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Habitat accessibility and snares impact large cats and their prey in Northeast Tiger and Leopard National Park, China

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ABSTRACT

Snares have become a widespread hunting technique, with their indiscriminate nature jeopardizing terrestrial large carnivores. Since the lack of reliable data on snare distribution hampers an accurate evaluation of the potential risk. Continuous winter SMART (the Spatial Monitoring and Reporting Tool) patrols have operated in the Amur tiger (*Panthera tigris altaica*) and Amur leopard (*Panthera pardus orientalis*) ranges in northeast China from 2013 to 2020 to record wildlife occurrences and remove snares. Our research aimed to assess the snares threat pose in the Amur tiger range and whether snare removal benefited roe deer (*Capreolus pygargus*) and wild boar (*Sus scrofa*) recovery. We modeled the spatial distribution of Amur tigers, Amur leopards, and prey species, comparing their distributions before and after anti-poaching measures. We also assessed the overlap between Amur tigers, Amur leopards, and snare locations. Additionally, we used the MaxEnt model to predict snare distribution, and then we analyzed the overlap of snares with species at different periods between 2013 and 2015 and 2018–2020.

Our results showed that the probability of occurrence of ungulates increased significantly around roads after snares were removed. Furthermore, we found that Amur tigers and Amur leopards distribution overlapped with snare locations suggesting that snares pose a serious risk to these non-target species. The overlap between snares and species has the same trend as the growing with species distribution, and the least significant increase is in DHZ-XNC.

Removing snares not only aids target game species but also protects sympatric large carnivores.

1. Introduction

As top predators, large carnivores mainly face man-made risk factors such as habitat loss and fragmentation, declining prey population, and hunting (Ceballos and Ehrlich, 2002; Morrison et al., 2007), with the latter a direct threat to their survival. Illegal hunting and trading pose the greatest threat to the conservation of rare species, which are the focal point of global conservation efforts. As perhaps the world's most iconic and revered wild species, the number of wild tigers has significantly reduced from 100,000 a century ago to as low as 4485 in 2022 (Goodrich et al., 2022; Sanderson et al., 2006; Morell, 2007), putting them at risk of functional extinction. Recognizing the urgency of the situation, tiger conservation has become a global concern. In November 2010, 13 tiger-range governments and conservation partners adopted the Global tiger Program in St. Petersburg, Russia, making a bold commitment to double the global tiger populations to over 6000 individuals by 2022 (Global Tiger Recovery Program (GTRP), 2010).

Amur tigers (*P. tigris altaica*) and Amur leopards (*P. pardus orientalis*) are primarily distributed in the Far East of Russia and the northeast of China (Jinzhe et al., 2021; Guangshun et al., 2017), which are highly threatened by habitat loss, population fragmentation, poaching, and declining prey populations (Hebblewhite et al., 2014; Kerley et al., 2002). Over the past century, rapid population growth and extensive road development in China have led to substantial habitat declines for

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these subspecies (Wang et al., 2017; Wang et al., 2018). To safeguard their populations, strict anti-poaching measures for Amur tigers and Amur leopards have been in place in China since the 1950s (Meng et al., 1995). In recent decades, the Chinese government has further intensified protection efforts through national initiatives, including the Natural Forests Protection Project and the establishment of the Northeast Tiger and Leopard National Park (NTLNP), which now houses >50 Amur tigers and 60 Amur leopards (National Park Administration, 2022). Despite being as apex carnivores, Amur tigers and Amur leopards are rarely vulnerable to lethal pressure from other wildlife, as in many other Asian locations, they are at risk of capture in snares set for other species, which poses the greatest threat to their survival (Robinson et al., 2015; Goodrich et al., 2008; Aziz et al., 2013).

Since the implementation of the guns ban in 1996, snares have become one of the most commonly used methods of poaching in China. Snares in China's tiger range are typically set to capture ungulates, either for consumption or as a control strategy to mitigate crop destruction caused by animals such as wild boar (*S. scrofa*). Snares used in northeast China are usually constructed with a 3-mm wire or cable snares. Poachers attach them to trees and set open nooses above the ground along animal trails. This method is widely employed and inexpensive (approximately $2 US). Moreover, snares can be easily deployed in large numbers and are inconspicuous compared to other hunting techniques such as gun hunting, hunting with dogs, or electric shock hunting (Gray et al., 2018; Harrison et al., 2016). Consequently, they carry a lower risk of detection by law enforcement agencies (Afriyie et al., 2021). These characteristics make snaring one of the most widespread poaching methods in Africa and Asia (Noss, 2000; Rasphone et al., 2019; WWF, 2019), especially given the increased crackdown on poaching activities (MacMillan and Nguyen, 2014). Another reason for the popularity of snares is their longevity in the field, as they can remain active for up to two years after being set and can capture a wide variety of species (Noss, 1998). In Northeast China, snares are typically set during winter to facilitate the storage of bushmeat and enable easier tracking of animal movements. Unfortunately, poachers seldom remove their snares, posing an ongoing threat to wildlife and leading to cumulative effects. The impacts of snaring can persist for years after initial deployment (Fryxell et al., 1991), directly and long-term affecting the survival, growth, and reproduction of wild species, which can result in more severe consequences compared to other hunting methods (Gray et al., 2017).

Despite increased interest in addressing the threat of snares, accurately and reliably assessing the impact of snares on wild animals remains a significant challenge for anti-poaching initiatives. Furthermore, the distribution of snares may be associated with various factors including habitat type, geography, agricultural history, and even the personal background of hunters (Wato et al., 2006; Fizgibbon et al., 1995; Kümpel et al., 2009), which makes it challenging to reliably estimate snaring preferences (O'Kellya et al., 2018) and thus hard to systematically sample. Moreover, the number of snares does not often reflect capture rates. For example, Muchaal and Ngandjui (1999) found that while snare density was lowest farthest from human settlements, the capture rate was higher at these locations. Therefore, how to assess the harm caused by snares to wild animals has become a critical limitation in wildlife conservation and anti-poaching management efforts.

In order to combat poaching, the Wildlife Conservation Society (WCS) initiated the Management Information System (MIST) in the Hunchun Nature Reserve, China in 2008. Then, in 2013, the Spatial Monitoring and Reporting Tool (SMART) patrol system was introduced in the Hunchun Forestry Bureau and later expanded to the Wangqing Nature Reserve, China.

In the case of the Jilin Wangqing Nature Reserve, China, a study conducted in 2010 based on daily patrol records revealed a density of 1.6 snares per 10 km (Peng, 2013). Since 2013, this reserve has been utilizing SMART patrols to investigate wildlife occurrences and remove snares. This provides an opportunity to evaluate the impact of law enforcement efforts by comparing data from before and after the implementation of anti-poaching management measures, which can provide valuable insights for local and regional anti-poaching management strategies.

In this study, we utilized the dataset of SMART patrols conducted between 2013 and 2020 in the Jilin Wangqing National Nature Reserve to address two objectives. Firstly, we tested whether the removal of snares benefitted the recovery of roe deer (*C. pygargus*), and wild boar, which are dominant and most abundant prey for both predators in the region. Secondly, we examined how the presence of snares in the field posed a threat to the survival of Amur tigers and Amur leopards.

2. Materials and methods

2.1. Study area

The field work was carried out within the 670 km^2 zone of the Jilin Wangqing National Nature Reserve, located in the TLNP region of northeast China (coordinates: E 129°56′-131°04′, N 43°05′-43°40′). The area experiences an average annual temperature of 1.5 °C, with extreme maximum temperature reaching 35.6 °C and extreme minimum temperature dropping to −36 °C. Winter typically begins in mid-November and lasts until mid-March, during which the study area is covered with snow, enabling effective tracking of both human activities and wildlife.

We chose three forest farms (FF) as study pilots: Lanjia FF (LJ), Duhuangzi-Xinancha FF (DHZ-XNC), and Dahuanggou FF (DHG). These forest farms represent the minimum unit engaged in forest management (Fig. 1). The study area was identified as one of the First Priority Area for Amur tiger protection in China, as specified by Li et al. (2010a). This area is considered essential habitat for Amur tigers and Amur leopards, serving as a migration corridor for breeding populations to the interior (Quanhua et al., 2015; Qi et al., 2015).

The ungulate community in the study area comprises six species, with roe deer and wild boar being dominant species that serve as important food sources for Amur tigers and Amur leopards. Other sympatric prey species, such as red deer (*Cervus elaphus*), sika deer (*Cervus nippon*), musk deer (*Moschus moschiferus*), and Chinese goral (*Naemorhedus caudatus*) are rare, occurring at relative low densities and being limited to few small habitat patches (Miquelle et al., 1996).

The effects of humans and human-related disturbances on apex predators include the impacts of roads, settlements, farmlands, logging, poaching, grazing and quarrying. (Bhattarai and Kindlmann, 2013; Kerley et al., 2002; Barber-Meyer et al., 2013). In the case of the Wangqing Reserve, which serves as one of the core habitats of Amur tigers and Amur leopards in Northeast China, strict restrictions are in place to mitigate grazing activities. Among the three study pilots, only DHZ-XNC includes one village and engages in limited agricultural activities, but farmlands are only located around the village. For the other two sites, only the forestry stations serve the forestry workers and reserve rangers throughout the year. Thus, the primary human activities in the forested areas involve forest management, collection of forest products, and some nature tourism. Hunting is strictly prohibited, and some poaching mainly using snares was carried out before the reserve was established in 2010.

2.2. Data collection

At the beginning of 2010, we conducted comprehensive training for 342 rangers and 13 forest bureau or nature reserve managers from Amur tiger and Amur leopard ranges in China. The training covered techniques for data collection, information-based patrolling, data management, and snare removal. The participants from Wangqing, Suiyang and Dongning were provided with Global Positioning System (GPS) units, field notebooks, pens, and CyberTracker (version 4.4.0) for recording and transmitting patrolling data on human activities and wildlife occurrences. Jilin Wangqing National Nature Reserve began SMART patrol

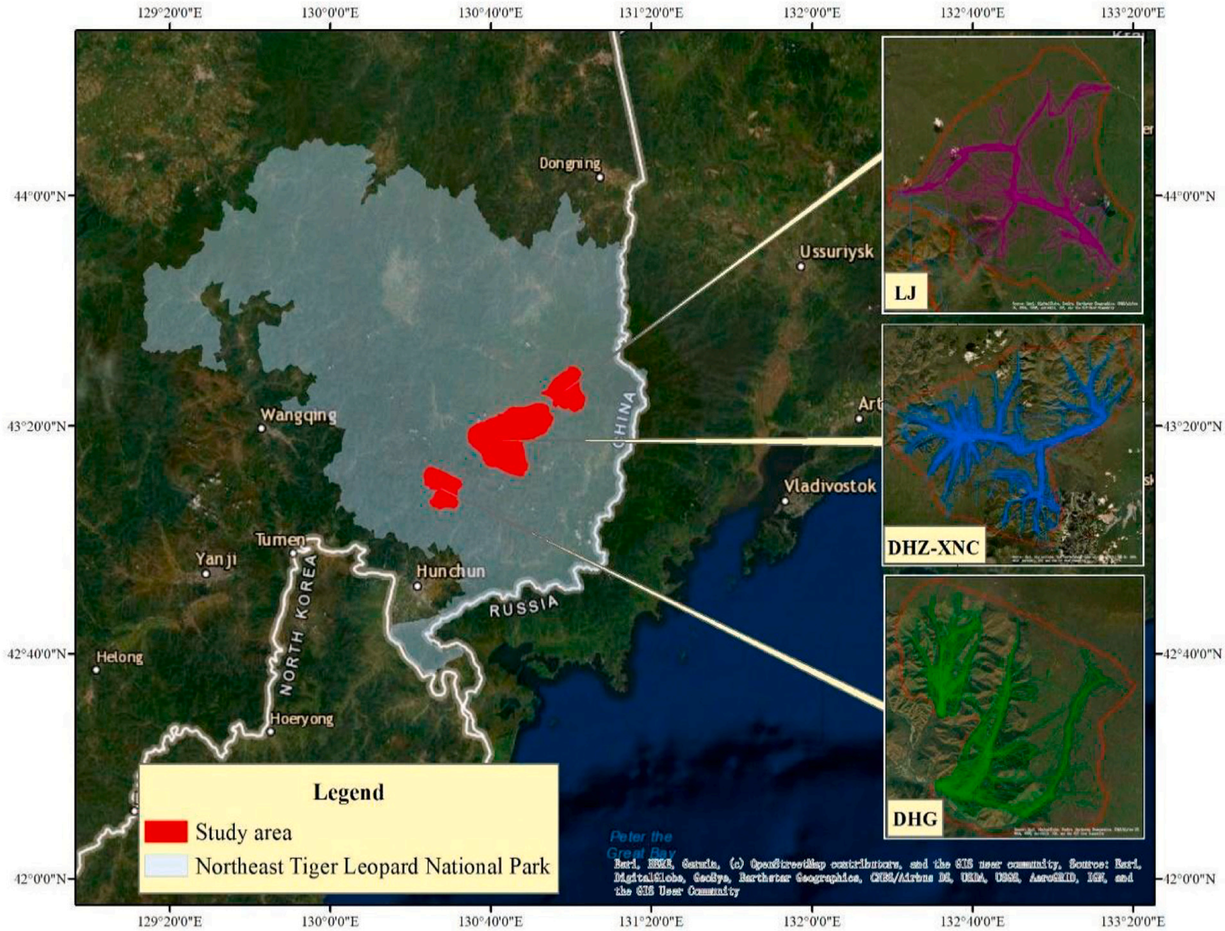


Fig. 1. Map of the study area in Jilin Wangqing Reserve, Northeast Tiger and Leopard National Park, China, during 2013–2020, showing the survey routes and intensity with green, blue and purple colors in Dahuanggou forest farm (DHG), Duhuangzi-Xinancha forest farm (DHZ-XNC) and LanJia forest farm (LJ). (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

throughout its jurisdiction in 2013, and it has continued until now. Rangers were responsible for documenting all instances of human disturbances and wildlife sightings they encountered during their patrols. This information included the category or species of the encountered wildlife, the number of individuals, and geographic coordinates. Additionally, rangers were instructed to remove any snares they encountered. The individu- al numbers recorded in each encounter were used to estimate the average group size of wildlife. As track identification and detection could be challenging without a snowpack, we utilized only winter data (November to March) for modeling the spatial distribution of target species. Furthermore, due to the freshness of the snow tracks, we recorded information within a 24-h timeframe. By considering the number, direction, and freshness of the footprint chain, we were able to estimate the individual numbers of animals within a group (Zhang et al., 2013).

For this study, we utilized the SMART (version 5.0.2) patrol records from 2013 to 2020 in the three pilots for further analysis. Ranger-based detection of snares was imperfect as patrol coverage focused along roads and search efforts were uneven. We could assess the age and condition of the snares in the field, and old snares were described as tightly embedded in the tree, showing signs of rust and corrosion, losing elasticity. These records helped us determine the timeframe of poaching incidents. Our data showed that the majority of recorded snares throughout the survey period were categorized as “old” (>2 years), suggesting a very low probability of changes in the snare distribution pattern. Furthermore, it is worth noting that the total number of snares found in 2020 was minimal (as shown in Fig. 1). Considering these factors, we concluded that all the snare records from 2013 to 2020 could be treated as a reliable snare distribution layer for this study. (See Fig. 2.)

2.3. Environment variables

In Northeast China, Amur tigers and Amur leopards occupy different habitats due to variation in altitude distribution (Li et al., 2019). Amur leopards prefer high-altitude ridges (Carroll and Miquelle, 2006), while Amur tigers favor low-altitude areas (Wang et al., 2018). Therefore, we didn't expect Amur tiger to be an influence on Amur leopard presence. Based on previous studies of Amur tigers and ungulates resource selection (Hebblewhite et al., 2014; Jiang et al., 2014), we identified 11 variables that have been found to influence wildlife habitat. These variables include geographic factors such as slope, aspect, location, geographic topography, and river, as well as biotic factors such as dominant tree species, average diameter at breast height (DBH). Additionally, we considered anthropogenic factors such as main roads, secondary roads, villages, and human density (Supplementary data 1). To analyze the road network, we categorized it into two levels: 1) the main roads that connect villages or settlements, and 2) secondary roads that extend from a village or residential area to a forest patch. All roads in the study area are unpaved, but they are managed differently during winter. The snow on main roads is cleared for motor vehicles, tourism and non-timber forest products production, while secondary roads tend to be covered with thick snow.

We obtained the layers of vegetation-referenced variables, including

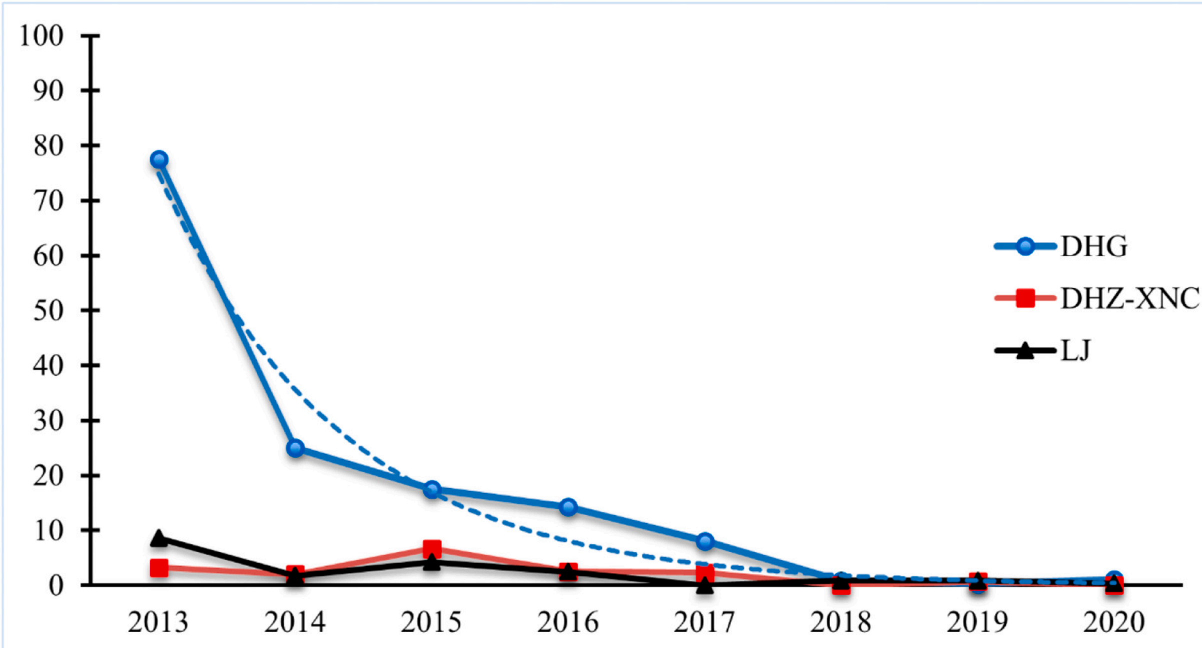


Fig. 2. Encounter rates of snares from 2013 to 2020 per 100 km in Dahuanggou Forest Farm (DHG), Duhuangzi-Xinancha Forest Farm (DHZ-XNC), and LanJia Forest Farm (LJ), showing a rapid decline trend: $y = 157.48e^{-0.743x}$, $R^2 = 0.9621$.

dominant species and DBH, as well as other layers such as rivers, roads, and villages, from the local forestry departments. DEM (Digital Elevation Model), geographic position, and human density layers were downloaded freely from the Resource and Environment Science and Data Center of China.[2] We then used the spatial analysis tool of ArcGIS10.2 (ESRI, 2013) to convert the DEM layer into slope, aspect, and location layers, respectively. All variable layers were standardized to 30 m × 30 m spatial resolution for further analysis. This standardization ensures consistency and compatibility among the different layers in terms of their spatial representation.

2.4. Species distribution models

We conducted the spatial distribution analysis of snares using the MaxEnt[3] software, which is a machine learning algorithm based on the principle of maximum entropy methods MaxEnt is commonly used to predict species distribution using presence-only data and environmental variables (Schadt et al., 2002). Prior to model construction, we examined the collinearity among the variables and excluded those with a pairwise correlation >0.5 to avoid redundancy in subsequent analyses. We also removed variables that had a total contribution rate of <10 % in estimating the occurrence probabilities of the target species. This refinement process aimed to improve the model's accuracy and precision.

The snare occurrence data were randomly divided into two subsets: a training subset (75 % of sampling data) and a test subset (remaining 25 % of data) for model construction and model validation. We used the area under the receiver operating characteristics curve (AUC) of test data to evaluate the predictive performance of models. AUC values between 0.7 and 0.9 were considered useful, while values above 0.9 indicated high accuracy in model predictions (Hanley and Mcneil, 1982).

We used the Jackknife tests to evaluate the weight of each environmental factor in determining the distribution of species and snares. This analysis helps in understanding the relative contribution of different variables in the model's predictions. Previous studies have demonstrated the effectiveness of using MaxEnt model in assessing environmental parameters and predicting suitable habitats for Amur tigers (Miquelle et al., 2014; Graham et al., 2004). Therefore, we adopted this method to investigate the influence of environmental factors on the spatial distribution of snares. To ensure robustness and reliability, we conducted 20 replicate runs of the model. This approach involved repeated subsampling and cross-validation, which is commonly used in ecological modeling (Boria et al., 2014; Pearce and Boyce, 2006). By randomly splitting the occurrence data into folds, we obtained ROC curves with error bars and calculated the average AUC (area under the receiver operating characteristic curve) across the models. Additionally, we generated summary response curves with one standard deviation error bar to illustrate the model's predicted response to different environmental variables.

Acknowledging the potential sampling bias caused by the overlap of patrol routes with roads, we took measures to address this issue and ensure the reliability and accuracy of our research results. Previous studies have shown that roadside sampling covers wide environmental gradients, along dirt or gravel roads can still provide valuable and unbiased information for building species distribution models (Mccarthy et al., 2012; Kadmon et al., 2004). To correct the sampling biases, we implemented a systematic sampling method, which has consistently performed well across different conditions (Fourcade et al., 2014; Stockwell and Peterson, 2002). We created a grid with a defined size of 30 m × 30 m and randomly selected one occurrence per grid cell. This approach helped reduce the spatial aggregation of records and accounted for the observation distance of ranger patrols. We also converted the patrol survey effort into survey density (m/m^2) per grid cell to create a bias layer. This allowed us to correct for the sampling deviation and account for variations in patrol effort across different areas. We employed the density analysis function of the spatial analyst tools in ArcGIS10.2 to calculate the survey density per grid cell. By implementing these methods, we aimed to minimize the potential biases introduced by the patrol routes and ensure a more representative and unbiased sampling approach for our analysis. Habitat suitability models

[2] https://www.resdc.cn (accessed 5 Feb 2022).
[3] https://biodiversityinformatics.amnh.org/open_source/maxent (accessed 9 Jan 2023).

may be imperfect due to a range of factors, including sampling related issues, but that we made every effort to incorporate as much information as was available to make them as useful as possible.

To analyze the habitat changes of roe deer and wild boar, we obtained spatial distribution data during two periods of 2013–2015 and 2018–2020, representing conditions before and after the implementation of anti-poaching measures and law enforcement. Due to the limited sample size for Amur tigers in Dahuanggou (DHG) and Amur leopards in Lanjia (LJ), we integrated all the occurrences data from 2013 to 2020 for an overall layer. This approach enabled us to analyze the habitat preferences and distribution patterns of Amur tigers and Amur leopards across the entire study, providing insights into their responses to snare occurrence and other environmental factors.

2.5. The potential influences of snares on target species

To estimate the potential influences of snares on the target species, we applied the non-parametric method of relevant samples to test if there are significant differences in occurrence probabilities changes of roe deer and wild boar, as well as occurrence probabilities of Amur tigers and Amur leopards between the entire study area and areas with snares. This analysis allowed us to examine the potential effects of snare presence on the distribution and occurrence of these species. Additionally, we calculated the spatial overlap between the snare distribution and the predicted distribution maps of target species using the raster calculation tool in ArcGIS 10.2. To evaluate the spatial changes in occurrence probabilities of roe deer and wild boar, we compared the distribution maps for the two periods : 2013–2015 and 2018–2020. By subtracting the distribution maps of these two periods, we estimated the changes in occurrence probabilities and identified areas where significant alterations occurred. These analyses allowed us to assess the potential impacts of snares on the target species and understand the spatial changes in their occurrence probabilities over time.

3. Results

3.1. Description of basic information on survey intensity and data collection

From 2013 to 2020, a total of 12,900 km of patrols were completed in an area of 638 km^2 (Fig. 3C; Supplementary data 2b). In total, 1456 snares were recorded and removed, with 1423 discovered before 2018, and only 5 recorded in 2020. Additionally, during 2013–2020, a total of 3642 roe deer and 1091 wild boar occurrences were recorded. Although DHZ-XNC had the largest patrol mileage, most records of roe deer and wild boar information were in LJ. Based on the rate of encounter (records/100 km), the species activity of roe deer and wild boar showed considerable fluctuation during the survey period (Fig. 3A). However, there is no increasing trend shown in any pilot. The average group sizes of roe deer and wild boar showed similar trends with occurrence numbers, but they are larger in DHG with the most records of Amur tigers and LJ with the most records of Amur leopards than in DHZ-XNC, where tigers and leopards were not recorded frequently (Fig. 3B; Supplementary data 2a). At the same time, a total of 60 tiger information points were found by experienced rangers, most of whom were formerly hunters transferred and trained to distinguish wildlife tracks and collect samples by Feline research center. These points include 55 records in DHG, 2 in LJ, and 3 in DHZ-XNC. We also found 190 occurrences of Amur leopard, mainly concentrated in LJ with 177, 3 in DHG, and 10 in DHZ-XNC. Footprint identification techniques (FIT) and an informative monitoring network verified the investigation (Alibhai et al., 2023; Gu et al., 2014).

3.2. Spatial distribution of snares

The MaxEnt models for all three areas (DHG, LJ, and DHZ-XNC) exhibited a good fit, with mean AUC values for 20 replicates exceeding 0.7. The models indicated that the highest occurrence probability of snares was found in DHG, with an average value of 0.45 (±0.24). In LJ, the occurrence probability was lower, with an average value of 0.18 (±0.21), while in DHZ-XNC, it was slightly higher at 0.31 (±0.25).

The spatial distribution of snares differed significantly within the three pilots, as shown in Fig. 4. In DHG, the distribution of snares was strongly influenced by environmental factors such as rivers, aspect, and DBH, which collectively contributed 56.8 % to the model. Among the human disturbance factors, the village had the highest contribution rate exceeding 10 %. In DHZ-XNC, the distribution of snares was primarily influenced by the presence of secondary roads, which had a contribution rate of 18.2 %. Other important factors with contribution rates exceeding 10 % were geographic factors such as location, slope, and geographic topography. In LJ, human disturbance factors played a significant role in the distribution of snares, although LJ had the lowest level of human disturbance compared to the other two areas. The village and main road collectively contributed 65.7 %, while other factors had contribution rates below 10 %. These findings highlight the variation in snare distribution patterns across the study areas and the different factors influencing their occurrence. Understanding these patterns and factors is crucial for designing effective management strategies to mitigate the impact of snares on wildlife populations in each specific area.

3.3. Spatial distribution of the target species

We selected seven variables (e.g., sroad, mroad, river, village, slope, human density and location) in the modeling of roe deer and wild boar, and six variables (e.g., sroad, river, location, mroad, human density, and village) for Amur tigers and Amur leopards (e.g., sroad, village, mroad, domitree, human density, and slope) models, respectively, after correlation analysis. AUC values of all models were above 0.8 (indicating satisfactory fits) except for three models, which had AUC values of 0.77, 0.76, and 0.67 (Supplementary data 4). Our spatial distribution models showed that roads had a significant impact on all target species in winter (Fig. 5; Table 3). However, the responses of roe deer and wild boar varied between models. Most models showed a negative correlation between the probability of ungulate occurrence and distance to road, indicating a preference for areas in proximity to roads, the supplementary feeding carried out by locals during extreme weather may affect the road preference of ungulates. Compared to roe deer models, more U-shaped curves appeared in the wild boar models (Supplementary data 5; Supplementary data 6). For example, the relationship between the occurrence probability (y) of wild boar and the distance to the main road (x) was U-shaped in both periods in LJ, suggesting that highly suitable habitats are distributed in the main road and areas over 3 km away from the road. Additionally, the secondary roads were more frequently selected in the wild boar models. For roe deer, the 2013–2015 model in DHG showed significant avoidance of both the main road and the secondary road. Furthermore, the overall impact of human factors, including roads and other human disturbances, on the spatial distribution of roe deer and wild boar decreased during 2018–2020 compared to 2013–2015 (Fig. 5A, B).

The distribution of Amur leopards in LJ was predominantly influenced by human disturbance (82.7 %), whereas in DHG, the distribution of Amur tiger was relatively more influenced by environmental factors, despite the importance of secondary roads (Table 1; Supplementary data 7).

3.4. The potential influences of snares to target species

The spatial overlap between snares and target species was generally small across all species and sub-study area (Table 2). While, for roe deer and wild boar, there was a noticeable increase in overlap values between

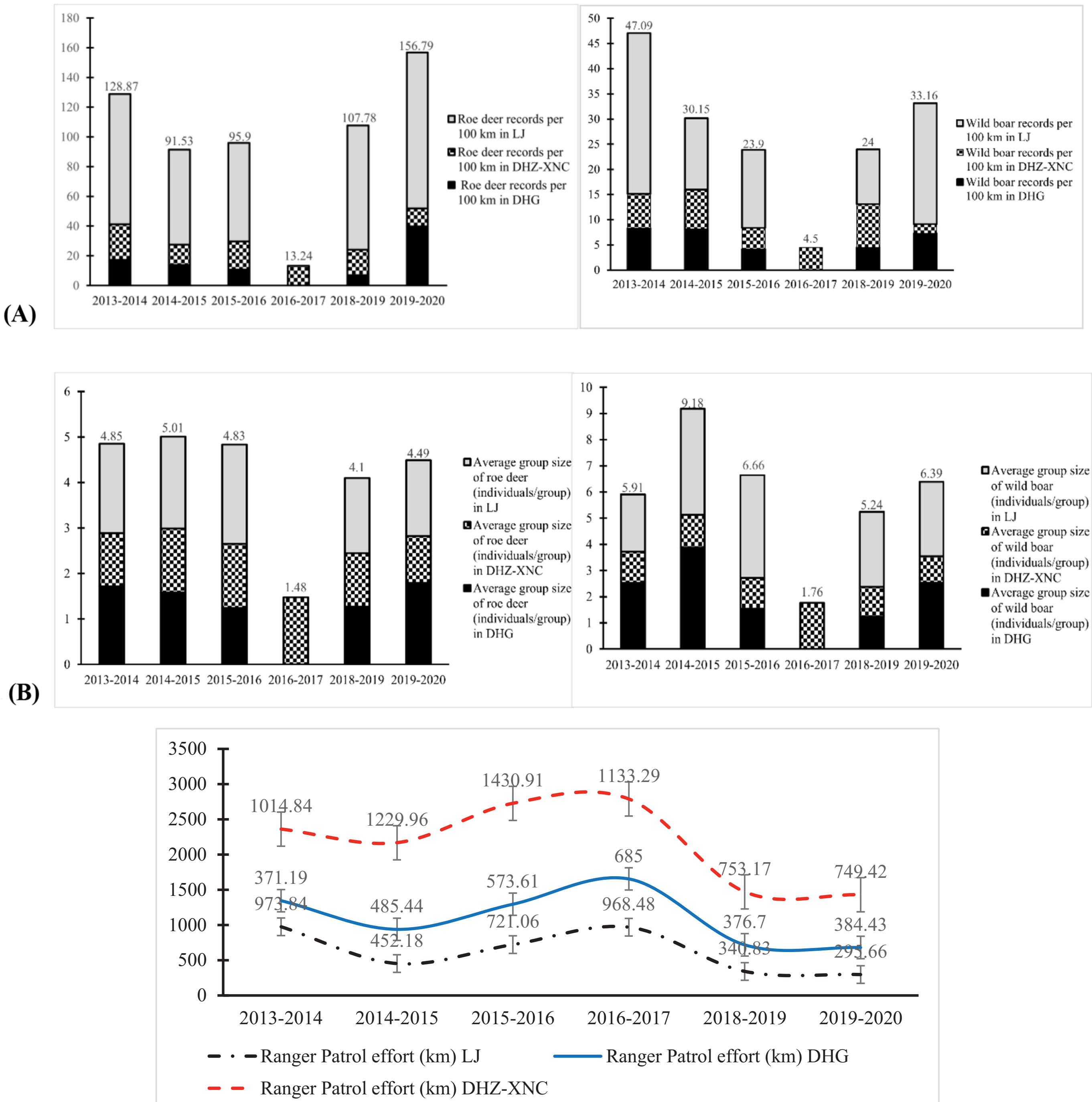


Fig. 3. **(A)** Records of roe deer and wild boar from 2013 to 2020 per 100 km in Dahuanggou Forest Farm (DHG), Duhuangzi-Xinancha Forest Farm (DHZ-XNC), and LanJia Forest Farm (LJ). The encounter rate of roe deer or wild boar per 100 km generally showed a U-shaped trend.
Fig. 3 (B) Average group size of roe deer and wild boar from 2013 to 2020 per 100 km in Dahuanggou Forest Farm (DHG), Duhuangzi-Xinancha Forest Farm (DHZ-XNC), and LanJia Forest Farm (LJ). The average group size of roe deer compared between 2013 and 2015 and 2018–2020 was significant ($p < 0.0001$).
Fig. 3 (C) Ranger patrol effort from 2013 to 2020 in Dahuanggou Forest Farm (DHG), Duhuangzi-Xinancha Forest Farm (DHZ-XNC), and LanJia Forest Farm (LJ).

the two periods (2013–2015 and 2018–2020) in all three areas.

Comparing the two survey periods (2013–2015 and 2018–2020) showed a general increase in habitat accessibility of roe deer and wild boar in both DHG and LJ, with an average occurrence probability increase of no <0.14 (Fig. 6; Table 3). Notably, most of the increases occurred in areas around secondary roads away from settlements and main roads. However, there was not much improvement in the average occurrence probability of the two species in DHZ-XNC.

Additionally, we observed that in both LJ and DHG, the proportions of changes in habitat suitability were very similar between the snare distribution areas and the entire survey area for roe deer and wild boar (Supplementary data 3). However, in DHZ-XNC, there were significant differences in the changes of habitat suitability for roe deer and wild boar between the snare distribution area and the whole survey area (Supplementary data 3).

We also discovered the average occurrence probability of Amur tigers in areas with snare distribution was 0.48, which is significantly higher than the overall occurrence probability of 0.19 throughout DHG

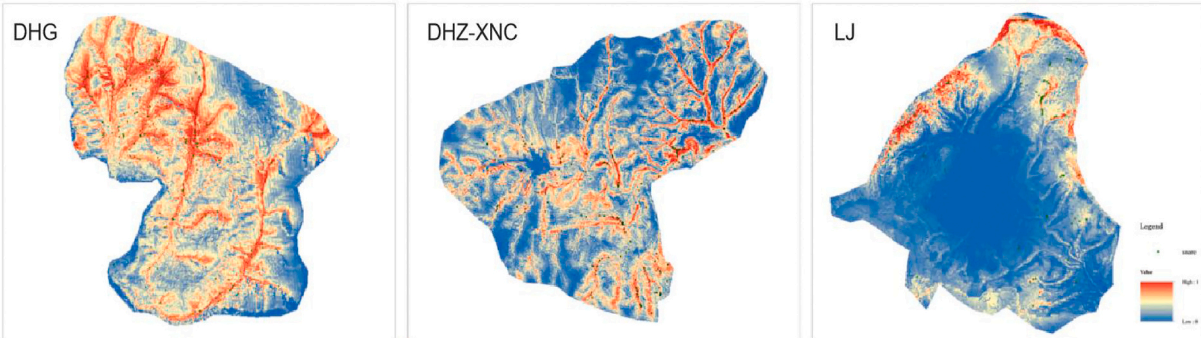


Fig. 4. Spatial distribution of snares in Dahuanggou Forest Farm (DHG), Duhuangzi-Xinancha Forest Farm (DHZ-XNC), and LanJia Forest Farm (LJ). The color scheme indicates the probability of occurrence, with blue representing a low probability and red indicating a high probability. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

(Table 3). In contrast, the Amur leopard in LJ appears to be less affected by snares, as the overlap between the distribution of snares and the hotspots of leopard occurrences was less significant (Fig. 7). Furthermore, the average occurrence probability of Amur leopards in areas with snare distribution (0.136) was slightly lower than that of the entire LJ (0.194) (Table 3). These findings suggest that the influence of snares on the distribution of Amur leopards in LJ is relatively minimal compared to Amur tigers in DHG.

4. Discussion

4.1. Spatial distribution of two big cats and prey

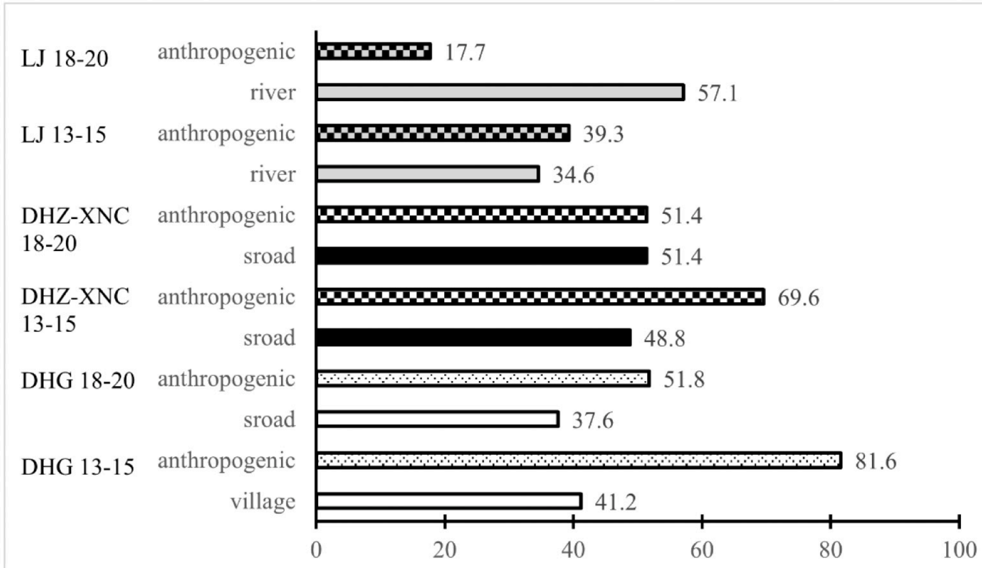
Amur tigers and Amur leopards co-occur in northeast China and Russia. Studies have shown that Amur leopard, as the subordinate competitors, tend to avoid Amur tigers both in terms of space and time interactions to reduce the risk of potentially fatal encounters. While Amur tigers generally thrive with high prey density of medium to large wild ungulates. Their diet predominantly consists of red deer, wild boar, sika deer and roe deer. The spatial distribution of Amur tigers and Amur leopards is influenced by factors such as prey species richness and patterns of human disturbance. The impact of roads on wildlife is complex and can have a double-edged effect. Roads are considered a significant disturbance for wildlife (Miquelle et al., 1999), leading to decreased survivorship and reproductive success of carnivores (Noss et al., 1996), also provide easier access for poachers and result in disturbance from tourism and non-timber products collection. Also, roads can serve as efficient movement pathways, acting as travel corridors for Amur tigers (Kerley et al., 2002), were more likely to be detected in locations closer to forest roads, which often present a higher density of shrubs along their sides benefit ungulates (Li et al., 2019). Studies have indicated that wild boar exhibit a preference for areas near agricultural land and roads (Zhao et al., 2019). For example, a previous study showed that the number of wolverines (*Gulo gulo*) is 15 times higher within 1–3 km of roads compared to areas outside this range, as roads facilitate their movement (Hornocker and Hash, 1981). Additionally, the road in the Changbai Mountains has a grouping effect on yellow mongooses (*Mustela sibirica*), which occurs within 50 m on both sides of the road due to the presence of food sources (Wang et al., 2010). However, roads can also alter species interactions and cause avoidance in some species. For instance, predators like wolves often use roads in winter when local prey density is low (Kittle et al., 2017), increasing the risk of ungulates using roads (Kittle et al., 2008). This avoidance is also observed in other species such as elk and wild boar (Prokopenko et al., 2017; Theuerkauf and Rouys, 2008). One of the most effective strategies for the conservation of Amur tigers, Amur leopards and their prey is road closures (Slaght et al., 2019).

Our results indicate that the relationship between the occurrences probability of roe deer or wild boar and road distance generally exhibits declining trends or U-shaped curves (Supplementary data 5; Supplementary data 6), indicating the important role of roads in shaping the spatial distribution of roe deer and wild boar in winter throughout the study area. Roads located at the foot of mountains provides shelter for wildlife against cold winter winds. Additionally, reserve managers provide supplementary feed for ungulates near the main roads during extreme weather (Felton et al., 2016; Lambert and Demarais, 2001; Terada et al., 2010). The thick snow cover in the forests of northeast China during winter increases the energy cost for ungulates in terms of food foraging and spatial movement (Moen and Boomer, 2005; Tyler, 1991; Harris et al., 2014), roads offer an attractive option for these animals to find food and move around, likely explaining why roe deer and wild boar to choose habitats near roads in winter. The U-shaped curve indicates that while animals utilize the road itself, they tend to avoid the immediate habitat adjacent to the road.

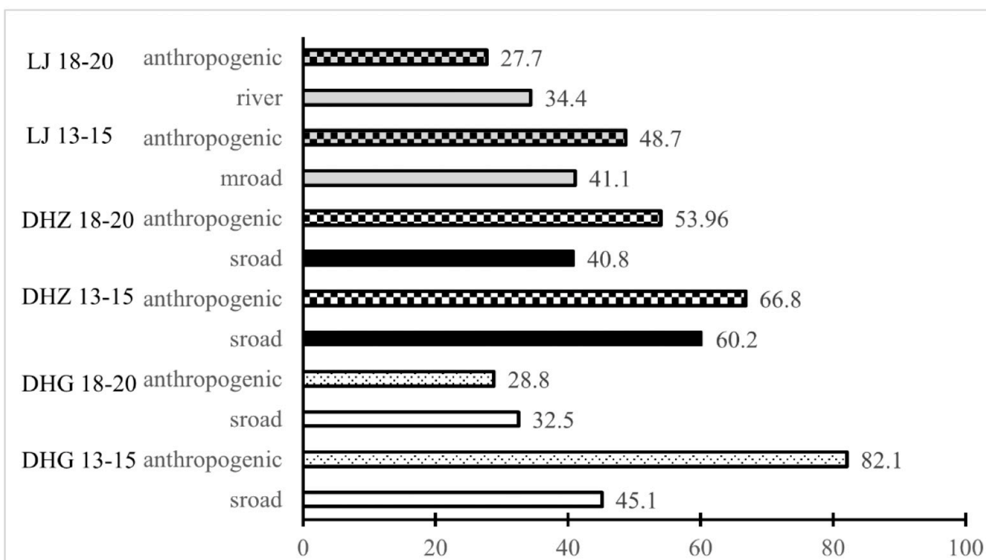
Our study findings indicate that roe deer and wild boar respond differently to the presence of roads, highlighting the complexity of the impact of human disturbance on wildlife (Fig. 6; Supplementary data 6; Supplementary data 7). We classified roads into main roads and secondary roads based on their connectivity to residential areas and snow clearance. Our results revealed that secondary roads can function as ecological corridors for wildlife, providing access to forested areas with lower levels of human disturbance. These roads are not actively cleared of snow, making them suitable habitats for ungulates. On the other hand, main roads, which are regularly cleared of snow, facilitate easier movement for ungulates but also introduce higher levels of human activities, particularly vehicles, causing greater uneasiness. The uncertain reactive behavior of roe deer and wild boar to main roads may be attributed to a dynamic balance between utilizing road corridors and avoiding human activities.

In this study, we considered the combined effects of roads, villages, and hunting snares on the habitat suitability of roe deer and wild boar. Our results demonstrated that LJ and DHG had higher habitat accessibility than DHZ-XNC (Fig. 6). This is due to the absence of villages and the majority of habitat improvements occurring in areas far away from villages and main roads. The consistent variation trend in occurrence probability between the snare distribution area and the whole study area (Table 3) indicates that national conservation policies and projects, such as the ban of commercial forest logging since 2015 in Jilin province,[4] have contributed to the increase in ungulate food and habitat suitability. However, in DHZ-XNC, where larger residential areas and more human disturbances were present, the total average occurrences of roe deer and wild boar changed only minimally. The uncertainty of occurrence probability fluctuations and the variation trend difference between the

[4] http://www.ecns.cn/2015/03-27/159643.shtml (accessed 8 Jan 2023).



(A)



(B)

Fig. 5. **(A)** The influence on roe deer distribution, measured as a percentage of the total, of the top and anthropogenic variables for each study area (LJ, DHZ-XNC and DHG) during each period (2013–2015 and 2018–2020).
Fig. (B) The influence on wild boar distribution, measured as a percentage of the total, of the top and anthropogenic variables for each study area (LJ, DHZ-XNC and DHG) during each period (2013–2015 and 2018–2020). The trend of anthropogenic variables contribution is decreasing compared 2013–2015 to 2018–2020.

Table 1
The variable contribution of Amur tiger and leopard models in DHG and LJ.

Study area	Species	Variable	Percent contribution
DHG	Tiger	Sroad	49.3
		River	31.4
		Location	10.4
LJ	Leopard	Sroad	52.1
		Village	17.8
		Mroad	12.8

snare distribution area and the whole study area were more pronounced for both species (Table 3), indicating that indirect human disturbance can have a greater impact on prey than the direct threat of snares. The village size affects the intensity of road use, human accessibility to forests, villages, and farmlands. Additionally, this human activity can lead to habitat fragmentation and fear effect, resulting in complex responses in the temporal-spatial distribution and population dynamics of wildlife.

We also observed clustering behavior among roe deer and wild boar in regions with a higher occurrence of Amur tigers or leopards. This clustering behavior can be explained as a response to the increased risk of predation, as ungulates often exhibit group behaviors and increase their group size as a defensive strategy against predation (Manor and Saltz, 2003; Lima and Dill, 1990). While larger groups reduce individual predation risk, they also increase the probability of multiple individuals being captured by snares in the same location, which may limit the predators' survival.

Table 2
Average spatial overlap of snares and target species.

		Roe deer		wild boar		Amur tiger	Amur leopard
		2013–2015	2018–2020	2013–2015	2018–2020		
Snare	DHG	0.08 ± 0.14	0.16 ± 0.20	0.10 ± 0.15	0.20 ± 0.19	0.11 ± 0.18	
	DHZ-XNC	0.09 ± 0.16	0.09 ± 0.17	0.07 ± 0.15	0.09 ± 0.16		
	LJ	0.02 ± 0.05	0.05 ± 0.09	0.01 ± 0.04	0.04 ± 0.07		0.02 ± 0.05

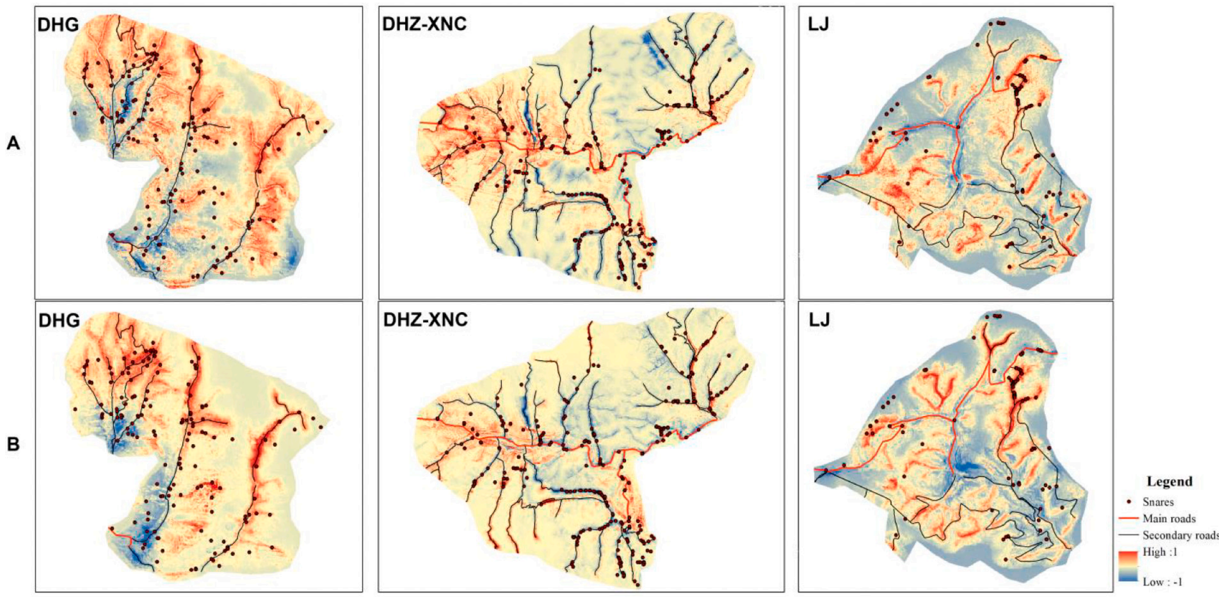


Fig. 6. Habitat changes between period of 2013–2015 and 2018–2020 for wild boar (A) and roe deer (B) in DHG, DHZ-XNC, and LJ. Red color represents an increase in occurrence probability increase, and bule color represents a decrease. Snares are shown as black dots.
Note: The changes in occurrence probability for roe deer and wild boar have clearly expanded in DHG, DHZ-XNC, but there has not been much improvement in LJ. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

Table 3
Main parameters of difference test basing on a Non-parametric test for occurrence probabilities of Amur tigers and Amur leopards (or change trend in occurrence probability of roe deer, wild boar) in snare distribution areas and that in entire subareas.

Study area	Species	Entire subareas (Average ± SD)	Snare distribution areas (Average ± SD)	p
DHG	Roe	0.1351 ± 0.2400	0.2232 ± 0.3396	0.289
DHG	Wild boar	0.1895 ± 0.2229	0.1020 ± 0.2667	<0.001
DHG	Tiger	0.1894 ± 0.2369	0.4827 ± 0.3352	<0.001
DHZ-XNC	Roe	−0.0033 ± 0.1442	−0.0242 ± 0.2413	0.019
DHZ-XNC	Wild boar	0.0573 ± 0.1886	0.0413 ± 0.2783	0.222
LJ	Roe	0.1848 ± 0.1690	0.3098 ± 0.2733	<0.001
LJ	Wild boar	0.1939 ± 0.1818	0.2766 ± 0.2349	<0.001
LJ	Leopard	0.1942 ± 0.2500	0.137 ± 0.0242	<0.001

4.2. *Potential threats of snares to Amur tiger and leopard*

Our study also revealed distinct spatial distribution patterns of the Amur tiger and Amur leopard. Amur tigers in DHG were found to prefer areas of lower elevation near rivers, which provide water sources and potential prey populations (Jhala et al., 2011; Valeix et al., 2010), and their response to human disturbances was limited. The U-shaped response curve for secondary roads, which was the only significant contributor to human disturbance in the model, suggests that tigers use these roads while avoiding habitats around them. Amur tigers exhibit avoidance behaviors towards human disturbances, which is an adaptation to minimize interactions with humans (Goodrich et al., 2011). In contrast, Amur leopards in LJ showed a greater susceptibility to human disturbances. They preferred secondary roads and residential areas while avoiding the main roads. The residential area in LJ only serves the forestry stations of forestry workers and reserve rangers, and we speculate that leopards may be lured by livestock or domestic dogs (Henschel and Ray, 2003), given that roe deer and wild boar also showed a certain preference for residential areas in this study.

These different spatial distribution patterns result in significantly different degrees of threat to Amur tigers and Amur leopards. The hotspots of Amur tiger occurrences in DHG largely overlap with the snare distribution area, while Amur leopards are better able to avoid snares concentrated areas. The average habitat suitability of the Amur tiger in snare distribution areas is also much higher than that throughout the DHG pilot (Fig. 7; Table 3), suggesting that snaring poses a bigger threat to tigers than leopards. This viewpoint aligns with local records, as five tigers were reported to have fallen victim to snares in China between 1998 and 2015 (GuangShun et al., 2020), while no such incidents were reported for leopards. It is crucial to note that our study does not imply the threat of snares to leopards could be ignored. As the Amur leopard is much rarer than the Amur tiger (Wang et al., 2016), any deaths resulting from snares would be especially lamentable.

4.3. *Implications for conservation*

As wild animal populations continue to recover in the habitats of tigers and leopards in northeast China, human-animal conflicts have increased, particularly due to crop damage caused by ungulates such as

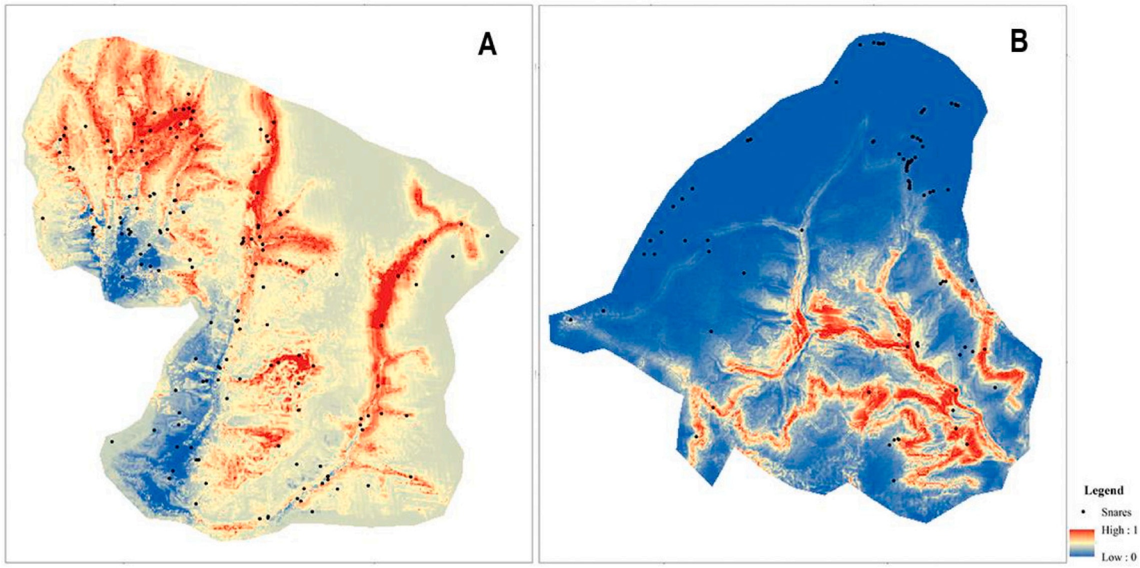


Fig. 7. Habitat suitability assessment results of Amur tigers in DHG (A) and Amur leopards in LJ (B). Snares are shown as black dots.

wild boar (Nyhus and Tilson, 2004; Schley et al., 2008; Li et al., 2010b). While anti-poaching measures have significantly reduced the poaching of tigers and leopards, the survival of Amur tigers remains threatened by snares set by local residents to protect their crops or express grievances against wildlife. To mitigate human-animal conflicts, forest managers should prioritize enhancing public awareness about conservation and addressing the underlying motivations for poaching, including commercial gain, household consumption and disagreement with wildlife crime (Muth and Bowe, 1998). Furthermore, it is important to recognize that hunting activities not targeted at endangered species can still result in immediate and fatal injuries to such species. The impact of untargeted hunts on endangered species varies depending on their ecological characteristics. Therefore, effective anti-poaching measures can be implemented to meet the demands of tigers, also there should be an ongoing campaign to ban snaring, based on patrolling information to carry out the law enforcement work. With the transition of the TLNP from a pilot phase to formal establishment, during the gap periods, there is a need for constant attention and support to ensure the continuity of conservation efforts. This includes enforcing the closure of secondary roads and other access routes, and promoting community engagement in wildlife conservation. Whether snaring occurs depends on the level of active enforcement of no-hunting regulations, and the level of compliance with such regulations is high. To ensure these aspects are effectively implemented, a protection management team should be formed, which strictly restricts human access to the reserve and carries out daily law enforcement and anti-poaching work. The SMART database has indicated a drastic decline in patrol intensity and data recording between 2017 and 2018. Therefore, ensuring sustained operation and adequate financial support are also necessary for effective conservation.

5. Conclusions

Snaring is widespread used to hunt a variety of wildlife and is often set along the border areas of cross management, including nature reserve boundaries. Both targeted and untargeted species are randomly caught, leading to mortality, serious injuries, stress responses and disabilities, jeopardizing the long-term survival of wildlife. Our study aimed to explore the impact of hunting snares on wildlife recovery and threats by combining data from the SMART (anti-poaching management tool) database of hunting snares with variables derived from MaxEnt models to access the habitat accessibility of Amur tigers, Amur leopards, and their prey after the removal of snares. Our finding suggest that Amur tiger are more susceptible to snares than Amur leopards and that human disturbance and roads are significant factors influencing wildlife occurrence.

As the tiger population and ecological environment recover, conflicts between humans and animals are likely to become more common in China. To mitigate these conflicts, an early warning system and community-based anti-poaching programs could be implemented to anticipate and prevent them (Critchlow et al., 2015). This study demonstrates that conservation organizations and government agencies can employ these methodologies to model wildlife habitats and threats using data from ranger patrols. With advancements in spatial statistics and machine learning, predictive models can be constantly enhanced by incorporating additional data from patrols and human disturbances, and these models can be extended for application in other protected areas.

Author contributions

Minghai Zhang and Qi Li designed the study. Jinzhe Qi and Jianyu Peng analyzed the GIS and SMART data for modeling. Qi Li wrote the manuscript. Li Qu, Jinzhe Qi, Peng Jianyu and Qing Xu conduct the field training. Qi Li, Jinzhe Qi, Qing Xu, Christine Wenzel and Minghai Zhang revised the manuscript. Qi Li, Jianyu Peng, Li Qu and Qing Xu contributed to data collection. Christine Wenzel contributed to discussion, input, and assistance during the conduct of the project.

CRediT authorship contribution statement

Qi Li: Conceptualization, Data curation, Formal analysis, Investigation, Methodology, Project administration, Visualization, Writing – original draft, Writing – review & editing. **Jinzhe Qi:** Formal analysis, Methodology, Resources, Software, Supervision, Validation. **Jianyu Peng:** Conceptualization, Data curation, Funding acquisition, Investigation, Methodology, Project administration, Software. **Li Qu:** Conceptualization, Data curation, Resources, Software. **Qing Xu:** Investigation, Software, Supervision, Writing – review & editing. **Christine Wenzel:** Resources, Supervision, Writing – original draft, Writing – review & editing. **Minghai Zhang:** Conceptualization, Funding acquisition, Investigation, Methodology, Resources, Supervision, Writing – review & editing.

Declaration of competing interest

Qing Xu and Christine Wenzel are employees of the startup company S-3 Research LLC. S-3 Research is a startup funded and currently supported by the National Institutes of Health, National Institute of Drug Abuse through a Small Business Innovation and Research contract for opioid-related social media research and technology commercialization. The authors declare that they have no competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data will be made available on request.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.biocon.2023.110414.

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