., 2016a). However, urial still occur in higher densities in the vicinity of the ranger stations (Ghoddousi et al., 2016a). It appears that the lack of regular systematic monitoring coupled with low detection probability of some species may have created an illusion of plenty among park managers, who may underestimate the decline in populations of hunted ungulates.

In Golestan National Park and other Iranian protected areas species living in open landscapes have been routinely monitored by annual total counts. Total counts do not follow a systematic sampling approach and the assumption of observation of all individuals in large areas can rarely be met (Buckland et al., 2001). Moreover, this method does not provide a measure of variance, which is necessary for assessing population trends over time (Suryawanshi et al., 2012). Therefore, it is necessary to adopt monitoring methods that are suitable for rugged landscapes and are sufficiently robust to detect trends in exploited populations at low densities (Singh & Milner-Gulland, 2011). We used a variety of monitoring methods successfully, and we recommend that the Department of Environment should initiate capacity-building programmes for rangers and invest in the equipment required to conduct similar surveys on a regular basis.

The differences in monitoring methods used in our study and those used to gather historical data are a potential source of bias. However, we are not aware of any other systematic surveys of these species since the establishment of the Park (Decker & Kowalski, 1972; Kiabi, 1978; Kiabi et al., 2004). Although our camera trapping was designed to target leopards Panthera pardus (Hamidi et al., 2014), we assume it did not produce a major bias, as movement patterns of herbivores are independent of those of carnivores (Cusack et al., 2015). Moreover, randomized encounter modelling of data for red deer and wild boar produced comparable results to other tested methods (Ghoddousi et al., 2016b; Soofi et al., 2017).

Our results demonstrate that conservation laws and enforcement measures have failed to stop poaching since the 1970s and therefore require reconsideration. We identified a combination of economic and non-economic incentives for poaching of ungulates in the Park, which may guide the selection of appropriate anti-poaching schemes. The variety of incentives to poach suggests that single policies are unlikely to succeed in deterring poaching, and that a combination of approaches is therefore required (Duffy et al., 2016). Poverty and the existence of a market for meat can be considered economic incentives, and creating alternative livelihoods for local communities is a common approach to tackle poaching stemming from such economic incentives (Duffy et al., 2016). However, subsistence poachers normally lack the skills, education and cultural capacities required for employment in many sectors (Nuno et al., 2013). Thus, integrated conservation and development projects could potentially explore livelihood opportunities in developing ecotourism or facilitating the establishment of community-based reserves, benefiting from poachers' local ecological knowledge. Awareness-raising campaigns against consumption of wild meat in urban areas could be considered, to target the demand side (Challender & MacMillan, 2014). In addition, the distribution and efficiency of law enforcement efforts in Golestan National Park should be improved (Ghoddousi et al., 2016a).

The existence of hunting incentives related to tradition and pleasure suggests that alternative livelihood programmes alone may fail to address the poaching problem (Waylen et al., 2009), but community outreach programmes aimed at building trust, awareness, motivation and opportunities have proven to be influential in controlling poaching in South-east Asia (Steinmetz et al., 2014). As the limited number of hunting permits issued annually by the Department of Environment is insufficient to satisfy demand, establishing community-based reserves could provide legal hunting opportunities for local communities. Integrated conservation and development programmes could investigate opportunities for creating such reserves.

As in a previous study in Iran (Ashayeri & Newing, 2012), conflict with conservation bodies and regulations was stated to be an incentive for poaching. The non-participatory and top-down approach to protected area management (Zendehdel et al., 2010), coupled with hostile encounters between rangers and local communities, causes conflict between the two parties. Additionally, hiring non-local rangers may overlook local ecological knowledge, leave local communities out of decision-making and cause conflict between local communities and conservation authorities. Nevertheless, we presume that conflict may exacerbate poaching but is not a root cause of it.

The future of hunted species in Golestan National Park and protected areas elsewhere in Iran is unclear. We recommend that the Department of Environment should adopt participatory conservation strategies, improve law enforcement practices and cooperate with international experts in resolving the poaching problem nationwide.

Acknowledgements

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Author contributions

AG, MS, AKH, SA, LE and MW conceived and designed the research. AG, MS, AKH, SA, LE and SG conducted the surveys. AG, MS, SA and LE analysed data. AG, MS, AKH, SA, LE, JS, IK, BHK and MW wrote the article.

Chapter 5

Assessing the relationship between poaching intensity, wild prey occurrence and livestock depredation rate by large carnivores

Chapter 5: Assessing the relationship between poaching intensity, wild prey occurrence and livestock depredation rate by large carnivores

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A female Persian leopard killed an adult cattle in Deilaman-Dorfak no-hunting area. *Photo* Department of Environment Gilan provincial office (camera trap).

Abstract

Poaching can reduce the prey base of carnivores, which in turn can increase human-carnivore conflict (HCC) through livestock depredation. However, a direct relationship between levels of poaching, wild prey abundance and livestock depredation has never been empirically proven. We surveyed 18 sites across the Hyrcanian forest in northern Iran, which is a global biodiversity hotspot under strong pressure of poaching, prey depletion and HCC. During three field visits, we walked 1,204 km in 93 4×4 km cells to count poaching signs and encounters with livestock and ungulate prey species of the Persian leopard (*Panthera pardus saxicolor*) and the grey wolf (*Canis lupus*). To document sheep/goat and cattle depredation we interviewed more than 201 herders. We analyzed the effects of forest cover, IUCN category of reserves, distance to villages, and wild prey and livestock encounter rates on carnivore depredation rates using generalized linear models. Prey poaching was found to be the most influential predictor for livestock depredation, as an increase of poaching occurrence by one sign/km significantly increased depredation up to three times, depending on the combination of livestock type and carnivore species. We also found significantly lower levels of poaching in national parks (cat. II) compared to other reserves and non-protected areas, though poaching signs were frequently found in majority of cells (58%). Red deer (*Cervus elaphus maral*) and roe deer (*Capreolus capreolus*) occurrence was significantly negatively associated with poaching and these species seem to be locally extinct in some of the surveyed sites. Our study shows the vital role of relationships between poaching, wild prey depletion and livestock depredation in carnivore conservation. Mitigating livestock depredation requires a combination of strict law enforcement of anti-poaching measures, upgrading the status of reserves, prey recovery, and application of participatory conservation and natural resource management.

5.1. Introduction

Most large mammalian carnivore populations are severely declining (Ripple et al., 2014; Wolf & Ripple, 2016), mainly due to conflicts with humans, prey depletion and habitat loss (Maxwell, Fuller, Brooks & Watson 2017; Benitez-Lopez et al., 2017). High metabolic demands, large home ranges and low wild prey availability in combination with livestock increasingly penetrating into suitable habitats lead to livestock depredation (Ripple et al., 2014; Kuiper et al., 2015; Chapron & López-Bao, 2016; Suryawanshi et al., 2017). Consequently, human-carnivore conflicts (HCC) arising from livestock depredation are widespread, causing socio-economic losses to local livelihoods and retaliatory killing of carnivores (Jędrzejewskia et al., 2017).

Poaching can lead to competition between humans and carnivores over limited prey resources, affecting large carnivore survival (Ghoddousi et al., 2017a) and is known to drive many species of carnivore prey towards extinction (Milner-Gulland, Bennett & SCB 2003; Brodie et al., 2015; Benitez-Lopez et al., 2017). Wild prey reduction may also increase livestock depredation rates by large carnivores (Khorozyan, Ghoddousi, Soofi, & Waltert 2015).

However, restoring high biomass of wild prey can as well have a positive effect as a negative one on livestock depredation by carnivores, which may differentially affect carnivore conservation (Khorozyan et al., 2015; Suryawanshi et al., 2017): It has been hypothesized contrastingly that (i) livestock depredation may arise in areas with high wild prey abundance and, consequently, may support higher densities of large predators (Stahl, Vandel, Herrenschmidt, & Migot 2001; Suryawanshi, Bhatnagar, Redpath, & Mishra 2013, Suryawanshi et al. 2017; Chetri, Odden, & Wegge 2017) or that (ii) livestock depredation increases in regions suffering from wild prey depletion where predators switch from wild to domestic prey (Treves et al., 2004; Khorozyan et al., 2015). Although there is evidence for both hypotheses, this important conservation issue remains understudied (Kuiper et al., 2015; Chetri et al., 2017; Suryawanshi et al., 2017). Generally, it has been found that depredation on livestock is more closely related to the loss of wild prey than to the density of predators or landscape protection status (Khorozyan et al., 2015; Wolf & Ripple, 2016). Ekernas et al. (2017) shows that pastoralists can have dichotomous effects on livestock depredation rates either by reducing predator density (i.e. killing) or by increasing predator density as livestock provide a plentiful prey resource capable of supporting high predator densities.

The effects of poaching on wildlife populations are not easy to study in the field (Millner-Gulland et al., 2003) and, to our knowledge, direct relationships between poaching, wildlife occurrence and the intensity of HCC have not yet been empirically described, especially at broad spatial scales. Existing studies dealing with HCC mainly focus on the identification and prediction of high conflict areas (Treves et al., 2004; Miller, 2015). However, to achieve effective and evidence-based conservation measures and to define management priorities, the relationship between poaching and HCC needs to be understood (Van Eeden et al., 2017).

The Hyrcanian forest in northern Iran provides a perfect example to study these relationships, as high levels of poaching and HCC are widespread there (Kiabi et al., 2004; Babrgir, Farhadinia, & Moqanaki 2017; Ghoddousi et al., 2017b; Khorozyan et al., 2015). For example, in Golestan National Park, an important biosphere reserve of the region, local ungulate populations have been depleted by 66-89% since the 1970s (Ghoddousi et al., 2017b). Previous local studies of HCC in the Hyrcanian forest suggest that high conflict intensity can be related mostly to prey depletion (Babrgir et al., 2017) or to the reduction of preferred prey (Ghoddousi et al., 2016).

In this study, we assess livestock depredation by the two carnivores most responsible for livestock depredation in the region, the endangered Persian leopard (Panthera pardus saxicolor) and the common grey wolf (*Canis lupus*). We hypothesized that there is a negative relationship between poaching rates and wild prey occurrence, positively influencing conflict rates between humans and large carnivores. Moreover, we expected that differences in predatory tactics of these carnivores lead to different patterns of livestock depredation (Garrott, Bruggerman, Becker, Kalinowskiki, & White 2007; Behdarvand et al., 2014; Imbert et al. 2016; Khorozyan et al. 2017), which may imply different management actions (Suryawanshi et al., 2013; Chetri et al., 2017).

5.2. Materials and methods

Study area

The Hyrcanian broadleaved forest expands from the Talysh Mountains in Azerbaijan through the northern slopes of the Alborz Mountains to Golestan National Park in eastern Iran with elevations ranging from -28 to 2800 m. The mean annual precipitation ranges from 530 to 1350 mm and occasionally peaks to 2200 mm in the western part. The mean air temperature of the hottest and coldest months varies from 28-35° C to 1.5-4°C, respectively (Sagheb-Talebi, Sajedi, & Pourhashemi 2014). The Hyrcanian forest is part of the Caucasus Biodiversity Hotspot due to the exceptional diversity of species and landscapes converged between Asia and Europe (Olson & Dinerstein, 2002). This forest harbors globally significant populations of the endangered Persian leopard and Caspian red deer (*Cervus elaphus maral*) and regionally important populations of the grey wolf, brown bear (*Ursus arctos*), roe deer (*Capreolus capreolus*), bezoar goat (*Capra aegagrus*) and wild boar (*Sus scrofa*) (Karami, Ghadirian, & Faizolahi 2017). The Iranian part of the Hyrcanian forest consists of 37 reserves: three national parks (NP), two wildlife refuges (WR), 19 protected areas (PA), 2 national natural monuments and 12 no-hunting areas (NHA; Darvishsefat, 2006). NHAs are established for a short period of time (~5 years) to allow the recovery of threatened populations in areas outside the formal reserves (Darvishsefat, 2006). Grazing of livestock in the Hyrcanian forest is banned inside the NPs and in the core zones of the PAs and WRs while ungulate hunting is strictly prohibited in all reserves (Makhdoum, 2008). Domestic animals in this landscape include cattle, sheep, goats, horses and dogs. Around 4 million cattle and small stock (sheep/goat) are herded or graze freely in small herds, with small stock being actively herded and kept in sheds at night (Ghoddousi et al., 2016). Traditionally, local herders legally move/migrate to highlands but tend to stay much longer than their livestock grazing permits specify (Sagheb-Talebi et al., 2014). Cattle are grazed predominantly in forests at lower elevations while sheep/goat are grazed mostly above the treeline.

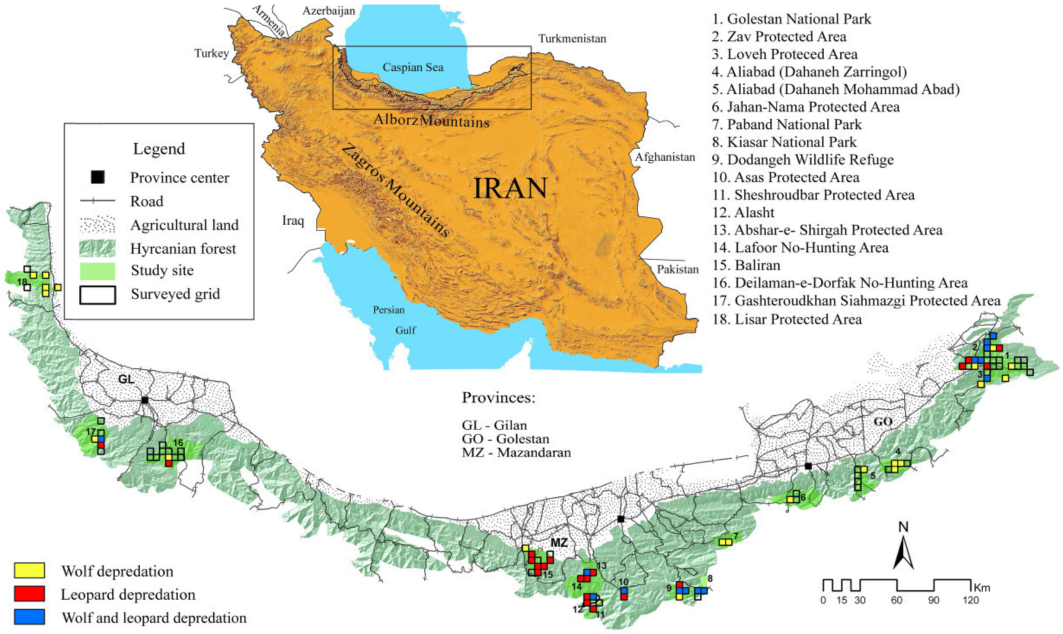

Fig. 1. Location of 18 study sites throughout the Hyrcanian forest, Iran. In each study site, the quadrates delineate the sampled grid cells (n = 93).

Study design

We selected 18 study sites covering 4112 km^2 and including 14 reserves (three NPs, eight PAs, one WR, two NHAs) and four non-protected areas (NPAs) across the Iranian Hyrcanian forest (Fig. 1). We placed a grid system of 4x4 km cells over all study sites. In each study site, we randomly selected ~45% of the total number of cells as sampling units using the Hawth's tools in ArcGIS v.10.2 (ESRI Co., USA; Beyer 2004). Cell size was based on an approximate average home range size of our targeted species, including wild prey and carnivores (Yackulic, Sanderson, & Uriatrtea 2011). During three discrete surveys (August-October 2015, February-April 2016 and August-October 2016), we visited each cell by a team of 2-3 people led by an experienced ranger or a local guide who were able to unambiguously identify signs of our targeted species and poaching. Overall, we surveyed 93 cells of which 45 cells were surveyed three times, 21 cells twice and 27 cells once. We walked 1204 km of trails during 147 field days.

Surveys of wild prey, poaching and livestock signs

In each cell, we walked along random trails of 2-13 km length and recorded the presence-absence of fresh signs (tracks, scratches, scrapes, feeding and resting places, rubbing posts and wallows) and direct observations (sightings and sounds) of the Caspian red deer, roe deer, bezoar goat and wild boar, which are the main prey species of leopard and wolf (Ghoddousi et al., 2016). Each survey team took photographs of signs for final identification/verification and to evaluate freshness. Each type of observed sign was assigned only once to each 200 m trail segment, allowing the standardized presence/absence records of the animal signs (Karanth et al., 2011). Concurrently, we recorded the occurrence of poaching and livestock (cattle, sheep/goat) signs. As poaching signs, we included the encounters with poachers, gun shells and gunshots heard (Laurance et al., 2008). As livestock signs, we documented the presence-absence of fresh signs (feces, tracks, corrals) and direct sightings (Karanth et al., 2011). We grouped data from sheep and goats as both these species graze together and are equally prone to predator attacks (Khorozyan et al., 2017). We rotated team members between grid cells to minimize the observer bias (MacKenzie et al. 2006).

Surveys of livestock depredation

We asked 201 herders encountered along random trails during the three subsequent survey visits across grid cells to report livestock depredation cases experienced in prior months using a standardized questionnaire (Table S2, Supporting Information). We assumed that local herders were skilled enough to distinguish between attacks of leopards and wolves based on depredation signs, e.g. spoor, injuries and direct sightings of the predator (Khorozyan et al., 2017). Predation signs are very distinct as leopards kill their prey mainly by strangulation from throat bites whereas wolves kill their prey mainly by laceration of flanks and hind legs. We discarded any ambiguous attack cases.

Field and GIS-based variables

We used the numbers of depredation events of cattle and sheep/goat by leopard and wolf in grid cells as two separate response variables. As field-based explanatory variables, we included the occurrences of poaching, wild (i.e. red deer, roe deer, wild boar, bezoar goat) and domestic prey (cattle, sheep/goat). These occurrences were measured as the proportions of random trail segments with signs present in relation to the total sampling effort (km of trails walked per cell and visit). We averaged the occurrence estimates of these variables over all visits if a cell was surveyed more than once. Additionally, we selected GIS-based variables, which are potentially relevant to livestock depredation (Table 1; Behdarvand et al., 2014; Miller, 2015). We calculated the mean distance to villages (DV) from the centroid of each cell, elevation as the mean elevation from a digital elevation model of 30-m resolution, forest cover as the proportion of forest cover (GoogleEarth) measured in each cell divided by total cell size and IUCN categories of reserves in study sites.

Analysis

We applied generalized linear models (GLMs) to analyze depredation events for each carnivore species (leopard, wolf) as a function of the variables described above. Before model construction, predictor variables were checked for multicollinearity using the Variance Inflation Factor (VIF) < 3 (Zuur, Leno, & Elphick 2010). Thus, we dropped elevation from all models for both carnivores. All remaining predictors were centered by dividing them by two standard deviations (Gelman, 2008; Grueber, Nakagawa, Laws, & Jamieson 2011). In the wolf data set, we identified one outlier from Cook’s distance < 1 (Zuur, Leno, & Elphick 2010), which we excluded from the analysis of the wolf data (on cell in study site Baliran, Fig. 1). From leopard data, we excluded six cells of Lisar PA (Fig. l) from the analysis due to local extinction of the leopard. Furthermore, to account for imperfect detections of livestock depredation events we used survey effort as an offset in the models (Kery & Royle 2016). To account for different observation intensities across cells potentially underestimating depredation events, we denoted effort *i* as an offset in each cell *i* (Kery & Royle 2016). Finally, we modelled depredation events for each carnivore as:

$$C_i \sim Poisson\left(Effort_i * \lambda_i\right)$$

For each grid cell i, an observed count of depredation events C was modelled as Poisson process, effort (km walked per cell i) is denoted as an offset, and λi is the expected count of depredation events per km in cell i.

$$\log\left(Effort_i * \lambda_i\right) = \beta_0.\log\left(Effort_i\right) + \alpha + \beta_1.x_{poaching_{,i}} + \beta_2.x_{forest_{,i}} + \beta_3.x_{reddeer_{,i}} + \beta_4.x_{roedeer,i} + \beta_5.x_{boar_{,i}} + \beta_6.x_{shoat_{,i}} + \beta_7.x_{IUCN_{,i}} + \beta_8.x_{DV_{,i}}$$

with α as the intercept and β n as the coefficients estimated for the offset β0 = 1. We measured the goodness-of-fit of the models with McFadden pseudo-R2 using the "pscl" R-package in R Statistic software (version 3.3.2, R Core Team 2015; Jackman et al. 2015). There were a number of competing models in each model set. To combine the strengths of the models, we applied model-averaging by multi-model inference using the R-package “MuMIn” (Barton 2009) for all candidate models with Δ QAIC < 2 (Grueber et al., 2011). We used quasi Akaike's Information Criterion (QAICc) corrected for small sample size to control for overdispersion (Grueber et al., 2011). We measured the relative variable importance (RI; Fig. 3) by QAICc-weighted standardized coefficients of the original model (Grueber et al., 2011). We calculated the odds ratio exp(β) to quantify the effect sizes of predictors on the response variable (Rauset et al., 2016) (Table S1, Supporting Information). We used Spearman’s rho correlation test to measure the relationship between poaching and wild prey occurrences. Additionally, we performed Mann-Whitney U test to compare poaching intensities among the study sites.

Sensitivity analysis

Since our data on depredation rates stemmed from herders, we wanted to assess any potential effects of data uncertainties on our results. Thus, to test for consistency of the original model results (Table S1), we conducted a sensitivity analysis for our models. Specifically, we simulated 99-fold replications of the response variables by adding or substracting a random number drawn from a Poisson distribution to the observed depredation events data (with λ = 0.2; Crawley, 2013). This simulates the potential inaccuracies of information provided by herders, which could underestimate or overestimate actual depredation rates by the two carnivores. If relative importance of a variable did not change much in these simulation models, we were confident that the variable had an effect on actual depredation rates. In the following, we term the models resulting from the sensitivity analysis as “simulation models”.

5.3. Results

We found that leopards mainly killed cattle (79%, n = 115) compared to sheep/goat (21%, n = 30) and that they only killed one individual per attack independent of the livestock species. By contrast, wolf mainly killed sheep and goat (96%, n = 501) compared to cattle (4%, n = 23). On average, wolves killed 8.64 ± 1.85 sheep and goat and 1.28 ± 0.19 cattle per attack in a total of 58 depredation events. We found that both carnivores had a narrow spatial overlap (14% of grid cells) in relation to livestock depredation. In regard to the protection status, we found that leopards killed 57% of sheep/goat and 37% of cattle outside reserves. Only 22% and 30% of attacks by wolves on sheep/goat and cattle, respectively, occurred outside reserves.

We found that poaching sign occurrence was significantly lower in NPs vs. WRs ($U = 213$, $p = 0.002$); NPs vs. PAs, ($U = 1017$, $p = 0.000$); NPs vs. NHAs ($U = 711$, $p = 0.000$) and NPs vs. NPAs ($U = 875$, $p = 0.009$). We did not find significant differences in poaching sign occurrence between other categories, except for NHAs vs. NPAs, where poaching was significantly lower in NHAs ($U = 631.5$, $p = 0.013$).

McFadden's pseudo-R^2 in our models ranged from 0.12 to 0.30. The highest pseudo-R^2 was found for the leopard-sheep/goat model ($R^2 = 0.30$), followed by the leopard-cattle model ($R^2 = 0.21$), the wolf-sheep/goat model ($R^2 = 0.19$) and the wolf-cattle model ($R^2 = 0.12$). For the leopard-cattle model, poaching significantly increased depredation rates by 3.01 times per cell ($\beta = 1.10$, 95% CI = 0.67 to 1.53, Fig. 2a). Further, its relative importance of 1.00 vs. 1.00 was retained both in the original and in the simulation model, respectively (Table S1; Fig. 3a). Distance to villages appeared to inversely influence cattle depredation rate by leopard ($\beta = -1.243$, 95% CI = -1.94 to -0.54; RI = 1.00 vs. 0.87). As the CIs of all other covariates overlapped with zero, there is little evidence that these parameters affect cattle depredation by leopards (Table S1; Fig. 2).

Poaching also proved to be the most important parameter for the leopard-sheep/goat model and positively increased depredation rates by 2.47 times (β = 0.91, 95% CI = 0.14 to 1.67, Fig. 2b; Table S1). In the simulation analysis, the RI of poaching decreased slightly by 10% (Fig. 3b). The abundance of red deer had a negative effect on sheep/goat depredation (β = -6.26, 95% CI = -10.95 to -1.57), but its RI of 1.00 was descended by 43% in the simulation model.

In the wolf-cattle model, poaching increased cattle predation by 2.88 times (β = 1.058, 95% CI = 0.22 to 1.90, Fig. 2c). However, its RI of 1.00 descended to 0.44 in the simulation model (Fig. 3c, Table S1). No other predictors were present among the top models. In the wolf-sheep/goat model, poaching increased predation by 2.57 times (β = 0.942, 95% CI = 0.50 to 1.39, Fig. 2d) while forest cover had a negative effect on predation rates (β = -1.33 to 95% CI = -2.07 to -0.59). RI values of these predictors of 1.00 remained without change in the simulation analysis (Fig. 3, Table S1). Moreover, wolf depredation on sheep and goat significantly increased with distance to villages (β = 0.351, 95% CI = 0.16 to 0.55). Finally, the occurrence of bezoar goats had a significant positive effect on sheep and goat depredation by wolf and its moderate RI = 0.45 vs. 0.53 was nearly consistent in the sheep/goat model while in the cattle model the RI = 0.16 vs. 0.08 were notably lower (Figs. 2-3; Table S1). The Spearman's correlation test showed that poaching sign occurrence was negatively associated with all ungulate prey species occurrences, but this relationship was only significant for red deer and roe deer (rho = -0.22, p = 0.027; rho = -0.25, p = -0.016) respectively.

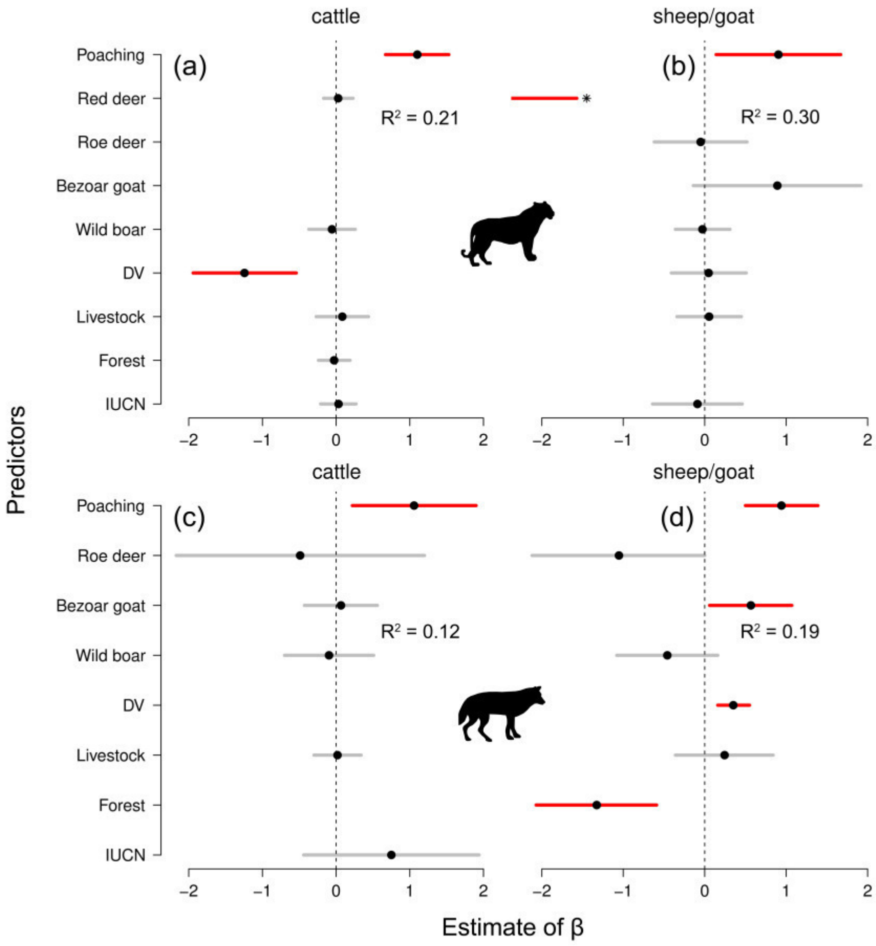

Fig. 2. Effect sizes (β, black dots) of GLMs after model averaging showing the effects of predictors on livestock depredation by the leopard (a – cattle, b – sheep/goat) and the grey wolf (c – cattle, d – sheep/goat) in the Hyrcanian forest, Iran. Bars around the estimates show a 95% confidence interval, with solid bars not overlapping zero. The asterisk (*) indicates that the mean and confidence interval exceed the scale of the figure. Abbreviations: DV - distance to village; forest - forest cover; IUCN – protection status of reserves defined by the International Union for Conservation of Nature.

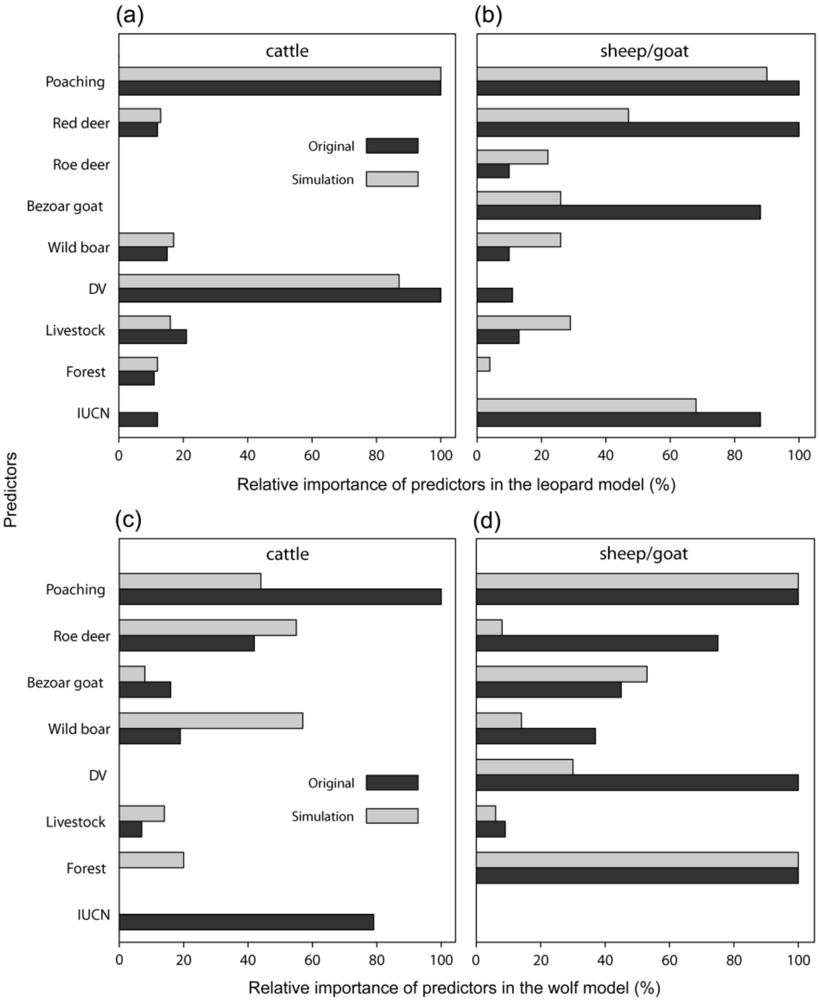

Fig. 3. Relative importance of predictors in original and simulation models for leopard (a cattle, b – sheep/goat) and wolf (c – cattle, d – sheep/goat) in the Hyrcanian forest.

5.4. Discussion

Despite poaching being a cryptic activity that is notoriously difficult to study (Milner-Gulland, Bennett, & SCB 2003; Laurance et al., 2008; Ghoddousi et al., 2017a,b), we were able to detect poaching signs throughout the reserves of the Hyrcanian forest. We show empirically, to our knowledge for the first time, that prey poaching can trigger livestock depredation by large carnivores and that depredation can be associated both with an increase or a decrease in prey occurrence.

Poaching was the most influential variable and its effect sizes were consistent across all our models. More specifically, an increase of poaching occurrence by one sign per kilometer increased the odds of cattle and sheep/goat depredation rates by leopard and wolf up to three times in each cell (range 2.47 to 3.01). In our study, decreasing red deer availability was significantly associated with increased depredation of sheep/goat by leopards. This may result from the fact that red deer are mostly scattered in highlands and living at low densities because of poaching pressure (Kiabi et al., 2004; Soofi et al., 2017a), whereas sheep/goat are comparatively easily available at these elevations (Ghoddousi et al., 2016; Imbert et al., 2016). This supports the hypothesis by Khorozyan et al. (2015) that large carnivores increase depredation on livestock, such as cattle and sheep/goat, when wild prey abundance is low.

On the other hand, we found wolf depredation on sheep/goat to be positively associated with the availability of bezoar goats. Domestic sheep and goats are highly abundant on plateaus above the treeline of the Hyrcanian forest and represent an easy prey compared to bezoar goats, which live in precipitous rocks alternating plateaus at higher elevations. This result is in line with Treves et al. (2004) who found wolves in the US to increasingly prey on livestock even when co-existing wild prey is abundant. These results are also in line with studies on snow leopards (Panthera uncia) which show depredation on livestock increase when both wild and domestic prey are available (Suryawanshi et al., 2013, 2017). Generally, these findings indicate that predators switch to easy prey where they are available (Stahl et al., 2001; Ghoddousi et al., 2016; Chetri et al., 2017; Ekernas et al., 2017).

In our study area, depredation of sheep/goat by both carnivores also increased with the distance to villages. This may be because sheep/goat are typically grazed in open scrublands at high altitudes (Abade, Dickman, & Macdonald 2014; Kuiper et al., 2015; Ghoddousi et al., 2016). Wolf depredation on sheep/goat was negatively associated with forests, but leopard depredation did not depend on landscapes. By contrast, leopard predation on cattle was higher in proximity to villages, perhaps because cattle are mostly grazed near villages inside the forest where wild prey is less abundant due to human activities (Tortato, Layme, Grawshaw, & Izzo 2015; Ghoddousi et al., 2016).

Overall, leopards were more likely to prey on cattle than on sheep/goat. Conversely, wolves were inclined to prey on sheep/goat and rarely killed cattle. This may arise from ecological differences in predation strategies of these carnivores (Ripple et al., 2014). Open areas are generally more suitable for chasing predators like wolves, what may explain their higher depredation on sheep/goat grazing in open pastures (Treves et al., 2004; Behdarvand et al. 2014; Imbert et al., 2016). Leopards preyed substantially more on cattle perhaps because forested areas are more favorable for stalking predator (Balme, Hunter, & Slotow 2007). Overall, the power of our models to assess livestock depredation by carnivores was varying, especially for the wolf-cattle model, suggesting that in certain conditions other predictors can be important (Treves et al., 2004). A recent study by Ekernas et al. (2017) found that the high-density population of wolves subsisting mainly on livestock may depress local wild prey resources by increasing predation. In our study, we did not obtain much evidence of a strong relationship between wild prey and livestock depredation, except for a negative association between red deer occurrence and sheep/goat depredation by leopard and a positive association between bezoar goat occurrence and sheep/goat depredation by wolf. So, the effects of wild prey on depredation can be both negative and positive, depending on local circumstances (Meriggi & Lovari, 1996; Stahl et al., 2001; Ekernas et al., 2017).

The Hyrcanian forest is a stronghold for the survival of the globally endangered Persian leopard in the Middle East (Kiabi, Dareshouri, Ghaemi, & Jahanshahi 2002; Farhadinia et al., 2015), so the results of our study are particularly relevant for the conservation of this species, which is the subject of research interest and public policy debate in the region (Breitenmoser et al., 2017; Farhadinia et al., 2015, 2017). Our results are also globally important for mitigation of human-carnivore conflicts as they show the large-scale effects of prey poaching and prey reduction on livestock depredation and carnivore status. Prey recovery and livestock protection plans should be set up especially in areas experiencing high defaunation and which are of high importance for landscape connectivity. For example, the disappearance of leopard in Lisar Protected Area threatens the survival of the population in the Caucasus where this big cat is critically endangered or locally extinct (Breitenmoser et al., 2017). However, prey recovery may increase depredation, therefore much attention should be paid to simultaneous application of livestock protection practices, like employing better herders and using well-trained guarding dogs (Khorozyan et al., 2017).

Overall, our study shows that recent intensive poaching depletes wild prey in the Hyrcanian forest, with livestock becoming hyperabundant in most localities, potentially out-competing wild herbivores, and leading to high depredation of livestock (Babrgir et al., 2017; Khorozyan et al., 2017). The Caspian red deer and roe deer became extinct from the western parts of the Hyrcanian forest; now they are patchily distributed in some reserves and continue to suffer from high levels of poaching (Kiabi et al., 2004; Soofi et al., 2017a; Ghoddousi et al., 2017b). The red deer numbers in Iran may now be only around one fifth of what was in 1977 (784 vs. 4350 individuals, Kiabi et al., 2004; Soofi et al., 2017b). The bezoar goat increasingly retreats into inaccessible rocky areas and is globally vulnerable (Weinberg et al., 2008; Ghoddousi et al., 2017b). Wild boar is the only abundant prey in the Hyrcanian forest as its hunting is rare due to Muslim religious beliefs (Goshtasb, Jahanshahi, Kavoosi, & Kiabi 2003; Ghoddousi et al., 2017a).

These results imply that enforcement measures have failed to curb poaching and grazing of livestock over the past decades (Kiabi et al., 2004; Ghoddousi et al., 2017a) and overgrazing is common even inside national parks (Sagheb-Talebi et al., 2014; Makhdoum, 2008). We suggest that, unless livestock grazing policy is strictly managed and ungulate prey resources are recovered, human-carnivore conflicts cannot be resolved. Essentially, livestock grazing should be banned not only inside national parks, but also in the core zones of lower-level reserves.

Mitigation of human-carnivore conflicts is unlikely to be successful without law enforcement. Traditionally, law enforcement as the principle conservation measure in Iran does not pay enough attention to local communities (Rauset et al., 2016, Ghoddousi et al., 2017a). In the future, conservation authorities should better consider local communities in conservation decision-making and address sustainable livelihoods near protected areas (Milner-Gulland et al., 2003; Rauset et al., 2016). In order to reduce poaching, motivations of poachers need to be understood and wildlife exploitation needs to be managed with support of local communities.

Although our study revealed that national parks (IUCN category II) have a significantly lower level of poaching than other reserves, they cover only ~0.01% of the total size of reserves in the Hyrcanian forest. In contrast, protected areas of category V represent 66% of reserves in the Hyrcanian forest. Priority actions are required in assessment of grazing capacities, allocation and enforcement of grazing quotas, and better coordination between governmental conservation and natural resource management organizations. Parts of protected areas and wildlife refuges retaining most natural habitats should be upgraded to category II.

Author's Contributions:

MS, AG, SS, LE, BK, IK and MW and NB conceived the ideas and designed methodology; MS, AJ, MS, MA, AQ, TG, AG, TZ collected the data; MS, TZ and NB analyzed the data; MW, NB and IK led the writing of manuscript. All authors contributed critically to the drafts and gave final approval for the publication.

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5.5. Supplementary material

Table S1. Model averaging results (original models) for the best candidate models from QAICc-based model selection for the Persian leopard and grey wolf depredation on cattle and sheep and goat in the Hyrcanian forest, Iran.

Relative importance		*P*	Exp (*β*)	Confidence interval	*β* ± unconditional SE	Parameter	Species
Simulation	Original						
-	-	0.207	1.45	(-0.19, 0.94)	0.370 ± 0.289	Intercept	Leopard (cattle depredation events)
1.00	1.00	0.000	3.01	(0.67, 1.53)	1.102 ± 0.218	Poaching	
0.13	0.12	0.780	1.03	(-0.17, 0.23)	0.028 ± 0.101	Red deer	
0.17	0.15	0.727	0.95	(-0.37, 0.26)	-0.056 ± 0.161	Wild boar	
0.87	1.00	0.000	0.29	(-1.94, -0.54)	-1.243 ± 0.357	DV	
0.16	0.21	0.644	1.09	(-0.27, 0.44)	0.084 ± 0.181	Cattle	
0.12	0.11	0.811	0.97	(-0.24, 0.19)	-0.027 ± 0.110	Forest	
-	0.12	0.785	1.03	(-0.21, 0.27)	0.033 ± 0.122	IUCN	
-	-	0.022	0.37	(-0.18, -1.16)	-0.990 ± 0.426	Intercept	Leopard (depredation sheep and goat)
0.90	1.00	0.022	2.47	(0.14, 1.67)	0.905 ± 0.388	Poaching	
0.47	1.00	0.001	0.00	(-10.95, -1.57)	-6.261 ± 2.391	Red deer	
0.26	0.88	0.092	2.44	(-0.14, 1.92)	0.892 ± 0.525	Bezoar goat	
0.04	-	-	-	-	-	Wild boar	
0.29	0.13	0.789	1.06	(-0.34, 0.45)	0.054 ± 0.200	Sheep and goat	
-	0.11	0.831	1.05	(-0.41, 0.51)	0.048 ± 0.239	DV	
0.22	0.10	0.864	0.95	(-0.62, 0.52)	-0.050 ± 0.289	Roe deer	
0.68	0.88	0.750	0.91	(-0.64, 0.46)	-0.089 ± 0.279	IUCN	
Relative importance		*P*	Exp (*β*)	Confidence interval	*β* ± unconditional SE	Parameter	
Simulation	Original						
		0.000	0.07	(-3.81, -1.27)	-2.543 ± 0.647	Intercept	Wolf (cattle depredation events)
0.44	1.00	0.015	2.88	(0.22, 1.90)	1.058 ± 0.429	Poaching	
-	0.79	0.222	2.11	(-0.44, 1.94)	0.750 ± 0.608	IUCN	
0.55	0.42	0.573	0.61	(-2.17, 1.20)	-0.488 ± 0.860	Roe deer	
0.57	0.19	0.756	0.91	(-0.70, 0.51)	-0.096 ± 0.307	Wild boar	
0.08	0.16	0.795	1.07	(-0.43, 0.56)	0.066 ± 0.251	Bezoar goat	
0.14	0.07	0.910	1.02	(-0.30, 0.34)	0.019 ± 0.163	Cattle	
0.20	-	-		-	- -	Red deer	
							Wolf (sheep and goat depredation events)
		0.288	0.68	(-1.084, 0.32)	-0.383 ± 0.358	Intercept	
1.00	1.00	0.000	2.57	(0.50, 1.39)	0.942 ± 0.227	Poaching	
0.30	1.00	0.001	1.42	(0.16, 0.55)	0.351 ± 0.100	DV	
1.00	1.00	0.001	0.27	(-2.07, -0.59)	-1.326 ± 0.378	Forest	
0.08	0.75	0.055	0.35	(-2.12, 0.00)	-1.053 ± 0.542	Roe deer	
0.53	0.45	0.031	1.76	(0.06, 1.07)	0.567 ± 0.259	Bezoar goat	
0.14	0.37	0.151	0.63	(-1.08, 0.16)	-0.459 ± 0.315	Wild boar	
0.06	0.09	0.433	1.28	(-0.36, 0.84)	0.244 ± 0.306	Sheep and goat	

Table S2. There are no conflicts of interests and all relevant parties involved in underlying research are either co-authors or listed in Acknowledgements.

Area name:	**Grid No:**	**Province:**	
Interviewer:	**Coordinates:** **X Y**	**Village name:**	
Shepherd/herder name:		**Survey date:**	
With which predator do you have conflict?	**Leopard:**	**Wolf:**	
CATTLE			
How many individuals attacked? Age: Sex:	**Kill site GPS:** **X Y**	**Date of attack:**	
When did attack happen?	**Locality name:**	**Landscape:**	
How do you recognize the predator from a kill?	**Strangulation:**	**Laceration:**	**Direct observation:**
SHEEP/GOAT			
How many individuals attacked? Age: Sex:	**Kill site GPS:** **X Y**	**Date of attack:**	
When did attack happen?	**Locality name:**	**Landscape:**	
How do you recognize the predator from a kill?	**Strangulation:**	**Laceration:**	**Direct observation:**

R codes used for statistical analysis

```
#-------------------------------------------#

## generalized linear modelling for depredation of livetsock by leopard and wolf

## EventSh represents the number of sheep/goat depRedDeerDeerated by Leopard [see LeopardWolf.csv]

#DistanceVillage: Distance to village

w <-
glm(Eventsh~BezoarGoat+Forest+RedDeer+RoeDeer+Boar+Poaching+SheepGoat+IUCN+DisanceVillage,

        data = leopard,

        offset=log(Effort),

        family = poisson)

### Multi Mudel Inferrence

library(MuMIn)

options(na.action = "na.fail")

m2    <- dredge (w, rank='QAICc', chat=deviance(w) / df.residual(w))

#------------------------------------------------------------------------------------#

# model averaging

myAvg <- model.avg(m2, subset = delta <= 2)

mav2  <- as.data.frame(coefTable(myAvg))

summary(myAvg)

# calculating McFadden's R2
```

```
#upload essential libraries
library(MASS)
library(lattice)
library(pscl)
#======================
# Sensitivitiy Analysis/Simulation
set.seed(123)
mostimportant <- list()
for (i in 1:99) {
  cat("Calculating", i, "of 99\n")
  # generate vector with random counts
  tab <- leopard
  rc <- rpois(nrow(tab),.2) - rpois(nrow(tab),.2)
  #Substract random vector from data
  tab$Eventsh <- tab$Eventsh - rc
  # Set negative values to zero
  tab[tab$Eventsh < 0,"Eventsh"] <- 0
    # Run model averaging routine ... change (in)dependent Variables
  m <-
glm(Eventsh~BezoarGoat+Forest+RedDeer+RoeDeer+Boar+Poaching+SheepGoat+IUCN+DistanceVillage,
 data = tab,
```

```
  offset=log(Effort),

  family = poisson )

  options(na.action = "na.fail")

  mumin2<- dredge (m, rank='QAICc', chat=deviance(m) / df.residual(m))

  myAvg <- model.avg(mumin2, subset = delta <= 2)

  imp <- rbind(data.frame(imp=c('(Intercept)'=NA)),data.frame(imp=importance(myAvg)))

  coTab <- coefTable(myAvg)

  mostimportant[[i]] <-   cbind(imp,coTab)

rm(imp,coTab)

}

varnam <- sapply(mostimportant,rownames)

varnam <- do.call("c",varnam)

result <- do.call("rbind",mostimportant)

result$rep <- rep(1:99,sapply(mostimportant,nrow))

result$var <- varnam

rownames(result) <- 1:nrow(result)

result <- result[,c(6,1:5)]
```

Table S1. Data used for the modelling

Poaching	Shoat	Effort	IUCN	Elevation	Forest	Rodeer	Rdeer	Boar	Bezoar	DV
0	1	0.92	0.55	1.43	0.53	0.45	0.29	0.92	0	0.48
0.19	0.41	1.91	0.55	0.95	0.64	0.2	0.2	0.61	0	0.51
0.31	0.54	1.98	0.55	0.77	0.79	1.24	0.49	1.78	0	1.08
0.28	0.49	1.05	1.37	1.23	0.08	0.3	0.07	1.69	0	2.28
0	0.13	0.92	1.37	1.22	0.56	0	0	1.29	0	1.64
0	0.58	1.58	1.37	1.06	0.68	0.03	0.05	1.75	0	0.83
0	0.48	1.45	1.37	0.97	0.9	0.17	0.61	1.6	0	0.9
0	0.26	1.45	1.37	0.56	0.72	0.14	0	1.68	0	0.91
0	0	1.32	1.37	0.72	0.6	0.31	0.48	1.11	1.59	0.5
0	0	1.32	1.37	1.64	0.11	0.53	0.87	1.04	0	2.7
0	0	1.84	1.37	1.67	0.15	0.55	0.17	1.25	0	1.45
0.16	0	1.28	1.37	1.65	0.01	1.1	0.96	0.38	1.24	2.25
0.28	0.85	0.99	1.37	1.28	0.11	0.24	0.48	1.2	0	0.61
0.17	0.08	0.99	1.37	1.27	0.64	1.05	0.91	1.06	0	1.02
0	0	1.12	1.37	0.97	0.75	1.26	0.68	1.32	2.59	1.82
0	0	0.86	1.37	1.38	0.45	2.31	0.63	1.74	0.58	1.87
0	0.05	1.58	1.37	0.89	0.94	0.8	0.44	1.68	0	1.16
1.24	0.9	1.12	0.55	0.57	0	0	0	0.29	0	0.29
0.37	0.68	1.25	0.55	0.67	0.72	0	0	0.21	0	0.77
0	0.97	1.25	0.55	0.56	0.56	0	0	0.26	0	0.48
0.58	0.87	1.32	0.55	0.8	0.6	0	0	0.15	0	0.34
1.12	0.57	0.92	0.55	0.76	0.64	0	0	1.07	0	0.37
0.37	1.05	0.92	0.55	0.55	0.6	0	0.14	0.94	0	0.57
0.6	0.33	1.68	0.55	0.6	0.15	0	0.87	0.55	0	0.42
0.64	0.52	1.32	0.55	0.48	0.38	0	0.58	0.37	0	0.42
0	0.5	1.38	0.55	1.87	0.02	2.78	0.32	0.83	1.67	0.6
0	0.09	0.92	0.55	1.92	0.01	1.46	0.84	0.11	2.5	0.51
0	0.24	0.92	0.55	1.31	0.72	4.33	2.5	0.3	1.39	1.22
0.55	0.99	0.92	0	0.38	0.75	0.17	0.14	1.26	0	0.51
0.43	0.35	1.25	0	0.3	0.9	0	0.14	1.25	0	0.97
0.44	0.67	1.12	0	0.22	0.6	0.07	0.27	0.54	0	0.49
0.18	0.08	5.5	0	936.45	1483.7	1.17	0.42	1.33	0	5601.1
0	0.6	0.92	0	0.19	0.23	0	0	0.65	0	0.3
0.25	0	1.38	0	0.93	0.92	0.53	1.33	1.52	0	0.92
0.48	0.56	0.66	0	0.1	0.15	0	0	0.68	0	0.05
0.12	0	1.12	0	0.46	0.9	0.23	0.52	1.03	0	1.38
0.28	0.97	1.25	0	0.18	0.75	0	0	0.38	0	0.14
1.04	0.69	0.92	0	0.33	4.44	0.13	0.22	0.1	0	1.1
0.28	0.33	2.24	0.68	1.89	0.3	1.44	0.12	0.82	0.29	0.67
0.68	1.25	1.25	0.68	1.16	0.3	2.98	1.12	1.65	0.91	0.39
0.06	0.05	2.04	0.68	1.66	0.64	1.31	0.19	0.63	0	0.79
0.12	0.07	1.38	0.68	1.4	0.49	1.16	0.35	0.54	0	0.7

0.44	0.8	1.12	0.68	0.94	0.23	0.2	0.36	0.53	0	0.28
0	2.35	0.79	1.37	1.74	0.02	0	0	0.77	0	0.37
0.14	0.8	1.84	1.37	1.47	0.04	1.92	0.16	1	0	0.29
0.55	1.28	1.25	0.55	0.36	0.75	0	0	0.7	0	0.29
0	0.28	0.92	0.55	0.46	0.75	0	0	0.46	0	0.48
1.24	0.52	0.79	0.55	1.12	0.64	0	0	0.64	0	0.37
1.66	0.69	0.86	0.55	-0.02	0	0	0	0.61	0	0.44
0.33	0.41	0.92	0.55	0.35	0.6	0	0	1.19	0	0.69
2.49	1.18	0.4	0.55	1.53	0.19	0	0	0	0	0.23
0.7	0.4	1.45	0.55	1.57	0.38	0	0	0.43	0.19	0.42
0.73	0.74	0.69	0.55	0.96	0.77	0.29	0.29	0.35	0	0.49
0	0.63	0.69	0	0.7	0.77	0.19	0	0	0	0.42
0	0.6	1.19	0	0.93	0.61	0.17	0	0.27	0	0.5
0	0.26	0.89	0	1.02	0.61	0.95	0	0.35	0	0.27
0	1	0.69	0	1.28	0.27	0.1	1.21	0.31	0	0.42
0	0.22	0.99	0	0.79	0.61	0.08	0.14	0.22	0	0.49
0.25	0.67	0.89	0	1.07	0.73	0	0.59	0.45	0	0.8
0	0.29	0.69	0	1.36	0.96	0.08	2.74	1.03	0	1.41
0	0.49	0.49	0	1.06	0.85	0	0.29	0.64	0	0.63
0	0.29	0.79	0	1.6	0.86	0.75	1.3	0.19	0	0.91
1.14	0.99	0.69	0.55	0.76	0.61	0	0	0.25	0	0.19
0.56	0.42	0.89	0.55	0.82	0.61	1.11	0.52	0.58	0	0.38
1.62	1.41	0.89	0	0.34	0.89	0	0	0.71	0	0.25
0.62	1.75	0.79	0	0.38	0.93	0	0	0.73	0	0.35
0	0.34	1.38	0	1.71	0.5	0	0	0.44	1.49	0.48
0.62	0.67	0.79	0	1.58	0.88	0.29	0	0.22	0	0.56
0	0.26	0.99	0	1.8	0.02	0.22	0	0.34	0	0.44
1.24	2.06	0.79	0	0.3	0.61	0	0	0.39	0	0.2
0.93	1.48	0.89	0.55	0.66	0.89	0	0.25	0.64	0	0.48
1.14	0	0.69	0	0.9	0.77	1.08	0	1.28	0	0.32
0	0	2.07	0	0.79	0.96	0.55	0.94	1.49	0	0.2
1.49	1.18	0.99	0	0.27	0.95	0	0	1.39	0	0.32
0	0	1.38	0	0.49	0.95	0	0	1.07	0	0.38
0	0	1.58	0	0.69	0.76	0	0.24	0.21	0	1.15
0	0	0.79	0	0.38	0.95	0	0	0.34	0	0.4
0.62	0.52	0.79	0	0.37	0.76	0	0.39	0.34	0	0.22
0.41	0.78	1.19	0	0.51	0.95	0.17	0	0.26	0	0.31
0.62	0	0.79	0	0.68	0.95	0.26	0.46	0	0	0.97
0.5	0	0.99	0	1.1	0.76	1.33	1.54	0.69	0	0.67
0	0	0.79	0	0.65	0.76	0.25	0.43	0	0	0.54
0.18	0	2.77	0	0.66	0.95	1.17	0.38	0.68	0	0.3
0.26	0.55	3.75	1.37	1.52	0.72	0.21	0	0.24	0	1.3
0	0.16	1.58	1.37	1.91	0.19	0.69	0	1.82	0	1.15
1.66	1.96	0.59	0.55	0.14	0.27	0	0	0.51	0	0.22
0.62	1.18	0.79	0.55	0.35	0.57	0	0	0.58	0	0.4

0	0.42	0.59	0.55	1.28	0.23	0.36	0	0.28	0	0.84
0	0.34	0.79	0.55	0.5	0.76	0.25	0.43	0.58	0	0.44
1.24	0.59	0.79	0.55	0.81	0.76	0	0	0.44	0	0.29
0	0	0.59	0.55	1.06	0.92	0.4	0	0.31	0	0.67
0	0.78	0.59	0.55	1.55	0.15	0.33	0	0	0	0.74
0	0.59	0.79	0.55	1.21	0.38	0.25	0.43	0.19	0	0.43

Abbreviations: DV - distance to village, Shoat - sheep and goat, Forest - proportion of forest cover, Rodeer - roe deer, Rdeer - red deer, Boar - wild boar

Chapter 6

Crop variety and prey richness affect spatial patterns of human-wildlife conflicts in Iran's Hyrcanian forests

Chapter 6: Crop variety and prey richness affect spatial patterns of human-wildlife conflicts in Iran's Hyrcanian forests

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A typical cropland in the Hyrcanian forest (Paband National Park). *Photo M.Soofi*

Abstract

Human-wildlife conflicts are a growing problem in Iran and pose a notable challenge to conservation efforts in its Hyrcanian forest region. We surveyed 162 households in 45 villages at four study areas to understand species-specific patterns of human-wildlife conflicts and people's reactions to these conflicts, and to suggest appropriate conflict mitigation measures. By using logistic regression, generalized linear and generalized linear mixed models we analysed socio-economic and ecological variables to find key determinants of the main conflict types around fifteen species of mammals, including large carnivores such as wolf, brown bear and Persian leopard. We also incorporated prey richness data (i.e. red deer, roe deer and wild boar) in our models. Wild boar and grey wolf were found to be the primary conflict species in regard to reported levels of severity; and crop loss by wild boars was reported by 97% of households. Logistic regression shows that these conflicts were positively influenced by the variety of cultivated crop species and the size of land under cultivation. Generalized linear mixed models show that human-wild boar conflicts increased in areas with lower human density, vegetation cover and distance to protected areas. Wolf conflicts were most frequently in the form of sheep attacks (81%) compared to goat (11%) and cattle (8%) attacks. These attacks were positively influenced by village and landscape elevation and increased in areas with lower prey richness and those located closer to, or inside, protected areas. Several cost effective mitigation measures should be used complementarily according to their effectiveness. These include avoiding planting of palatable seasonal crops near protected areas and establishing physical barriers around crop fields to lower large-scale crop damage by wild boars. To address livestock predation by wolf it will be essential to address the restoration of the wild prey community and efficiency of animal husbandry practices.

6.1. Introduction

Conservationists are challenged to resolve human-wildlife conflicts in a setting of rapid social and ecological changes across landscapes (Treves & Karanth 2003). Especially, borders of protected areas come into focus when investigating wildlife conflicts. It is here where human activities affect mortality of wildlife populations, making these areas shift from safe cradles of species survival into "population sinks" (Woodroffe & Ginsberg 1998). The competition for resources and space fosters depredation of crops and livestock by wildlife, often resulting in negative consequences for wildlife and people (Woodroffe et al. 2005; Karanth et al. 2013). The relative impact on rural peoples' livelihoods can be highly variable according to different husbandry and cultivation practices, people's economic dependence on rural activities, and conflict species' behaviour (Messmer 2000; Baker et al. 2008).

Little is known about the complexity of social, economic and ecological factors, which drive these conflicts (Dickmann 2010). Globally, experts highlight the difficulty to properly address and resolve conflicts between people and wildlife (Sillero-Zubiri et al. 2007; Dickmann 2010). Long-term strategies to reduce damage to agriculture are often lacking and research is limited to the question of how to manage these conflicts on the local scale. Facing these issues, it is claimed that the success of any conflict mitigation strategy will depend on an ability to reduce wildlife impacts on human lives and livelihoods to a level that people will accept, without reducing wildlife populations to unviable levels (Woodroffe & Frank 2005).

Here, we address human-wildlife conflicts in the Hyrcanian forests, northern Iran, where large mammals disappear at especially high rates (Dehagi et al. 2013; Ghoddousi et al. 2017a, 2017b). Today, it is the most threatened area of the country (Akhani et al. 2010). Outside protected areas, large areas of forest have been converted into croplands and isolation of protected areas from other natural areas is accelerating alongside with impoverishment of natural prey for large predators (Darvishsefat 2006; Nezami et al. 2014). Poaching remains a common practice, often applied inside protected areas (Gopalaswamy et al. 2012; Ghoddousi et al. 2017b). The close vicinity of villages to or inside forests and traditional agricultural practices intensify the pressure on the forest's integrity (Sagheb-

Talebi et al. 2014). In the face of these threats, the protection of large mammal populations is particular challenging. Whilst people in rural areas of Iran commonly experience human-wildlife conflicts, the scientific attention increases (Hosseini-Zavarei et al. 2013; Behdarvand et al. 2014; Khorozyan et al. 2017). However, more detailed descriptions on conflict patterns and a detailed understanding of their root causes are needed, especially in regions that are part of the Caucasus Biodiversity Hotspot (Mittermeier et al. 2011).

To properly resolve human-wildlife conflicts on the ground it is crucial to efficiently allocate and strengthen local resources in terms of knowledge. Mitigation measures require stakeholder commitment, an understanding of species-specific biological traits and behaviour, and should be used complementarily as they vary in their effectiveness (Distefano 2002; Sillero-Zubiri et al. 2007). The objectives of this paper are to (1) generate in-depth information on conflict species that were perceived to be of most severe concern for farmers in villages across different study areas in the Hyrcanian forests, (2) to assess socio-economic and ecological variables associated with these conflicts and (3) propose suitable measures to mitigate these conflicts. Due to the increasing population and destructive feeding behaviour of wild boar (*Sus scrofa*) in the Hyrcanian forests (Barrios-Garcia & Ballari 2012; Pandey et al. 2016; Ghoddousi et al. 2017a) we expected that this animal is a primary conflict species inflicting serious damages to many of the farming households. Given the rapid decline of wild ungulate species in Iran (Shams Esfandabad et al. 2010; Ghouddousi et al. 2017b; Soofi et al. 2017a) and higher chances of successful attacks when livestock is abundant (Patterson et al. 2004), we also expected that carnivore attacks on livestock are affected negatively by prey richness and positively by stock numbers.

6.2. Methods

Study area

The Hyrcanian forests are primary temperate broadleaved forest, fringing the southern coastline of the Caspian Sea from the Talysh Mountains in Azerbaijan eastwards over the northern slopes of the Arborz Ridge to Golestan National Park in Iran. The forests are approximately 800 km long and 110 km wide, covering about 1.85 mio ha (Sagheb-Talebi et al. 2014). As a relic of the Arcto-Tertiary forests they are one of the last remnants of natural deciduous forests in the world and one of the most important biodiversity hotspots of West Eurasia (Akhani et al. 2010). Being extremely lush and humid the forests' annual rainfall ranges between 530 mm in the east and 1350 mm in the west on average, reaching up to 2000 mm in the west (Sagheb-Talebi et al. 2014). The human population of the Hyrcanian forests is estimated at 1.3 million people living in 200,000 households (Sagheb-Talebi et al. 2014). In our survey crop cultivation and livestock breeding were the most important livelihood activities. The average farm size reported was 3.9 ha and a variety of 59 crops and fruits were grown. Rice, wheat and barley are the major crops in this area. Our surveyed village heads estimated pasture around villages to be 703.3 ha on average (range 6-10000 ha). Local people mainly raise cattle, sheep and goats, which usually graze freely inside forests and crop fields. We conducted the survey in six study sites located across the Hyrcanian forests (Fig. 1): (1) Golestan National Park (874.02 km2) and its well-connected buffer protected areas, (2) Zav A and B protected area (143.23 km2) to the north-west and west, and (3) Loveh protected area (35.89 km2) to the south west, (4) Dodangeh Wildlife Refuge (206.05 km2) and (5) Baliran non-protected area (155.31 km2) in central Mazandaran Province, and (6) Lisar protected area (PA), (311.42 km2) in Gilan Province.

Data collection and survey design

Using a semi-structured questionnaire we interviewed 162 respondents in 45 villages, consisting of 42 village representatives and 120 farmers (Table 1). Villages located in- and outside of protected and non-protected areas were selected on site, using contrast and snowball sampling (Newing 2011). Villages that indicated contrasting locations (such as villages on the other side of a PA or non-protected area, different altitudes and proximities to the borders of a PA or non-protected area) were particularly targeted in the attempt to include various influencing factors on conflicts. Within a distance of 5 km we selected 33 villages outside protected areas. Collected conflict data referred to the one-year period between October 2014 and September 2015.

Data analysis

Data was analysed qualitatively and quantitatively. Open questions were analysed using qualitative summarizing content analysis (Mayring 2000). The analysis was performed using SPSS 23 (IBM Deutschland GmbH, Ehningen, Germany) and R-statistics vr.3.3.2 software. In a first step, we identified primary conflict species and their main conflict types. In a second step, we analysed socio-economic and ecological variables associated with these conflicts through logistic regression and generalized linear model for each species (Backhaus et al. 2011; Abdey 2014). For each regression method, we applied a single model using the backward stepwise component with elimination at p=0.05 level to retain best predictors (Abdey 2014). We used Hosmer and Lemeshow-Goodness of fit to assess the fit and test the power of the logistic regression model (Backhaus et al. 2011). Response variables were (1) conflict presence (binary coded 1/0), (2) crop field size damage (m2) and (3) livestock loss (count). Predictor variables of both regressions included (i) crop and fruit variety of farmers (count), (ii) crop area of farmers (m2), (iii) livestock holding of farmers (count), (iv) elevation of surveyed village (m) and (v) village distance to nearest PA/non-PA (km). We used prey richness as a further predictor to assess the role of wild prey in depredation of livestock by wolf.

Crop and fruit variety was measured by the total amount of species being cultivated, and livestock holding was measured by the number of sheep and goats kept. We measured elevation of surveyed villages by using a hand held GPS 64s Garmin device. Distances between villages and PA borders were calculated using ArcGIS 10.3.1 (ESRI, Redlands, USA). In the logistic regression and generalized linear model we used spearman's rank correlation (rs) to explore relations among variables (Gelman 2008).

We then performed generalized linear mixed models (GLMMs). We analysed the presence-absence of main conflict types of wild boar and wolf conflicts and accounted for potential pseudoreplication (Crawley 2015). We checked multi-collinearity of further predictors by calculating variance inflation factors (VIFs) and excluded all predictor variables with VIFs > 3 using "corvif" function of R’s "usdm" package (Zuur et al. 2010). We incorporated "household" as the random effect in our modelling, because we recorded conflict incidences from several respondents in each village during one year, which might introduce a potential bias of pseudoreplication (Dorman et al. 2007; Grueber et al. 2011). We analyzed the presence-absence of conflict incidences of both species using "lmer" functions of GLMM (logit-link, binomial distribution error) in "lme4" package of R (Bolker et al. 2008) and ranked the models based on their delta (Δ) by multi-model inference “MuMIn” in R (Barton 2009). We then used Akaike's Information Criterion (AIC) to select the best models with $\Delta < 2$ (Burnham & Anderson 2002).

In the GLMM analysis we incorporated the following predictor variables: mean elevation (km), prey richness (number of prey species), crop variety, location of villages (inside or outside of protected areas), enhanced vegetation index (EVI), distance to nearest protected areas (km) and human population density (people/km^2). We extracted these predictors from sampled 4×4 km grid cells, which included our surveyed villages. The prey richness data (i.e. red deer, roe deer and wild boar) came from an independent study implemented during 2015-2016, which focused on occupancy of large mammals across the Hyrcanian forests (unpublished data). The presence-absence data of each species signs (e.g. spoor, wallowing spots, scrapes, resting places, scats, and feeding signs) and direct sightings were assigned along the random trails of every 200 m per grid cell. EVI was extracted from 30-m MOD13Q1 maps of the US Geological Survey (http://glovis.usgs.gov) and the mean value was calculated for each cell using ENVI v. 4.8 (Exelis Visual Information Solutions, Boulder, Colorado) (Jiang 2008). We measured mean elevation from the 90 m Shuttle Radar Topography Mission (SRTM) (http://glovis.usgs.gov). From maps produced by ArcGIS we detected whether villages were located in or outside of a protected area. We calculated average human densities in ArcGIS 10.2.1 using 1km resolution from Socio Economic Data and Applications Centers (SEDAC) (columbia.edu, 2015). All predictors were re-scaled in dividing them by twice the standard deviation in order to reduce overdispersion (Quinn & Keough 2002; Gelman 2008).

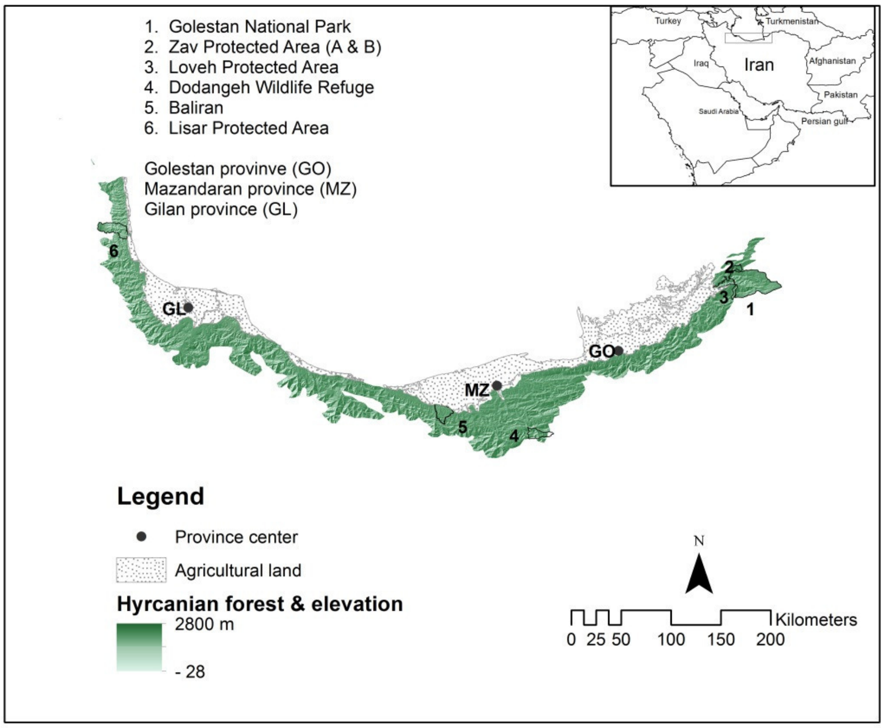

Figure 1. The six study sites within the Hyrcanian forests, represented as a narrow green band around the southern coast of the Caspian Sea.

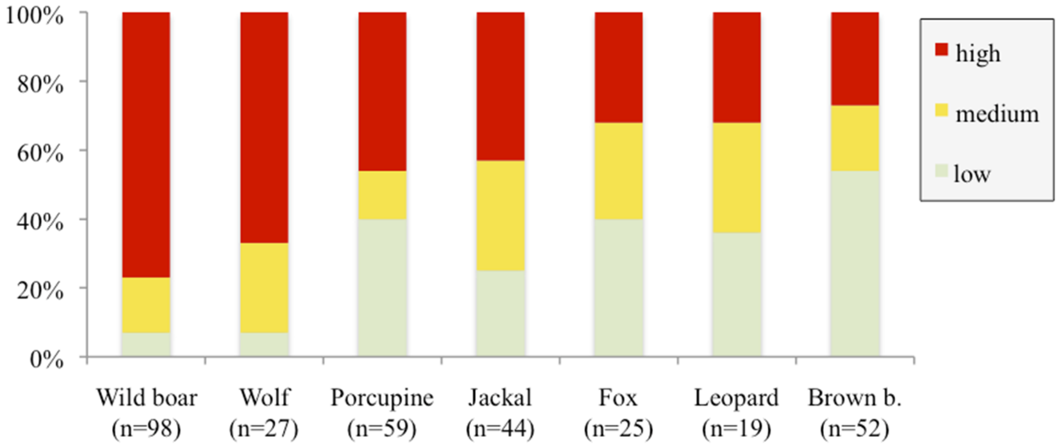

Figure 2. Farmers’ estimates of main conflict severity (n=reported cases of conflicts).

6.3. Results

Respondents reported on 15 wildlife species to exist near the surveyed villages. We identified seven primary conflict species, of which wild boar and wolf conflicts were perceived to be the most severe concern for farmers (Fig. 2). These two top ranked conflict species are subject to our investigations.

Wild boar conflicts were reported in 98 cases of which 94% experienced crop loss (n=92). Only 6% of all wild boar conflicts referred to fruit damages, making the species a primary crop raider. Wild boar damaged rice fields in every second event (n=49), followed by wheat, pumpkin and barley. The species damaged the largest area of crop fields as 80% of the total reported damage size was related to wild boars. The species damaged an average size of 0.257 ± 0.538 ha with a maximum of 3 ha (Dodangeh WR). Most wild boar conflicts occurred in Golestan (46%) and Dodangeh study site (24%). A significant predictor for wild boar crop damage presence was crop variety ($p < 0.05$). A rise in crop variety by two standard deviations increased the probability of conflict by 8.7 times (Table 2). The second predictor of conflict presence was crop area ($p < 0.05$) which increased the probability of conflict by more than 50 times. The model correctly predicted conflict presence in 94.7% of cases and absence in 29.4% of cases. GLM analysis showed that elevation ($p < 0.001$) and crop variety ($p < 0.01$) positively affected areas damaged by wild boar (Table 2).

There was a positive correlation between crop variety and elevation ($r_s=0.309$, $p < 0.01$, $n = 101$). We also found a negative correlation between crop variety and distance to PA/non-PA ($r_s = -0.2$, $p < 0.05$, $n = 114$). Further results indicated that crop variety was higher on fields which were associated with villages inside protected or non-protected area ($r_s = -0.19$, $p < 0.05$, $n = 114$). Moreover, crop area extended with higher village elevation ($r_s = 0.204$, $p < 0.05$, $n = 93$), but larger crop area was not related to crop area damage.

All reported wolf conflicts accounted for livestock losses and were most frequent on sheep attacks (81%) with an average kill of 3.3 ± 2.8 individuals per attack. Others included cattle and goat attacks. 67% of wolf conflicts occurred in the Golestan NP. No wolf conflicts were reported in Lisar and Baliran study site. We determined wolf attacks on sheep and goat as the main type of human-wolf conflicts. We obtained village elevation ($p < 0.01$) as the only significant predictor to explain wolf attacks on sheep and goat (Table 3). A rise in elevation by two standard deviations increases the probability of conflict by over 15 times. The model correctly predicted conflict presence in 61.1% of cases and absence in 84.6% of cases. In the additional analysis with GLMM the presence of human-wild boar conflicts increased with crop diversity and elevation, as well as in areas with lower human density, EVI and distances to protected areas (Table 4). Whether conflict villages were inside or outside of protected areas was not important for human-wild boar conflicts. The presence of human-wolf conflicts increased with elevation, EVI, and human density (Table 4). At the same time, wolf attacks on sheep and goat increased in areas with lower prey richness and in areas located closer to, or inside, protected areas.

Table 1. Interview questions (VH=village head, F=farmer)

Topic of interest	Questions	Assessment Scale
Land Use	How big is the grazing area (ha) in and around your village? (VH)	Open
	How big is your cropping area? (F)	Open
	What type of crops do you grow? (F)	Open
	How many individual livestock species do you have? (F)	Open
Wildlife conflicts	Define the conflicts with wildlife that you have experienced as a farmer in the last 12 months. (F)	Open
	a) Type of conflict	1=crop damage 2=horticulture damage 3=livestock loss
	b) Severity	1=low 2=medium 3=high
	c) Size being damaged	Open
	d) Livestock loss	Open
Socio-demographics	Age, gender, ethnic group	Open

Table 2. The output of logistic regression model of wild boar crop damage presence-absence and GLM model of crop area damaged by wild boar.

Predictor variable	Slope B	S.E.	Wald / Chi-Square	Sig.	Exp(B)
Wild boar crop damage presence-absence					
Crop variety	2.166	.917	5.579	.018	8.725
Crop area	3.943	1.897	4.32	.038	51.563
Constant*	-.809	.829	.953	.329	.445
Hosmer-Lemeshow GOF	$Chi^2 = 7.699$	p=.463			
Wild boar crop area damage					
Elevation	1.606	.1827	77.282	.000	
Crop variety	.729	.2397	9.261	.002	
Intercept	-3.702	.3777	96.094	.000	

Table 3. The output of logistic regression model of wolf conflict presence-absence. Degrees of freedom in the model were defined at 1.

Predictor variable	Slope B	S.E.	Wald	Sig.	Exp(B)
Elevation	2.730	.941	8.417	.004	15.33
Constant	-1.951	.637	9.368	.002	.142
Hosmer-Lemeshow GOF	$Chi^2 = 21.558$	p=.006			

Table 4. Summary of the top generalized linear mixed models (GLMMs) for human-wild boar and human-wolf conflicts in the Hyrcanian forest, Iran.

Rank	Model covariates	Parameter				Coefficient (± standard error)					
Wild boar conflict presence-absence											
		K	AIC	Δ	AIC_w	Crpvr	DisPA	EVI	HP	InOuPA	Elev
1	Crpvr	3	70.3	0.00	0.35	1.68 ± 1.01					
2	Crpvr + HP	4	71.3	1.03	0.21	1.70 ± 1.00			-0.76 ± 0.86		
3	Crpvr + Elev	4	72.8	1.49	0.17	1.57 ± 0.99					1.21 ± 1.97
4	Crpvr + DisPA	4	72.3	1.98	0.13	1.66 ± 1.01	-0.14 ± 1.13				
5	Crpvr + EVI	4	72.3	1.99	0.13	1.70 ± 1.03		-0.09 ± 1.06			
Wolf conflict presence-absence											
		K	AIC	Δ	AIC_w	DisPA	Elev	EVI	HP	InOuPA	Prey
1	Elev	3	83.6	0.00	0.33		3.34 ± 1.53				
2	Elev + Prey	4	85.1	1.52	0.16		4.06 ± 2.01				-0.99 ±1.53
3	Elev + InOuPA	4	85.4	1.83	0.13		3.35 ± 1.56			-0.44 1.04	
4	Elev + EVI	4	85.5	1.94	0.13		3.14 ± 1.60	0.28 ± 1.12			
5	Elev + DisPA	4	85.6	1.98	0.12	- 0.14 ± 1.04	3.34 ± 1.53				
6	Elev + HP	4	85.6	1.99	0.12		3.38 ± 1.62		0.06 ± 0.80		

Abbreviation of covariates: Crpvr, crop variety; Elev, elevation (m); EVI, Enhanced Vegetation Index; DisPA, distance to protected area (km); HP, human population density (persons/km^2); InOuPA, villages inside (0) and outside (1) of protected areas (PAs); Prey (0-3), Abbreviation of parameters: AIC, Akaike's Information Criterion; AIC_w, Akaike's model weight (%); Δ, model delta; K, number of model parameters.

6.4. Discussion

Crop damage by wild boar

As elsewhere in Golestan province, we found wild boars to be a key driver of crop loss also in the Hyrcanian forests, inflicting serious consequences to farming households, claiming that half of their production is affected (see also Ghadirian and Raeesi 2015). A similar level of damage in rice farms of Mazandaran Province was documented (Ghadirian & Raeesi 2015) and the species is one of the most destructive species in regard to crop damage (Barrios-Garcia & Ballari 2012; Pandey et al. 2016). The frequency of wild boar conflicts is linked to the species high abundance in the Hyrcanian forests, which in turn may be due to religious beliefs resulting into low poaching pressure in Iran (Ghoddousi et al. 2017a).

Wild boars can be tolerant to human activities as they typically roam along edges of habitats, consume and trample crops, and have a broad diet, including rice, wheat, sorghum, potatoes, beetroots, maize, beans and barley (Pandey et al. 2016, Schley & Roper 2003). The strong predictability of wild boar conflict by greater crop variety and larger crop areas can be explained by richer proteins, carbohydrates, and mineral nutrients of crops compared to wild plants and animals (Sukumar 1989). Since energy-rich plants are an important component of the species' diet, our results may suggest that foraging behaviour of wild boars is dictated more by quality than by quantity of food. Because wild boars invest more time to spot energy-rich plants they trample down more crops and extend the damaged area.

We found villages located at shorter distances to protected areas being more vulnerable to human-wild boar conflicts, showing that damages are not distributed randomly to forest edges (Thurfjell et al. 2009). Wild boar field damage is often located much closer to the forest edge than expected by chance, facilitating damage by combining escape cover with proximity to food (Drozd 1988; Lemel 1999; Thurfjell et al. 2009). Landscapes at higher elevations where crop fields belong to remote and low populated villages can therefore be vulnerable conflict places. Hence, human-wild boar conflicts in the Hyrcanian forests may be more common when agricultural fields are located within landscapes where human densities are low and escape opportunities for the animals through nearby forest edges are high. However, the presence of human-wild boar conflicts was negatively associated with vegetation cover (EVI) of the surroundings of crop fields. This might be due to seasonal variation in feeding ground selection and agricultural fields as a food source are of great interest for wild boars in seasons when crops are ripe (Thurfjell et al. 2009). We speculate that the deciduous forests as a natural habitat become less attractive during summer and autumn when crops are abundant. Food sources within dense vegetation cover might then be generally neglected or being used as refuge during inactive times as wild boars are nocturnal animals. Conversely, during winter and spring the forests might be preferably selected by the species as they provide good cover with sufficient feeding opportunities.

Livestock predation by wolf

Also wolves have been identified to be of serious concern to farmers. This is consistent with recent investigations on human-large carnivore conflicts in Iran (Behdarvand et al. 2014; Habibzadeh 2016). In line with our results, small stock domestic animals like sheep and goat are preferred wolf prey, but against our expectations, wolf predation on sheep was not influenced by stock numbers. Instead, we found that livestock depredation by wolf to increase with elevation, deficiency of wild prey richness, in areas located closer to, or inside, protected areas and to be positively associated with EVI and human density. Various interrelated factors explain these findings: First, wolves are inclined to choose the easiest prey available (Meriggi et al. 1996; Imbert et al. 2016) and where sheep and wolves are present in the same area, livestock depredation invariably occurs regardless of the presence or absence of wild prey (Gazzola et al. 2005). Second, if regions have very low densities of wild prey and high densities of livestock then wolf’s survival much depends on the presence of livestock (Vos 2000). Indeed, the abundance of wild ungulates is low in the Hyrcanian forests (Ghoddousi et al. 2017b). Especially the Caspian red and roe deer suffer from high levels of poaching, habitat fragmentation and loss, and ineffective management (Soofi et al. 2017a, 2017b). Hence, we may conclude that low densities of wild ungulates are a major threat to wolves in the Hyrcanian forests by increasing human-wolf conflicts. Third, favourable and better environmental conditions for livestock grazing in high mountain ranges in summer most likely have triggered wolf attacks (Behdarvand et al. 2014).

Consequently, more herders temporarily live in these highlands, which may suggests that the lower the human presence the higher the probability of livestock depredation risk by wolves. Fourth, wolves often avoid human settlements (Eggermann et al. 2011). Remote forest regions with dense vegetation cover and the proximity to borders of PAs provide a favourable condition for wolves as they can escape from persecution.

Although shepherd and dog absence was not claimed to be a cause of conflict, this result needs further verification as it contradicts data from a small part of the study region (Khorozyan et al. 2017). Unattended livestock, which is grazing away from corrals, as well as the use of unsuitable or untrained guarding dogs can clearly trigger predation of ungulates by carnivores (Ogada et al. 2003; Khorozyan et al. 2017). Presumably across our study region, there exists ineffectual herding practice due to an inability of shepherds and dogs to keep wolves away from herded animals (Dalmasso et al. 2012; Behdarvand et al. 2014). This requires attention when addressing mitigation of human-wildlife conflicts in the Hyrcanian forests.

Potential mitigation measures

Livestock and crop loss can be prevented when the right tools are being used. For example, to avoid great crop damage to wild boars without high guarding investment, we propose that palatable seasonal crops such as maize should generally not be planted on forest edges (Naughton-Treves 1998). Further, we see the necessity of physical barriers around crop fields and livestock. Incentives are there where fields are of small or medium sizes as the construction of fences or walls are at moderate costs. However, farmers managing large crop fields and pastures are likely to be challenged with much greater investments for fencing or building walls. Physical barriers are most practicable when they are implemented from conflict species-specific perspectives. To prevent wild boar and carnivore conflicts, electric fences can greatly reduce depredation (Ogada et al. 2003; Honda et al. 2014). It is argued that they are more durable than simple, local-material fencing methods (Distefano 2002). We underline the importance of electric fencing and recommend its installation around whole pastures, night-time corrals, and around crop fields located at forest edges, instead of wire fences. A drawback is that electric fences include higher installation costs compared to simple fencing and maintenance needs constant insulation (Distefano 2002). We propose cost effective investments by sharing construction costs where fields are interconnected. Several farmers could then proportionally invest in fencing.

Inefficient herding practices can greatly influence predator attacks, especially when livestock is allowed to graze freely inside forests (Ogada et al. 2003). Although it is common throughout the study region that livestock is being returned to thorn corrals at night and offspring is being kept in corrals or in proximity to humans, many poor and unqualified shepherds fail to guard their stock appropriately. First and foremost, free grazing livestock needs to be sufficiently supervised and permanently safeguarded against carnivore predation. This requires skilled herders, properly built enclosures and the use of trained guarding dogs. When livestock is moved to higher elevations during warm seasons, livestock could be prevented from straying around, kept away from thick cover, and should graze in reduced herd sizes and not in areas where predators are known to be around (Sillero-Zubiri & Laurenson 2001; Ogada et al. 2003). Further, it is important to remove, bury or burn carcasses in and around the area where livestock is kept (Behdarvand et al. 2014). Another approach to improve the efficacy of shepherding in the region is seen in joint herding by building larger flocks (Khorozyan et al. 2017). Through collective livestock guarding the individuals can not only share responsibility over stock safety but also strengthen mutual learning. Moreover, we recommend cost effective investments in installation of electric fences around whole pastures and night-time corrals, instead of wire fences.

Finally, mitigating human-wolf conflicts in the Hyrcanian forests must not ignore the low availability of wild prey. The most effective mitigation strategy for livestock predation is to restore the wild prey community, especially populations of threatened species across the whole forest range, through connected and larger habitats and increased protection outside protected areas (Sidorovich et al. 2003; Imbert et al. 2016). Also, it is crucial to effectively reduce the level of poaching pressure in protected areas by creating alternative livelihoods for local poachers (Duffy et al. 2016). Where poachers lack the educational capacity a manageable sector for alternative activities can be ecotourism (Ghoddousi et al. 2017b).

These direct and indirect methods shall not be regarded in isolation when attempting to mitigate or prevent wildlife conflicts as they vary in their effectiveness. Species' behaviour as well as local-specific settings must be considered in search for appropriate measures. A complementary use of conflict mitigation methods is strongly recommended and systematic monitoring on the ground is needed to identify which methods perform and which do not.

Human-wildlife conflicts in the Hyrcanian forests are a growing conservation challenge. Not only do conservationists face notable challenges in the protection of endangered wildlife, but also rural people's livelihoods can be substantially affected. Wild boars and wolves are primary conflict species in the region. Our results show that negative encounters with these species may result in irreplaceable crop and livestock losses across a large-scale landscape. Knowing about socio-economic and ecological factors that may influence conflicts, and biological traits of conflict species helps to identify core conservation challenges and key solutions in this landscape. We conclude that conflict mitigation approaches should primarily focus on non-lethal control methods as they can offer a proactive and “green” tool to allow the co-existence of farming activities and wildlife. Yet, the long-term success of any strategy will depend on local community attitudes and tolerance of wildlife to a level that sustains viable wildlife populations. In the Hyrcanian forests, conflict mitigation implications require stakeholder commitment, capacity building, appropriate law enforcement, operated conservation policies and mutual benefits between conservationists and local farmers. Furthermore, conservation managers need more information on how and why intense conflicts occur. Survey data can provide insights into possible causal factors driving and mitigating local human-wildlife conflicts and respective monitoring should be applied in the future.

6.5. Supplementary material

Appendix 1: Data analysis output

Analysis outputs: Logistic regression and generalized linear model applied in for wild boar crop damage presence – LR

Table1: Variables in the Equation – wild boar crop damage

		B	S.E.	Wald	df	Sig.	Exp(B)	95% C.I.for EXP(B)	
								Lower	Upper
Step 1[a]	Elevation_dividedby2SD by Village_in_outside_PA(1)	1.037	2.446	.180	1	.672	2.820	.023	340.565
	Crop_area_SD2 by Village_in_outside_PA(1)	-.695	6.928	.010	1	.920	.499	.000	394019.443
	Distance_dividedby_2SD	-.861	.605	2.025	1	.155	.423	.129	1.384
	Elevation_dividedby2SD	.681	.707	.928	1	.335	1.976	.494	7.895
	Crop_variety_dividedby2SD	1.925	.930	4.290	1	.038	6.857	1.109	42.400
	Crop_area_SD2	4.097	2.040	4.035	1	.045	60.187	1.104	3280.270
	Constant	-1.089	.875	1.549	1	.213	.337		
Step 2[a]	Elevation_dividedby2SD by Village_in_outside_PA(1)	.896	1.962	.209	1	.648	2.450	.052	114.638
	Distance_dividedby_2SD	-.847	.588	2.074	1	.150	.429	.135	1.358
	Elevation_dividedby2SD	.696	.692	1.013	1	.314	2.006	.517	7.782
	Crop_variety_dividedby2SD	1.916	.923	4.309	1	.038	6.795	1.113	41.484
	Crop_area_SD2	4.087	2.034	4.038	1	.044	59.568	1.106	3207.641
	Constant	-1.102	.866	1.621	1	.203	.332		
Step 3[a]	Distance_dividedby_2SD	-.916	.579	2.508	1	.113	.400	.129	1.243
	Elevation_dividedby2SD	.751	.696	1.166	1	.280	2.120	.542	8.291
	Crop_variety_dividedby2SD	1.991	.922	4.668	1	.031	7.324	1.203	44.582
	Crop_area_SD2	4.039	2.031	3.954	1	.047	56.797	1.060	3044.663
	Constant	-1.085	.869	1.559	1	.212	.338		
Step 4[a]	Distance_dividedby_2SD	-.971	.570	2.906	1	.088	.379	.124	1.157
	Crop_variety_dividedby2SD	2.166	.917	5.579	1	.018	8.725	1.446	52.650
	Crop_area_SD2	3.943	1.897	4.320	1	.038	51.563	1.252	2123.351
	Constant	-.809	.829	.953	1	.329	.445		

a. Variable(s) entered on step 1: Elevation_dividedby2SD * Village_in_outside_PA , Crop_area_SD2 * Village_in_outside_PA , Distance_dividedby_2SD, Elevation_dividedby2SD, Crop_variety_dividedby2SD, Crop_area_SD2.

Table 2: Hosmer and Lemeshow Test - wild boar crop damage

Step	Chi-square	df	Sig.
1	13.368	8	.,100
2	9.581	8	.296
3	9.972	8	.267
4	7.699	8	.463

Table 3: Classification Table wild boar crop damage

Observed			Predicted Cropconflic Absence	Predicted Cropconflic presence	Percentage correct
Step 1	Cropconflict_presence	absence	5	12	29.4
		presence	4	72	94.7
	Overall Percentage				82.8
Step 2	Cropconflict_presence	absence	5	12	29.4
		presence	4	72	94.7
	Overall Percentage				82.8
Step 3	Cropconflict_presence	absence	5	12	29.4
		presence	4	72	94.7
	Overall Percentage				82.8
Step 4	Cropconflict_presence	absence	5	12	29.4
		presence	4	72	94.7
	Overall Percentage				82.8

Table 4: Parameter Estimates - wild boar crop area damage - wild boar crop area damage – GLM (Tweedie log-link)

			95% Wald Confidence Interval		Hypothesis Test		
Parameter	B	Std. Error	Lower	Upper	Wald Chi-Square	df	Sig.
(Intercept)	-3.702	.3777	-4.442	-2.962	96.094	1	.000
Distance_dividedby_2SD	-.111	.2789	-.657	.436	.158	1	.691
Elevation_dividedby2SD	1.606	.1827	1.248	1.965	77.282	1	.000
Crop_variety_dividedby2SD	.729	.2397	.260	1.199	9.261	1	.002
Crop_area_SD2	.190	.3886	-.571	.952	.240	1	.624
[Village_in_outside_PA=1] * Crop_area_SD2	.012	.5335	-1.034	1.057	.000	1	.982
[Village_in_outside_PA=2] * Crop_area_SD2	0[a]	.	.	.	.	.	.
[Village_in_outside_PA=1] * Elevation_dividedby2SD	-.928	.4887	-1.885	.030	3.602	1	.058
[Village_in_outside_PA=2] * Elevation_dividedby2SD	0[a]	.	.	.	.	.	.
(Scale)	.458[b]	.0552	.361	.580			

a. Set to zero because this parameter is redundant.
b. Maximum likelihood estimate.
Dependent Variable: WB_crop_damage_size_2SD
Model: (Intercept), Distance_dividedby_2SD, Elevation_dividedby2SD, Crop_variety_dividedby2SD, Crop_area_SD2, Village_in_outside_PA * Crop_area_SD2, Village_in_outside_PA * Elevation_dividedby2SD

Table 5: Goodness of Fit[a] - wild boar crop area damage

	Value	df	Value/df
Deviance	45.084	69	.653
Scaled Deviance	98.525	69	
Pearson Chi-Square	4.606	69	.88
Scaled Pearson Chi-Square	88.738	69	
Log Likelihood[b]	9.246		
Akaike's Information Criterion (AIC)	-2.491		
Finite Sample Corrected AIC (AICC)	-.342		
Bayesian Information Criterion (BIC)	16.154		
Consistent AIC (CAIC)	24.154		

Dependent Variable: WB_crop_damage_size_2SD
Model: (Intercept), Distance_dividedby_2SD, Elevation_dividedby2SD, Crop_variety_dividedby2SD, Crop_area_SD2, Village_in_outside_PA * Crop_area_SD2, Village_in_outside_PA * Elevation_dividedby2SD
a. Information criteria are in smaller-is-better form.
b. The full log likelihood function is displayed and used in computing information criteria

Table 6: Correlations wild boar conflicts

Parameter			Village_in_outside_PA	Distance_dividedby_2SD	Elevation_dividedby2SD	Crop_variety_dividedby2SD	Crop_area_SD2	WB_crop_damage_size_2SD
Spearman's rho	Village_in_outside_PA	Correlation Coefficient	1.000	.676**	-.050	-.190*	.078	-.140
		Sig. (2-tailed)	.	.000	.621	.043	.441	.186
		N	114	114	101	114	101	91
	Distance_dividedby_2SD	Correlation Coefficient	.676**	1.000	-.139	-.200*	.159	-.149
		Sig. (2-tailed)	.000	.	.164	.033	.112	.158
		N	114	114	101	114	101	91
	Elevation_dividedby2SD	Correlation Coefficient	-.050	-.139	1.000	.309**	.204*	.460**
		Sig. (2-tailed)	.621	.164	.	.002	.050	.000
		N	101	101	101	101	93	81
	Crop_variety_dividedby2SD	Correlation Coefficient	-.190*	-.200*	.309**	1.000	.161	.470**
		Sig. (2-tailed)	.043	.033	.002	.	.108	.000
		N	114	114	101	114	101	91
	Crop_area_SD2	Correlation Coefficient	.078	.159	.204*	.161	1.000	.163
		Sig. (2-tailed)	.441	.112	.050	.108	.	.140
		N	101	101	93	101	101	83
	WB_crop_damage_size_2SD	Correlation Coefficient	-.140	-.149	.460**	.470**	.163	1.000
		Sig. (2-tailed)	.186	.158	.000	.000	.140	.
		N	91	91	81	91	83	91

**. Correlation is significant at the 0.01 level (2-tailed). *. Correlation is significant at the 0.05 level (2-tailed).

Wolf livestock predation presence – LR

Table 33: Variables in the Equation - wolf livestock predation presence

								95% CI for EXP(B)	
		B	S.E.	Wald	df	Sig.	Exp(B)	Lower	Upper
Step 1[a]	Elevation_dividedby2SD by Village_in_outside_PA(1)	1.102	1.483	.552	1	.457	3.011	.165	55.089
	Village_in_outside_PA(1) by sheepgoat_holding_dividedbySDx2	-.603	2.166	.078	1	.781	.547	.008	38.186
	Elevaiton_dividedby2SD	2.112	.984	4.611	1	.032	8.268	1.202	56.849
	Distance_dividedby2SD	.006	.797	.000	1	.994	1.006	.211	4.792
	sheepgoat_holding_dividedbySDx2	1.370	1.592	.741	1	.389	3.936	.174	89.119
	Constant	-2.081	.761	7.480	1	.006	.125		
Step 2[a]	Elevation_dividedby2SD by Village_in_outside_PA(1)	1.100	1.446	.578	1	.447	3.004	.176	51.158
	Village_in_outside_PA(1) by sheepgoat_holding_dividedbySDx2	-.607	2.110	.083	1	.774	.545	.009	34.060
	Elevation_dividedby2SD	2.112	.984	4.611	1	.032	8.268	1.202	56.852
	sheepgoat_holding_dividedbySDx2	1.372	1.566	.768	1	.381	3.944	.183	84.929
	Constant	-2.078	.666	9.728	1	.002	.125		
Step 3[a]	Elevation_dividedby2SD by Village_in_outside_PA(1)	.895	1.248	.515	1	.473	2.448	.212	28.230
	Elevation_dividedby2SD	2.183	.964	5.135	1	.023	8.877	1.343	58.677
	sheepgoat_holding_dividedbySDx2	1.089	1.068	1.038	1	.308	2.971	.366	24.112
	Constant	-2.079	.660	9.911	1	.002	.125		
Step 4[a]	Elevation_dividedby2SD	2.471	.945	6.835	1	.009	11.834	1.856	75.454
	sheepgoat_holding_dividedbySDx2	1.204	1.089	1.221	1	.269	3.332	.394	28.167
	Constant	-2.131	.672	10.040	1	.002	.119		
Step 5[a]	Elevation_dividedby2SD	2.730	.941	8.417	1	.004	15.330	2.425	96.931
	Constant	-1,951	,637	9.368	1	.002	.142		

a. Variable(s) entered on step 1: Elevation_dividedby2SD * Village_in_outside_PA , Village_in_outside_PA * sheepgoat_holding_dividedbySDx2 , Elevation_dividedby2SD, Distance_dividedby2SD, sheepgoat_holding_dividedbySDx2

Table 34: Hosmer and Lemeshow Test- wolf livestock predation presence

Step	Chi-square	df	Sig.
1	10.040	8	.262
2	10.044	8	.262
3	9.869	8	.274
4	7.867	8	.447
5	21.558	8	.006

Table 35: Classification Table - wolf livestock predation presence

Observed			Predicted		
			sheepandgoatconflict_presence		Percentage Correct
			absence	presence	
Step 1	sheepandgoatconflict_presence	absence	23	3	88.5
		presence	7	11	61.1
	Overall Percentage				77.3
Step 2	sheepandgoatconflict_presence	absence	23	3	88.5
		presence	7	11	61.1
	Overall Percentage				77.3
Step 3	sheepandgoatconflict_presence	absence	23	3	88.5
		presence	7	11	61.1
	Overall Percentage				77.3
Step 4	sheepandgoatconflict_presence	absence	22	4	84.6
		presence	7	11	61.1
	Overall Percentage				75.0
Step 5	sheepandgoatconflict_presence	absence	22	4	84.6
		presence	7	11	61.1
	Overall Percentage				75.0

Wolf livestock loss - GLM

Table 36: Parameter Estimates - wolf livestock loss

Parameter	B	Std. Error	95% Wald Confidence Interval		Hypothesis Test		
			Lower	Upper	Wald Chi-Square	df	Sig.
(Intercept)	-2.715	.6287	-3.947	-1.483	18.647	1	.000
Elevation_dividedby2SD	1.373	.7782	-.152	2.899	3.115	1	.078
Distance_dividedby2SD	-1.157	.8939	-2.909	.595	1.675	1	.196
sheepgoat_holding_dividedbySDx2	.578	.6244	-.645	1.802	.858	1	.354
[Village_in_outside_PA=1] * Elevation_dividedby2SD	.187	1.0705	-1.911	2.285	.031	1	.861
[Village_in_outside_PA=2] * Elevation_dividedby2SD	0[a]	.	.	.	.	.	.
[Village_in_outside_PA=1] * sheepgoat_holding_dividedbySDx2	-.256	1.1565	-2.522	2.011	.049	1	.825
[Village_in_outside_PA=2] * sheepgoat_holding_dividedbySDx2	0[a]	.	.	.	.	.	.
(Scale)	1.412[b]	.2906	.943	2.113			

Dependent Variable: sheepandgoat_loss_dividedbySDx2

Model: (Intercept), Elevation_dividedby2SD, Distance_dividedby2SD, sheepgoat_holding_dividedbySDx2, Village_in_outside_PA * Elevation_dividedby2SD, Village_in_outside_PA * sheepgoat_holding_dividedbySDx2

a. Set to zero because this parameter is redundant.

b. Maximum likelihood estimate.

Table 37: Goodness of Fit[a] -- wolf livestock loss

	Value	df	Value/df
Deviance	44,118	36	1,225
Scaled Deviance	31,255	36	
Pearson Chi-Square	43,297	36	1,203
Scaled Pearson Chi-Square	30,674	36	
Log Likelihood[b]	-26,119		
Akaike's Information Criterion (AIC)	66,238		
Finite Sample Corrected AIC (AICC)	69,532		
Bayesian Information Criterion (BIC)	78,402		
Consistent AIC (CAIC)	85,402		

Dependent Variable: sheepandgoat_loss_dividedbySDx2
Model: (Intercept), Elevation_dividedby2SD, Distance_dividedby2SD, sheepgoat_holding_dividedbySDx2,Village_in_outside_PA * Elevation_dividedby2SD, Village_in_outside_PA * sheepgoat_holding_dividedbySDx2
a. Information criteria are in smaller-is-better form.
b. The full log likelihood function is displayed and used in computing information criteria.

Table 38: Correlations *- wolf livestock loss

			Village_in_outside_PA	Elevation_dividedby2SD	Distance_dividedby2SD	sheepgoat_holding_dividedbySDx2
Spearman's rho	Village_in_outside_PA	Correlation Coefficient	1.000	-.068	.710[**]	-.189
		Sig. (2-tailed)	.	.494	.000	.208
		N	118	102	118	46
	Elevation_dividedby2SD	Correlation Coefficient	-.068	1.000	-.158	.028
		Sig. (2-tailed)	.494	.	.114	.856
		N	102	102	102	44
	Distance_dividedby2SD	Correlation Coefficient	.710[**]	-.158	1.000	-.022
		Sig. (2-tailed)	.000	.114	.	.886
		N	118	102	118	46
	sheepgoat_holding_dividedbySDx2	Correlation Coefficient	-.189	.028	-.022	1.000
		Sig. (2-tailed)	.208	.856	.886	.
		N	46	44	46	46

**. Correlation is significant at the 0.01 level (2-tailed).

Chapter 7

Synthesis

A typical local community living inside protected areas (Lisar PA). *Photo by M. Soofi.*

Overall conclusion

This PhD thesis shows that despite of the growing tendency in the establishment of protected areas (PAs) in Iran and their associated law enforcement measures, the known threats of poaching, livestock grazing and logging are still dramatically affecting the distribution and viability of large mammal populations across protected and non-protected areas. Livestock has already become hyperabundant, particularly due to logging that opens up good quality grazing areas, even inside the core zones of PAs.

I found that livestock grazing strongly and negatively affects the distribution of Persian leopard, Caspian red deer and roe deer. These species are threatened either globally or nationally and have a very patchy distribution in the Hyrcanian forest. The leopard and the two deer species now appear to be locally extinct in some PAs, and red deer distribution is strongly affected by logging. Most importantly, grazing intensity is high in PAs (IUCN category V), no-hunting areas, non-protected areas and wildlife refuges (IV), but is significantly lower in national parks (II). The PAs of category V cover 66%, wildlife refuges 0.02% and national parks 0.10% of all reserves in the Hyrcanian forest. Also, 80% of their lands are officially allowed for livestock grazing what has direct implications for existing conservation policies.

I was not able to address the effects of poaching on the distribution of targeted mammals by occupancy modelling, possibly because of low detection probability of poaching occurrence signs. However, I assessed the effects of poaching pressure on livestock depredation rates by Persian leopard (*Panthera pardus saxicolor*) and grey wolf (*Canis lupus*). As prey poaching functions in synergy with livestock grazing and logging, it was the main factor driving a sharp decline of wild prey populations (i.e. Caspian red deer, roe deer, bezoar goat, wild boar) in the Hyrcanian forest. Additionally, in regard to the consequences of prey poaching, I concluded that large carnivores increase depredation on livestock (i.e. cattle and sheep/goat) when prey poaching is high, ultimately leading to severe human-carnivore conflicts in the region. This suggests foraging optimization as the depletion of wild prey forces predators to switch from wild prey to domestic animals in order to balance their energy intake. Such a shift in dietary behaviors of large predators predisposes them to retaliatory killings and persecution, thus threatening their long-term viability.

Conservation authorities should consider upgrading parts of protected areas and wildlife refuges retaining natural habitats to IUCN category II. PAs require priority actions in the assessment of grazing capacities and allocation and enforcement of grazing quotas. Moreover, better cross-sectoral coordination is needed among conservation authorities to avoid further depletion of the mammal community in the Hyrcanian forest, and to address sustainable livelihoods near PAs. The poachers' incentives need to be clearly understood and focused upon in follow-up studies. Mitigating livestock depredation requires a combination of strict law enforcement of anti-poaching measures, upgrading the status of reserves, and wild prey recovery plans. The persistence of protection-reliant species depends on their existence outside and inside PAs, ecological requirements and law enforcement measures. Thus, a holistic participatory approach is essential for sustainable wildlife conservation. Importantly, this increase in protection needs to be in line with providing viable alternative livelihoods to local communities.

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Declaration

Specific chapter contributions

I am the first author of the manuscripts included as Chapter 2, Chapter 3 and Chapter 5 of this dissertation. For these manuscripts, I developed study design, implemented fieldwork (data collection) and statistical analysis, and wrote most of the text of the manuscripts. For Chapter 4 and Chapter 6, I personally contributed to study design, fieldwork, data collection and statistical analysis, did a significant amount of writing and editing, and co-supervised a master student from University of Gottingen.

I, Mahmood Soofi, hereby declare that I am the author of this dissertation entitled 'Effects of anthropogenic pressure on large mammal species in the Hyrcanian forest, Iran'. All references and data sources that were used in this dissertation have been appropriately acknowledged. Furthermore, I declare that this work has not been submitted elsewhere in any form as part of another dissertation procedure.

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Mahmood Soofi