

LETTER

National Parks in Northern Sweden as Refuges for Illegal Killing of Large Carnivores

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Keywords

Brown bear; Eurasian lynx; management; poaching; survival; land sparing; wolverine.

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Received

5 October 2015

Accepted

18 December 2015

Editor

Reed Noss

doi: 10.1111/conl.12226

Abstract

Large protected areas are often considered to be as important as population size in reducing extinction risk for large carnivores. However, the effectiveness of protected areas for large carnivore survival has rarely been tested where surrounding areas also provide suitable habitats. Using individual-based long-term data, we here show that three species of large carnivores all suffered higher risk of illegal killing inside three large national parks than in surrounding unprotected areas in northern Sweden. We suggest that this illegal killing is the result of low enforcement and public presence in these remote parks, which results in a low probability for poachers to be discovered. Our results demonstrate that size of protected areas alone may be a poor predictor of their conservation value for large carnivores. We warn against passive national park management and advocate considering the ecological and socioeconomic context present inside as well as outside protected areas.

Introduction

Originally, national parks were created to preserve monuments and wonders of nature. Today, together with other forms of protected areas (PAs), they are one of the most important tools in biodiversity conservation (Margules & Pressey 2000; Hoffmann *et al.* 2010). The conservation success of PAs has spurred scientific effort into designing representative and persistent reserves. Still, areas with grand scenery and wilderness are often prioritized for protection, as they usually include remote and rugged areas, with little human use, and thus are of lower economic value (Pressey 1994; Soule & Sanjayan 1998; Margules & Pressey 2000). Nonetheless, large and remote PAs can have substantial conservation value as they often function as refuges for large carnivores, which

are especially vulnerable to human persecution, due to their high trophic levels, low population densities, and slow life histories (Purvis *et al.* 2000; Cardillo *et al.* 2004; Ripple *et al.* 2014).

Woodroffe & Ginsberg (1998) analyzed historical data and suggested that “critical reserve size” could predict local large carnivore population extinctions. This perspective has had a great influence and, consequently, the “land sparing” policy of securing large PAs and reducing negative edge effects has become a contemporary paradigm for conservation of large carnivores. In line with this perspective, many African parks are protected islands that increase large carnivore survival in otherwise human-influenced surroundings, but also experience substantial edge effects in terms of anthropogenic mortality (Loveridge *et al.* 2007; Kiffner *et al.* 2009; Balme

et al. 2010). In North America, gray wolf (*Canis lupus*) mortality risk was lower in the Greater Yellowstone Area and Central Idaho, relative to northwest Montana, likely due to larger core areas that offered stronger wolf protection (Smith *et al.* 2010). Johnson *et al.* (2004) also found higher mortality risk for grizzly bears (*Ursus arctos*) in areas with high road densities outside Yellowstone National Park. However, a long-term concern in conservation is that some PAs appear on the map, but have little conservation value (i.e., “paper parks”; Di Minin & Toivonen 2015). The major limitations to effective management of PAs arise from a lack of financial resources or deficiencies of management (Di Minin & Toivonen 2015), and such conditions are especially evident during times of governmental and social instability (e.g., civil wars; Plumptre *et al.* 2007). In the case of Wolong National Reserve for Giant Pandas, China, an unexpected negative effect followed the founding of the PA; habitat loss as the result of increased economic activity (Liu *et al.* 2001).

An alternative perspective is “land sharing” (Fischer *et al.* 2014), where large carnivores coexist with human activities in the landscape (Chapron *et al.* 2014). It is based on the fact that many large carnivore populations in North America and Europe have been generally stable or increasing throughout recent decades, despite high human population densities. This perspective advocates that large carnivore conservation is possible at high human densities when management is favorable and stable political institutions ensure proper law enforcement (Linnell *et al.* 2001). Sweden is a case in point; eradication programs exterminated or reduced large carnivore populations to very low numbers during the 1800s and 1900s (Haglund 1965; Swenson *et al.* 1994; Linnell *et al.* 2001). Today, after decades of favorable management policies, including protection in large national parks, the populations of brown bear (*Ursus arctos*), Eurasian lynx (*Lynx lynx*), and wolverine (*Gulo gulo*) have recovered and are now widely distributed in multiuse landscapes, often on privately owned land outside PAs (Chapron *et al.* 2014). In northern Sweden, the main large prey of large carnivores is semidomestic reindeer (*Rangifer tarandus*) (i.e., private property), which has created a conflict between large carnivore conservation and the indigenous Sámi reindeer husbandry (Swenson & Andrén 2005; Mattisson *et al.* 2011; Hobbs *et al.* 2012). To mitigate the economic impacts and ensure carnivore persistence, Sweden implemented a conservation performance payment system for large carnivores, combined with intensive population monitoring (Zabel & Holm-Muller 2008; Persson *et al.* 2015).

We have evaluated the protection from persecution provided to the three large mammalian carnivores (brown bears, Eurasian lynx, and wolverines) present in the largest national parks in northern Sweden (Sarek,

Stora Sjöfallet, and Padjelanta; Figure 1) by studying spatial variation in risk of illegal mortality inside these parks as well as in surrounding unprotected areas.

Materials and methods

Study area

Sarek and Stora Sjöfallet were among the first national parks created in Europe in 1909, chosen for their scenic and recreational value on public land of low commercial value, but also established to protect the then critically endangered Scandinavian brown bear population (Swenson *et al.* 1994). The area also contained most of the remnant Swedish wolverines during a bottleneck in the mid-1900s (Haglund 1965). Together with other PAs, these national parks and Padjelanta National Park form the 9,400 km² Laponia UNESCO World Heritage Site, one of the largest PA networks in Europe. The justification for World Heritage designation included its natural qualities (criteria vii, viii, and ix, UNESCO Ref:774) and the indigenous Sámi reindeer herding culture (criteria iii and v). The study area provides important spring-to-fall grazing areas and spring calving grounds for semidomestic reindeer both inside and outside the national parks. Snowmobile access during winter is not allowed for the general public inside the parks, but is allowed for reindeer herders and provincial rangers (Swedish Environmental Protection Agency, NFS 2013:10). The surrounding areas experience substantial recreational snowmobile activity. There is limited human infrastructure and agriculture both inside and outside the national parks (road density 0.017 km road/km² inside the parks vs. 0.15 outside, human settlement and agriculture, 0.0044% of the parks' area vs. 0.024% outside). Large game hunting is not allowed inside the parks.

The study area (48,536 km²) contained a vegetation gradient from Scots pine (*Pinus sylvestris*) and Norway spruce (*Picea abies*) forests starting at about 200 m a.s.l. via mountain birch (*Betula pubescens*) forests, heaths, and grass to mountain peaks and plateaus of bare rock and glaciers above 2,000 m. The climate was continental and the ground was usually snow covered from November until May.

Data

We used individual-based demographic and telemetry location data from radio-marked brown bears, Eurasian lynx, and wolverines from 1984 to 2010. In total, the data included 455 large carnivore individuals, monitored over 132,119 radio-days (Table S1). For details on capture and monitoring, see Supplementary Methods (online Supporting Information). Individuals were

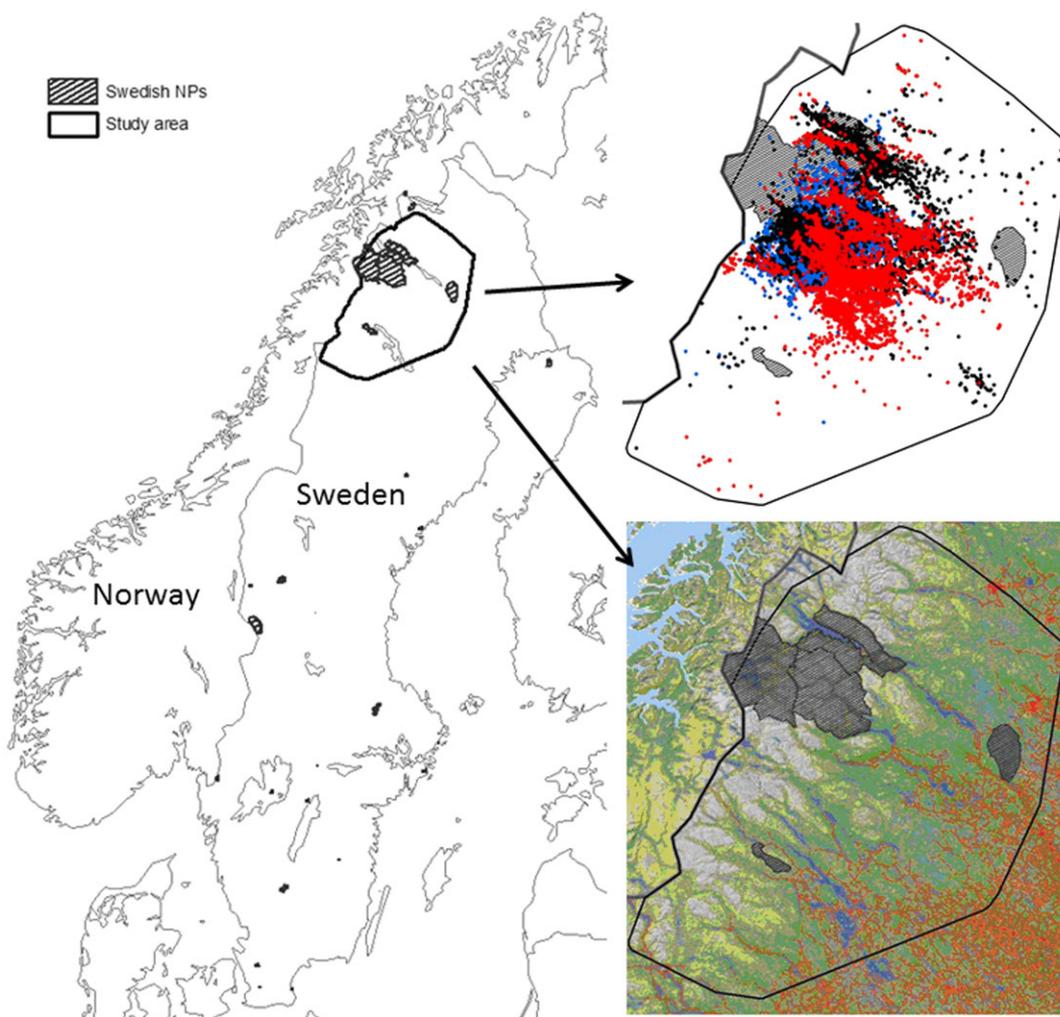


Figure 1 Location of the study area, including the national parks within it (gray shading), for brown bears, Eurasian lynx, and wolverines in northern Sweden, 1984–2010 (left); with animal locations (upper right; red = brown bears, black = lynx, blue = wolverines), vegetation and road distribution (lower right) in relation to the study area and national park borders.

classified by age group (juveniles, subadults, and adults) and sex. Cause of death was established, based on a list of predefined methods and criteria (Supplementary Methods; Andrén *et al.* 2006; Bischof *et al.* 2009; Persson *et al.* 2009), and the animals' fates were grouped into the classes "censored/alive," "illegally killed," or "other mortality" (Table S2).

Each telemetry location was assigned several measures of habitat features (vegetation class, forest, terrain steepness, and elevation) as well as human land use (national parks and reindeer calving grounds). Locations were also split by season (snow cover vs. no snow) and by telemetry method (very high frequency [VHF] or global positioning satellite [GPS]) (Table S3). For details on environmental data handling, see Supplementary Methods (online Supporting Information).

Risk modeling

To estimate the risk of mortality among individuals of the three species, we applied the Andersen–Gill model, a counting process formulation of the Cox proportional hazards model (Andersen & Gill 1982). The Andersen–Gill model allows for left (i.e., staggered entry) and right censoring of data (i.e., animals lost for follow-up or out-living the study), time-varying continuous and categorical variables, multiple events (i.e., fate of animals), and discontinuous time intervals of risk (Johnson *et al.* 2004; Murray 2006). We treated each time interval between sequential locations as a unique interval of risk, which was attributed different strata (i.e., sex and age of the individual, Table S3 b.) and environmental variables (i.e., forest and terrain steepness Table S3 a.) belonging to the end

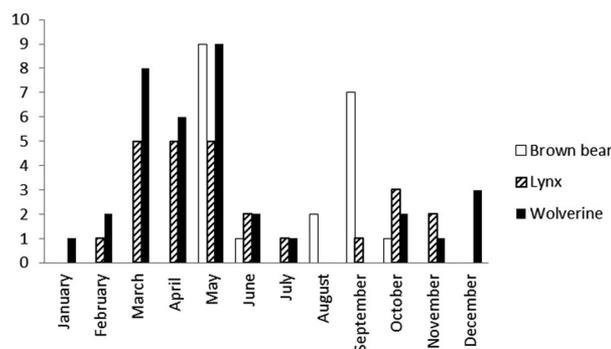


Figure 2 Monthly distribution of illegally killed radio-marked brown bears, Eurasian lynx, and wolverines in the study area in northern Sweden, 1984–2010.

point of the time interval (Johnson *et al.* 2004). Events were coded as 1 for mortality and 0 for right censoring. We excluded monitoring intervals > 30 days, considering these as discontinuous time intervals (Johnson *et al.* 2004). We added the additional number of days equal to 40% of the mean monitoring interval when right-censoring data (Johnson 1979). We built models in R library *Survival* (Therneau 2011) separately for each of the three species, and we built separate hazard models for “illegal killing” (risk of being killed illegally), “other mortality” (risk of mortality from other causes than illegal killing), and “total mortality” (for details on mortality cause verifications, see Supplementary Methods). We stratified all models by sex, age class, season, and telemetry method (VHF vs. GPS data, Table S3 b.), allowing for the calculation of separate baseline hazards for each stratum. Our analysis at three spatial scales (i.e., point value, 3,000 m buffer, and 6,000 m buffer, see Supplementary Methods) yielded comparable results (Table S5), thus we chose to keep the models with a buffer distance of 3,000 m in our final analyses. For the best models, we tested the model assumption of proportional hazards and plotted Schoenfeld residuals along with the smoothed hazard curves as well as Martingale residuals (Murray 2006). We assessed relative model fit by Akaike’s information criterion (AIC) and calculated model averaged parameter estimates and variable relative importance (VRI) among the models with $\Delta\text{AIC} < 4$ to the top models (Burnham & Anderson 2002).

Results

Illegal killing was an important source of mortality for all three large carnivore species (42–69% of total mortality, Table S2). We found an increased mortality risk for all species during the late snow season (Figure 2). This pattern was also reflected in the risk models, where the in-

Table 1 The effect of national parks in Andersen–Gill models of the risk of illegal mortality, the risk of mortality from other sources, and the total mortality among radio-marked brown bears, Eurasian lynx, and wolverines in northern Sweden, 1984–2010. Parameter estimates ($\beta \pm \text{SE}$), effect size ($\exp(\beta)$), and Akaike’s information criterion relative to the null models (ΔAIC)

a. Brown bear			
Mortality source	$\beta^* \pm \text{SE}$	$\exp(\beta)$	ΔAIC
Illegal	1.013 ± 0.456	2.75	–2.9
Other	$–0.458 \pm 0.467$	0.63	1.0
Total mortality	0.232 ± 0.307	1.26	1.4
b. Lynx			
Mortality source	$\beta \pm \text{SE}$	$\exp(\beta)$	ΔAIC
Illegal	0.925 ± 0.463	2.52	–2.3
Other	$–0.310 \pm 0.716$	0.73	1.8
Total mortality	0.549 ± 0.365	1.73	–0.3
c. Wolverine			
Mortality source	$\beta \pm \text{SE}$	$\exp(\beta)$	ΔAIC
Illegal	0.841 ± 0.364	2.32	–3.5
Other	$–0.084 \pm 0.310$	0.92	1.9
Total mortality	0.308 ± 0.233	1.36	0.3

*Positive coefficient indicates increased risk of mortality inside national parks.

clusion of season (snow vs. no snow) improved model fit substantially for all species (Table S4). Wolverines were especially vulnerable, with 3.2 times higher risk of mortality during the snow season. Brown bears varied somewhat from this general trend, as they also suffered illegal killing in September during the regular bear and moose (*Alces alces*) hunting season (Figure 2). Risk models stratified by sex also performed substantially better (Table S4), though the effect of sex differed substantially between species; wolverine females suffered 0.39 the risk of illegal mortality relative to males, whereas this difference was weaker in lynx (0.63) and brown bears (0.78).

The risk of illegal mortality was higher inside national parks for radio-collared individuals of all three species (i.e., positive parameter estimates; Table 1). The magnitude of this effect was consistent across all species, with 2.3–2.8 times higher risk inside the parks. However, when correcting for varying habitat composition inside and outside the parks (Table S6a), this effect was reduced for lynx where the lower amount of forest inside the parks contributed more to the higher risk than the park effect per se (*Forest*: VRI = 1.0, $\beta = -0.09 \pm 0.04$ [SE]). Brown bear risk also interacted with forest availability where the nonforested areas inside the parks represented high-risk areas (*NP*Forest*: VRI = 0.84, $\beta = -0.21 \pm 0.10$ [SE]). For wolverines, national parks as well as terrain steepness contributed the most to risk (steep: VRI = 1.0, $\beta = -0.15 \pm 0.07$ [SE]). The presence of reindeer calving grounds was included in several top ranking models. However, model averaged parameters of reindeer calving

grounds in all three species were associated with large error estimates, and no general conclusions could be drawn. Common proxies of human disturbance, such as distance to human infrastructure or permanent human activity, had no predicative power on mortality risk at local or regional scales in our study (Supplementary Methods).

The risk of mortality from other causes than illegal killing ("other mortality") was not affected by national parks (negative risk parameters, but with relatively large SEs; Table 1); thus, the risk of all sources of mortality ("total mortality") exhibited a weaker national parks effect (positive risk parameters, but with relatively large SEs; Table 1) than risk of illegal mortality alone. For wolverines, steep areas represented overall a lower risk of mortality, and for brown bear and wolverines, there was a tendency for reindeer calving grounds to represent higher overall risk (Table S6c).

Discussion

Our results clearly show that the risk of illegal mortality was higher inside the national parks than it was in the surrounding and unprotected areas for the animals in our study. We also found an increased mortality risk for all species during the late snow season, potentially due to increased human accessibility on snowmobiles during this period, because of increased day length and increased suitability of the snow for snowmobiling (Andr en *et al.* 2006; Persson *et al.* 2009). Habitat factors that impair snowmobile use, steep terrain and forest, are generally selected by large carnivores in the area (Rauset *et al.* 2013), and similarly were important factors reducing illegal mortality. Permanent human activity, represented by proximity to human infrastructure or agriculture, had no explanatory power for illegal mortality, further supporting our conclusion that large carnivores were killed mainly from snowmobiles capable of traversing throughout the parks.

We conclude that the national parks in our study had a negative effect on large carnivore conservation, contrary to the general conservation paradigm and the historical intention of the parks as a refuge for the brown bear (Swenson *et al.* 1994). That the national parks seemingly provided increased opportunities for illegally killing carnivores was, however, not an effect of the parks *per se*, but we suggest that it was rather a result of snowmobile restrictions for the general public. These restrictions were intended to limit human disturbance of wildlife and reindeer inside the parks. However, the combination of low public presence and low intensity of patrolling by provincial rangers in these large, remote areas also implied that poachers faced a low risk of detection. The

conservation value represented by these parks therefore has changed substantially since their establishment, not due to human development and habitat depletion (Liu *et al.* 2001), but due primarily to improved poacher mobility with modern snowmobiles. Thus, critical PA sizes built on historical data may be a poor predictor of future carnivore persistence. In fact, the network of national parks in our study is considerably larger than a suggested PA size for grizzly bears in North America (Woodroffe & Ginsberg 1998).

Efficient law enforcement is necessary for all natural resource management (Keane *et al.* 2008). However, law enforcement measures often lead to the criticism that PAs are created by and for an elitist few, in conflict with local community interests (Wittemyer *et al.* 2008). Therefore, measures promoting acceptance of large carnivores by local people often are of crucial importance for successful conservation programs (Treves & Karanth 2003). This is especially important in Lapland and similar areas having dual conservation goals, ensuring the conservation of native biodiversity and indigenous cultural heritage and livelihood. Such measures include incentives to compensate local costs of carnivores, such as direct compensation of losses or conservation performance payment systems, depredation prevention measures, controlled legal harvest, and by promoting activities giving carnivores a direct value (Treves & Karanth 2003; Dickman *et al.* 2011). The Swedish conservation performance payment system has been instrumental in the recovery of wolverines in Sweden, by providing protection for adult female wolverines through a combination of direct monetary value and indirect protection by monitoring denning activities (Persson *et al.* 2015). The payment system for lynx and brown bears is not linked to denning females; lynx are paid per family group, based on tracks observed in the winter following reproductive events, whereas payment for bears is based on occurrence, regardless of density and reproduction (Zabel & Holm-Muller 2008). The greater sex differences in risk of illegal mortality for wolverines than lynx or brown bears thus suggest that there is a link between payment system and risk of illegal mortality, with a lesser inclination to kill individuals that the payment system makes most "profitable." Nonetheless, our results show that rates of illegal killing of large carnivores are still high, especially when poachers face a low risk of detection. Hence, economic incentives may partly change poaching behavior, but are not a panacea (Treves & Bruskotter 2014). On the other hand, our results suggest that presence by the general public has the potential to reduce illegal killing of large carnivores. Poaching does not happen in a social vacuum, and the general public's tolerance to carnivores, and to illegal killing of such, strongly affect poaching behavior.

Our results emphasize the importance of critically evaluating the conservation performance of PAs, even large and remote national parks in a highly developed country, and considering the ecological and socioeconomic context present inside as well as outside them. We caution against passive management and high public expectations of positive effects of PAs on large carnivore conservation, based on the assumption that sparing land is enough.

Acknowledgments

We thank Jon Arnemo for veterinary assistance in the field, and numerous field workers for data collection. The study was funded by the Swedish Environmental Protection Agency, the Norwegian Environment Agency, the World Wide Fund for Nature (Sweden), the Swedish University of Agricultural Sciences, the Research Council of Norway, the European Association of Zoos and Aquaria (EAZA), and the Swedish Research Council for Environment, Agricultural Sciences and Spatial Planning (FORMAS). All handling of animals were approved by the Animal Ethics Committee for northern Sweden, Umeå. Data and models reported in the article are available from the corresponding author upon request. The authors declare no competing financial interests. Authorship: GRR, HA, JP, GS, and PS designed the project; all authors contributed to the collection and preparation of data; GRR analyzed the data and wrote the article with the help of JES, JP, HA, AZ, and GS.

Supporting Information

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

Table S1. Summary of data used for survival analyses of radio-marked brown bears, Eurasian lynx, and wolverines in northern Sweden, 1984–2010.

Table S2. Cause-specific mortality of radio-marked brown bears, Eurasian lynx, and wolverines in northern Sweden, 1984–2010.

Table S3. a) Environmental variables used and b) strata included and corrected for in the survival analysis of large carnivores in northern Sweden, 1984–2010. SMD vegcode refers to the Swedish Land Cover vegetation map from the National Land Survey of Sweden. "Definition" describes how the variables were estimated; "area" represents the area within a buffer of either 3,000 m or 6,000 m around a location, "distance" represents the shortest distance between a location and a specific habitat; "binary" represents whether a location is within or outside a specific habitat or whether a buffer contained the specific habitat or not.

Table S4. The effects of strata (refer to Table S3b.) on risk of being killed illegally for brown bears, Eurasian lynx, and wolverines in northern Sweden, 1984–2010. Parameter estimates in Andersen–Gill models are $(\beta \pm SE)$, and $\exp(\beta)$ represents the effect size, i.e., the difference in risk of group 1 relative to group 0 of Sex (males=0/females=1), Season (snow=0/no snow=1), Age01 (juvenile=0/subadult = 1, Age02 (juvenile=0/adult =1), Age12 (subadult=0/adult =1), VHF (GPS data=0/VHF=1). The ΔAIC represents the change in AIC when adding the stratum to the nonstratified model.

Table S5. The effect of national parks on the risk of being killed illegally for radio-marked a) brown bears, b) Eurasian lynx, and c) wolverines in northern Sweden, 1984–2010, at three different scales (animal locations and buffers of 3,000 m and 6,000 m around the locations). Parameter estimates $(\beta \pm SE)$, effect size $(\exp(\beta))$, and model AIC relative to the best models (ΔAIC) are for Andersen–Gill models.

Table S6. Environmental variables influencing a) risk of illegal mortality, b) risk of mortality other than illegal killing, and c) total risk of mortality of large carnivores in northern Sweden, 1984–2010. Parameter estimates (mean $\beta \pm SE$) and the Variable Relative Importance (VRI) are model-averaged parameters of Andersen–Gill models with $\Delta AIC < 4$ relative to the best models. The set of candidate models included all combinations of the environmental variables; "national park" (NP), "reindeer calving ground" (Calf), "steep terrain" (Steep), and "Forest" and their two-way interactions. NP and Calf are binary variables, whereas Steep and Forest represent the area (range 0 to 28.3 km²) within a 3,000-m buffer around the location (refer to Table S3). The effects of the strata (refer to Table S4) were corrected for and therefore always included.

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