MEDIUM AND LARGE MAMMAL COMMUNITY AND SPATIOTEMPORAL ACTIVITY AND INTERSPECIFIC INTERACTIONS OF THE BROWN BEAR (*Ursus arctos*) IN YENİCE FOREST - IMPLICATIONS FOR CONSERVATION

by

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Submitted to the Institute of Environmental Sciences in fulfillment of

the requirements for the degree of

Doctor of Philosophy

in

Environmental Sciences

Boğaziçi University

2024

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DATE OF APPROVAL: 22/03/2024

To beloved Gazi Yolseven, who saved so many starfish

ACKNOWLEDGEMENTS

I extend my deepest gratitude to all those who supported me throughout this journey. Beginning with my supervisor, Raşit Bilgin, whose constant support and guidance from the very beginning to the end made this dissertation a reality. His constant belief in me and support during challenging times meant the world. I also want to thank my co-supervisor, Christos Astaras, who introduced me to the methods I used in this dissertation when I was studying at WildCRU, University of Oxford, and guided me throughout this dissertation. His support and encouragement were immeasurable. I would like to acknowledge the contributions of my committee members, Andrzej Furman, Anıl Soyumert Ertürk, Çağatay Tavşanoğlu, and Ulaş Tezel, which greatly enriched this work.

The camera traps used in this study were supplied by the Ministry of Forestry and Agriculture. I want to especially thank to Ahmet Işık, Ali Bozkurt, İsmail Sevimler, Saime Demirtaş, and Bayram Doğdu from Karabük Directorate of Nature Protection and National Parks, as well as my dear friends and colleagues, Nehir Sevim and Utku Ergün, for their invaluable assistance during fieldwork. Nazmi Kaya was my initial companion during the early fieldwork stages. It saddens me deeply that he is no longer with us to witness the completion of this dissertation.

I want to express my heartfelt thanks to the Bilgin lab members, as well as my friends Esra Özdemir and Hakan Can, who were always there for me when I needed it most. Lastly, I am incredibly thankful to my parents Nazmi and Fikriye Mergenci from whom I inherited the love and curiosity for animals and wildlife, and who fully encouraged me on this journey. The initial inspiration for studying biology was planted by my late grandfather Faruk Mergenci in his apple garden when I was a child.

ABSTRACT

MEDIUM AND LARGE MAMMAL COMMUNITY AND SPATIOTEMPORAL ACTIVITY AND INTERSPECIFIC INTERACTIONS OF THE BROWN BEAR (*Ursus arctos*) IN YENİCE FOREST - IMPLICATIONS FOR CONSERVATION

Understanding their ecological requirements is essential for the conservation planning of the protected species in a designated protected area. Otherwise, protected areas become symbolic reserves safeguarded only in documentation. In this study, we carried out the first extensive systematic camera trapping in Yenice Forest, Karabük, Turkey, covering around 500 km^2 area of the forest. We examined the distribution and habitat use of medium and large mammals in Yenice Wildlife Development Area and the surrounding state hunting grounds. We further evaluated the spatiotemporal patterns and species interactions of the brown bear which is a keystone, umbrella, and flagship species. A multiscale analysis of environmental covariates showed that mean elevation and slope within 16km radius and water bodies within 4km radius were the most influential parameters and positively influenced the occurrence of brown bears in Yenice Forest. The human-related covariates did not affect the spatial use of habitat, however, there was temporal partitioning of space use between humans and brown bears. At the community level, abundance was found higher in the areas with vehicle captures compared to the areas where no vehicles were recorded. Mean slope within 16km radius and water bodies within 4km radius were also important covariates at the community level. Protected area status and distance to the protected area did not change the habitat use of the target species. Target species showed mostly nocturnal and crepuscular activity. Further systematic monitoring of these species would help understand the trends of these populations in time and better inform conservation decisions.

ÖZET

YENİCE ORMANLARINDAKİ ORTA VE BÜYÜK MEMELİ KOMÜNİTESİ VE BOZAYININ (*Ursus arctos*) ZAMANSAL VE MEKANSAL AKTİVİTESİ VE DİĞER TÜRLERLE ETKİLEŞİMİ - KORUMA İÇİN ÇIKARIMLAR

Koruma alanlarında korunan türlerin koruma planlaması için ekolojik gereksinimlerinin anlaşılması elzemdir. Aksi takdirde, korunan alanlar sadece belge üzerinde korunan sembolik rezervler haline gelir. Bu çalışmada, Karabük, Türkiye'deki Yenice Ormanı'nda, ormanın yaklaşık 500 km^2'lik alanını kapsayan ilk kapsamlı sistematik fotokapan çalışmasını gerçekleştirdik. Yenice Yaban Hayatı Geliştirme Sahası ve çevresindeki avlaklarda orta ve büyük memelilerin dağılımını ve habitat kullanımını inceledik. Ayrıca, kilit taşı, şemsiye ve bayrak tür olan boz ayının mekânsal-zamansal örüntülerini ve diğer türlerle etkileşimlerini değerlendirdik. Çevresel değişkenlerin çok ölçekli bir analizi, 16 km yarıçapta ortalama yükseklik ve eğimin ve 4 km yarıçapta su kaynaklarının en etkili parametreler olduğunu ve Yenice Ormanı'ndaki bozayı varlığını olumlu yönde etkilediğini göstermiştir. İnsanla ilişkili değişkenlerin habitatın mekânsal kullanımı üzerinde bir etkisi olmamıştır, ancak insanlar ve bozayılar arasında alan kullanımının zamansal olarak bölünmesi söz konusudur. Komünite düzeyinde bolluk, araç tespit edilen alanlarda tespit edilmeyen alanlara kıyasla daha yüksek olmuştur. 16 km yarıçapta ortalama eğim ve 4 km yarıçapta su kaynakları komünite düzeyinde de etkili olmuştur. Korunan alan statüsü ve korunan alana olan mesafe, hedef türlerin alan kullanımında etkili bulunmamıştır. Hedef türler çoğunlukla gececil ve krepüsküler aktivite göstermiştir. Bu türlerin sistematik olarak daha uzun süre izlenmesi, bu popülasyonların zaman içindeki eğilimlerinin anlaşılmasına ve koruma kararlarının daha iyi bilgilendirilmesine yardımcı olacaktır.

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1. DIVERSITY OF MEDIUM AND LARGE MAMMALS OF YENICE FOREST BASED ON CAMERA TRAPS

1.1. Introduction

1.1.1. Yenice Forest

Yenice Forest is one of the largest intact forests and a biodiversity hotspot that is home for mammal species such as the grey wolf (*Canis lupus*), brown bear (*Ursus arctos*), golden jackal (*Canis aureus*), red fox (*Vulpes vulpes*), European wildcat (*Felis silvestris*), roe deer (*Capreolus capreolus*), red deer (*Cervus elaphus*), wild boar (*Sus scrofa*), badger (*Meles meles*), and weasel (*Mustela nivalis*). The small mammals identified in Yenice forest include yellow-necked mouse *(Apodemus flavicollis),* wood mouse *(Apodemus sylvaticus),* edible dormouse *(Glis glis),* bank vole *(Myodes glareolus),* European pine marten *(Martes martes)*, European hare *(Lepus europaeus),* least weasel *(Mustela nivalis),* and Anatolian squirrel *(Sciurus anomalus)* (Ünal, 2012; Can, 2008).

Yenice Forest is a part of the Temperate Broadleaf and Mixed Forests Biome, and is in the western segment of Euxin province that belongs to the Euro-Siberian phyto-region (Davis et al., 1988). It is in a transition zone of climatic conditions where the Black Sea climate meets the steppe climate (Öner, & Abay, 2005). Therefore, besides plant species that are typical to the Euxin province, it also contains plant species that belong to the Mediterranean and Iran-Turonian phyto-regions (Ünal, 2012). The most common tree species in the forest are fir (*Abies* sp.), beech (*Fagus* sp.), pine (*Pinus* sp.), and oak (*Quercus* sp.) (Öner, & Abay, 2005). It has two relict forests of Istranca oak (*Quercus hartwissiana*) and common yew (*Taxus baccata*) (Aksoy, 1985) that are under protection. Turkish pine (*Pinus brutia*), European hornbeam (*Carpinus betulus*), maple (*Acer* sp.), European ash (*Fraxinus excelsior*), European bladdernut (*Staphyllea pinnata), g*oat willow *(Salix caprea),* Oriental plane *(Platanus orientalis),* European box *(Buxus sempervirens),* common alder (*Alnus glutinosa*), hazel (*Corylus* sp.*),* and European hop-hornbeam *(Ostrya carpinifolia)* are among other species that form the forest vegetation (Türkiş, & Elmas 2018; Arslan, & Ketenoğlu, 2020; Pulatoğlu, & Güney 2022; Uyar, Alataş, Ören, & Keçeli, 2007). The pseudomacchie is mostly composed of Greek strawberry tree (*Arbutus andrachne*), Gallipoli rose (*Cistus*

salviifolius), tree heath (*Erica arborea*), prickly juniper (*Juniperus oxycedrus*), false olive tree (*Phillyrea latifolia*), and turpentine tree (*Pistacia terebinthus*) (Uyar et al., 2007).

In YWLDA, north-facing areas are under the influence of the humid climate of the Black Sea and are characterized by a maritime climate. Beech is the dominant tree species in these areas. South-facing areas are under the influence of a drier climate and oak species are dominant in these areas. Pseudomacchie components are distributed in lower elevations and do not cover extensive areas. They can be observed in the areas where the broad-leaved and coniferous forest is deteriorated, at valley entrances, and near the creeks (Pulatoğlu, & Güney, 2022). The forest is also categorized as an Important Bird Area (IBA006) (Kılıç, & Eken, 2004), an Important Plant Area (IPA) (Özhatay, Byfield, & Atay, 2005), and an Important Nature Area (INA) (Eken, Bozdoğan, İsfendiyaroğlu, Kılıç, & Lise, 2006). It also covers natural habitats that are listed under Bern Convention Endangered Natural Habitats (41.1 El - Western Pontic beech forests, 41. HU - Western Euxinian mixed forests, 42.172 - Bornmueller's fir forests, 42.5FI 1 - Western subpontic Scots pine forests) (Özhatay, Byfield, & Atay, 2005).

However, all these titles and being a home for so many different and important species did not culminate in protection priority for this area. About 26.000 hectares of this forest which ranges over 75.000 hectares, has been designated as a wildlife Development area (WLDA). The WLDA status[1] does not provide actual protection status, the way that a "wildlife conservation area[2]" or "nature conservation area[3]" does. In WLDAs, we cannot speak of absolute protection where hunting is totally forbidden. Instead, they are lands used to rear game animals, and hunting is allowed within private hunting schemes. Kavaklı (4km^2) and Çitdere (8km^2) Nature Protection Areas of Yenice Forest comprise less than 2% of the entire forest. In addition to this, Karabük-Büyükdüz Research Forest (23 km^2) and Yaylacık Research Forest (50 km^2) are the areas dedicated for research.

[1] **Wildlife Enhancement Area:** According to Land Hunting Law No. 4915; the areas where game and wild animals and wildlife are protected, game animals are introduced, measures are taken to improve their habitats, and where hunting is allowed when necessary within the framework of special hunting plans.

[2] **Wildlife Conservation Area:** According to Land Hunting Law No. 4915; the areas that exhibit 'wildlife value', where habitats need to be protected are absolutely protected and maintained, along with plant and animal species.

[3] **Nature Conservation Area:** Rare, endangered or nearly extinct ecosystems, species, and places that contain distinct examples that result from natural events, which are important in terms of science and education, and need absolute protection, and are reserved for science and educational purposes.

Yenice Wildlife Development Area (YWLDA) is surrounded by Pürenkaya State Hunting Ground, Göktepe State Hunting Ground, Şimşirdere State Hunting Ground, Keltepe State Hunting Ground, Ulupınar State Hunting Ground, and Pazarköy State Hunting Ground, and there are no measures to keep the hunters outside of this area. In addition to hunting, logging also poses a threat to protected species in this forest. Since it does not have an 'absolute' protection status, logging is allowed within the YWLDA boundaries. Loggers are usually villagers coming from the villages around the area, and they are allowed to build temporary housings called "sayvan", and they live within the forest throughout logging seasons, which almost span the whole year. Although it is illegal within the area, many loggers carry rifles, arguing that they need the rifles to protect themselves from wild animals. During our study visits, although they had been observed with their hunting dogs and rifles, some loggers claimed that they were not hunting.

Logging activities can disturb the animals also because of the high frequency of site visits, noise of heavy machinery and trucks, destruction or invasion of denning sites, and cutting down of fruit trees that animals use as food sources. Intensive forestry activities can lead to a decline in animal populations (Buřivalová, Lee, Giam, Şekercioğlu, Wilcove, & Koh, 2015; Laurance & Laurance, 1996), changes in predator-prey dynamics (Houle, Fortin, Dussault, Courtois, & Ouellet, 2010), disturbance of animals during their nesting and denning periods (Laurance, & Laurance, 1996; Buřivalová et al., 2015), changes in their movement and activity patterns (Houle, et al., 2010; McLellan, & Shackleton, 1988; Martin, Basille Van Moorter, Kindberg, Allainé, & Swenson, 2010), force them to leave their territories (Laurance, & Laurance, 1996; Bearer, Linderman, Huang, An, He, & Liu, 2008), and ultimately decrease their chances of survival.

Animal husbandry, green housing and beekeeping are common in the villages of Yenice. Mountainous terrain does not allow growing crops such as grains. Agricultural production takes places near settlements in the form of green housing, orchards and gardening. Hazelnut, walnut, fig, persimmon, and vegetables are among the most produced crops. Animal husbandry is mostly based on cattle. Villagers also benefit from the forest for collecting berries, mushrooms and linden tree leaves.

1.1.2. Camera Trapping

Studying large mammal species is not easy due to their low densities and elusive behavior (Sutherland, 2000). In general, large carnivores are hard to observe, as they mostly hide in dense habitats, and many survey methods may fail to detect them. It may be either impossible to produce population estimates or they may require intensive fieldwork with expensive equipment like radio collars (Linnell, Swenson, Landa, & Kvam, 1998). In this perspective, camera trapping is a non-invasive, low-cost technique, which causes minimal environmental disturbance. It can be used for studying elusive animals in difficult terrains where using other techniques can be difficult or impossible (Rowcliffe, Field, Turvey, & Carbone, 2008). It can also be used to collect data on several different species simultaneously in large areas. They can operate day and night and detect rare and cryptic species. Due to these reasons, among others, camera traps are popular survey tools (Ancrenaz et al., 2012).

Camera trapping is based on gathering presence data using automated camera devices. The cameras are equipped with infrared sensors, which are triggered by animals passing by the camera traps and photos can be taken at predetermined intervals. Camera traps can be set to take photos continuously; however, their batteries and picture storage capacities are limiting factors. Camera trapping has been used for different purposes so far. For instance, estimating the abundance and population size of elusive big cats such as jaguars (*Panthera onca*) (Soisalo, & Cavalcanti, 2006), leopards (*Panthera pardus*) (Balme, Slotow, & Hunter, 2009), and tigers (*Panthera tigris*) (Karanth, & Nichols, 1998) were possible employing camera traps. Camera trapping also provides information about species' behavior such as mating, hunting, and foraging (Bridges, Fox, Olfenbuttel, & Vaughan, 2004), and activity patterns of animals and/or species interactions (Gomez, Wallace, Ayala, & Tejada, 2005; Oliveira-Santos, Tortato, & Graipel, 2008; Sunarto, 2011; Linkie, & Ridout, 2011; Sollmann et al., 2012). In wildlife management and conservation, camera traps are mostly used to investigate parameters related to species occurrence and distribution in an area, and associated population parameters (Kays, & Slauson, 2008) such as carnivore and prey density (Rovero, Tobler, & Sanderson, 2010), mapping distributions (Moruzzi, Fuller, DeGraaf, & Li, 2002), occupancy analysis and habitat use (Linkie, Dinata, Nugroho, & Haidir, 2007), and population dynamics (Karanth, Nichols, Kumar, & Hines, 2006; Gardner, Reppucci, Lucherini, Royle, & 2010), among others.

Camera trapping has advantages for studying certain species and for carrying out certain lines of research when compared to other methods, e.g., for species inventories, it is more effective than interview (Can, & Togan, 2009), scat collection (Davison et al., 2002; Weckel, Giuliano, & Silver, 2006), and tracking plots and direct observations (Silveira, Jácomo, & Diniz-Filho, 2003). However, other techniques can be more effective for other objectives. For example, radio or GPS telemetry are better approaches for the estimation of minimum home ranges of large mammals (Gompper et al., 2006; Franklin et al., 1999).

In Turkey, several camera trapping studies have been undertaken in different regions, and with various research objectives. In northwestern Turkey, some graduate studies based on camera trapping of medium and large mammals were carried out in Yaylacık Research Forest, Yenice (Can, 2008), in Bartın, northern Karabük, and western Kastamonu (Soyumert, 2010), in Zonguldak (Çoğal, 2016), in Sarıçam Forest Management Chiefdom, Bolu (Naiboğlu, 2016), in Sökü Wildlife Development Area, Bartın (Horasan, 2018), in Aksu Forest Management Chiefdom, Düzce (Çömeten, 2018), in Gölyaka Forest Management Chiefdom, Düzce (Kahraman, 2019), in Düzce province (Algan, 2019), in Ballıdağ and Kurtgirmez, Kastamonu (Ugış, 2023), in Ilgaz and Gavurdağı Wildlife Development Areas, Kastamonu (Kalleci, 2023). Some other graduate studies focused on species such as brown bear (*Ursus arctos*) and grey wolf (*Canis lupus*) in Sakarya (Töke, 2023), European otter (*Lutra lutra*) in Bolu Abant Lake (Turan, 2016), and grey wolf in Bartın (Ertürk, 2010). There are also published articles based on camera trap research in northwestern Anatolia including a study comparing local knowledge and camera trap records in Yaylacık Research Forest (Can, & Togan 2009), assessing wildcat status via camera trap records and interviews (Can, Kandemir, & Togan, 2011), an inventory and investigation of daily activity patterns of large mammals (Akbaba, & Ayaş, 2012), studies focused on distribution and habitat preference of roe deer (*Capreolus capreolus*) in Düzce (Keten, 2017), habitat preferences of wolf (Tokmak, & Ambarlı, 2018), the importance of lagomorphs in the Eurasian lynx (*Lynx lynx*) diet (Soyumert, 2019), a study reporting breeding of Eurasian lynx at Mount Ilgaz Wildlife Reserve (Soyumert, 2020a), and investigating coexistence and patterns of Eurasian lynx and wildcat *(Felis silvestris*) in Kastamonu (Soyumert, 2020b).

1.1.3. Aim of the Study

In Yenice Forest, so far, no comprehensive camera trap surveys have been carried out to list mammal diversity. Inventories carried out by local directorates of Nature Protection and National Parks have been based on traditional methods such as line transects during the day, for a few days in a certain month of the year. The foresters are expected to report estimates of population sizes based on the signs or sights of animals in the area. Other than this, a camera trapping study (Can, 2008) was carried out in Yaylacık Research Forest (about 50 km^2) which is a relatively small portion of Yenice Forest, which is about 750 km^2 in area. Because of the high costs of digital camera traps at the time, this study utilized analog cameras that used film and could take up to 36 photos. The data acquired were limited when compared to digital camera traps because frequent field visits to replace films were not possible. Despite these limitations, mammals such as brown bear, grey wolf, jackal, red fox, wild cat, wild boar, roe deer, red deer, pine marten, badger, and hare were captured by the camera traps in this previous study. However, although being reported during interviews with locals, lynx was not detected by camera traps, probably indicating that local knowledge is somewhat unreliable according to Can and Togan (2009). Lynx presence in the area has not been proven and remains a question.

In this part of the study, we aim to

- document the medium and large mammal biodiversity, capture rates, and distribution of medium and large mammal species in the study area,
- explore lynx presence in the area,

with a systematic large-scale camera trapping survey. Considering the human disturbance and poaching potential that comes with legal logging activities, the “wildlife development area” status does not seem to provide Yenice Forest actual protection. We hypothesize that there are no differences between YWLDA and the surrounding forest regarding medium and large mammal distributions and capture rates of medium and large mammals.

This chapter aims to provide a general overview of the study area and medium and large mammals in the study area. For further analysis of medium and large mammals in the study area, please see Chapter 4 Community Occupancy Modeling, Species Richness and Interaction of The Brown Bear with The Other Mammals in Yenice Forest.

1.2. Methods

The study area, Yenice Forest, is in northwestern Turkey, to the west of the city of Karabük and southeast of the city of Zonguldak. The study area is about 500 km^2 and covers YWLDA and a portion of the surrounding forest (Figure 1.1). We carried out systematic camera trap surveys by deploying 40 camera traps on forest roads and paths. Bushnell Model 119547 camera traps with hyper night vision and 24m infrared sensory distance were used and set to take three photos with one second intervals when triggered. Changing the locations of camera traps made it possible to survey a larger area than 40 camera traps can cover. Camera trapping was carried out in 2015 and 2016 in summer, and autumn, when field conditions were permitting, avoiding periods of heavy rain and snow, which made the forest inaccessible. On a few occasions, camera traps were not in use in the entire period described above, either because batteries died or SD cards filled up quickly. At the end of the study, only 24 of the 40 camera traps that we started were still present, due to theft.



Figure 1.1. Location of the study area, Yenice Wildlife Development area, and surrounding forest.

Camera trapping was carried out by placing 40 camera traps in a grid system. QGIS 3.4.2 (QGIS Development Team 2018) was used to design a regular grid for camera trap locations. 2 km x 2 km length grids were used for planning, following standard camera trapping protocols, to be able to detect mammals with smaller home ranges (Sanderson, 2014). However, due to field conditions and availability of spots that made the setting of the camera traps possible, the distance between traps varied ± 500 m (Figure 1.2 and Figure 1.3). At each session, half of the available camera traps were placed within the YWLDA, and the other half were placed outside of the YWLDA. Camera traps were placed on forest paths and roads to increase the probability of capture (Karanth, & Nichols, 2002; Simcharoen, Pattanavibool, Karanth, Nichols, & Kumar, 2007). Camera traps were checked every 15 days (when possible) or one month, to change their batteries and to retrieve the recorded data. At the end of each survey period, camera traps were transferred to a new location, in the adjacent forest patch. Camera traps were set monthly in June, July, August, and September in 2015, and bimonthly between mid-June and mid-August, and September-October in 2016 (Figure 1.2 and 1.3). The number of camera traps and the nature of the study area did not allow the camera traps be placed on the same day. In the first year, the first camera trap was set on 20 May 2015 and the last camera trap was collected on 19 October 2015. In the second year this period was between 14 June 2016 and 9 October 2016.

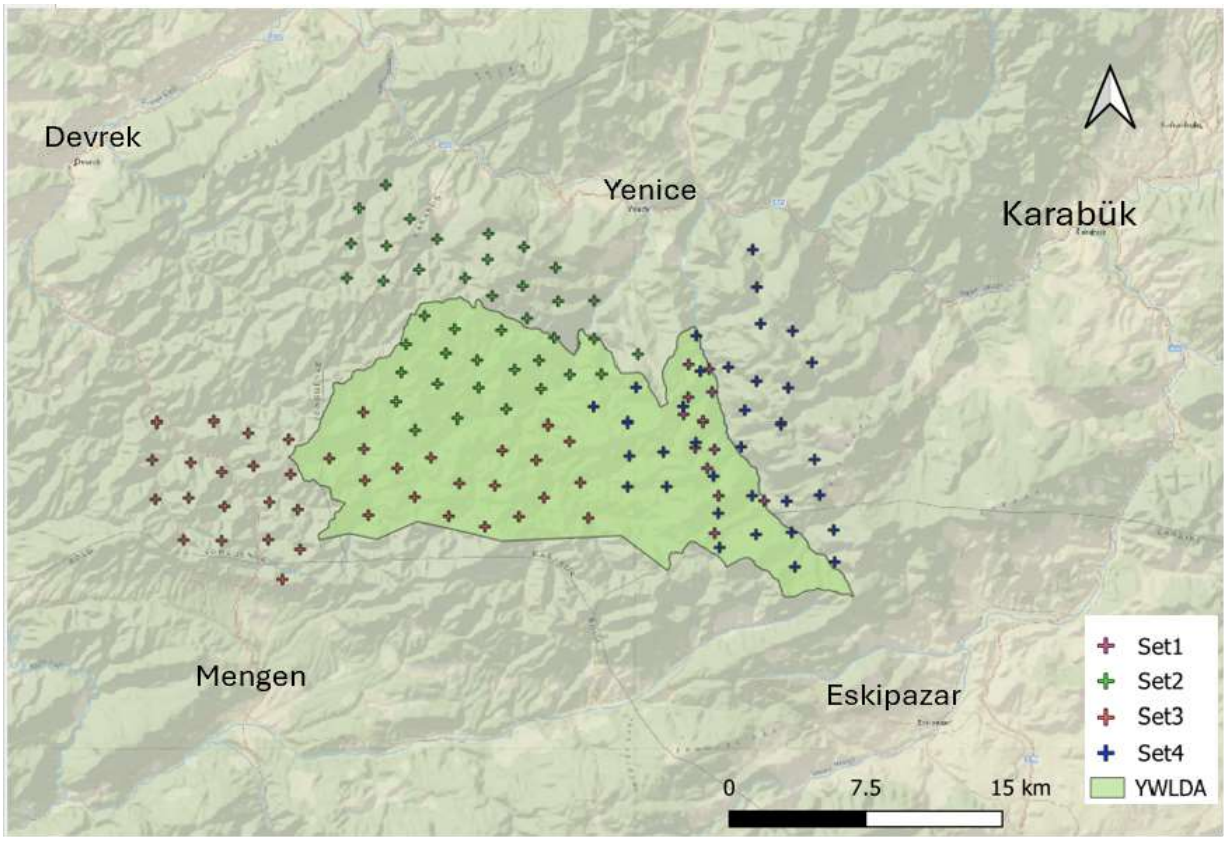


Figure 1.2. Camera trap setting in 2015.

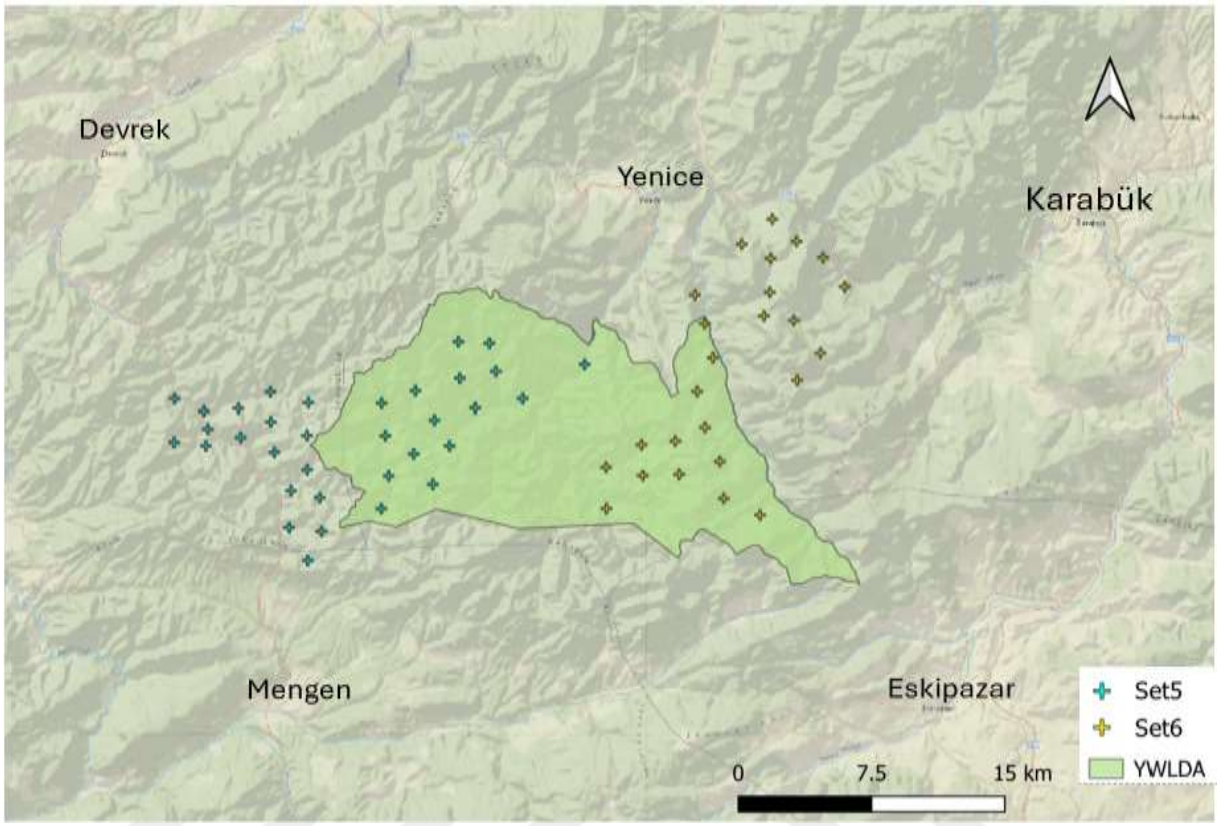


Figure 1.3. Camera trap setting in 2016.

Camera trap photographs were sorted and empty photos (e.g., photographs taken as a result of false triggers such as shade, and do not contain animal, human, vehicle etc.) were deleted. Photographs with an animal in it were kept and data obtained from these photographs were recorded. The camera trap database included coordinates of the camera trap, elevation, camera start, and end dates, species, date, and hour of capture.

Some camera traps did not function perfectly throughout the planned period. Therefore, the total number of days a camera trap was active (trap nights) was calculated for each camera trap location, and this variation was taken into account for the analyses. Successive photographs of the same species at a camera trap were considered independent captures when they were >15 min. apart. Capture rates (CR) were calculated based on independent captures of animals adjusted by the duration that camera traps were active (CR = [number of independent captures/trap nights] x 100). This formula has also been used in literature as the 'relative abundance index' (RAI). Although capture rates can be used to estimate population densities, camera trapping procedures must be standardized in time and over different sites, to obtain a valid density index for a target species (Rovero, & Marshall 2009). Detection probabilities of

species vary depending on the differences in study sites, behaviour of the species, life history traits, home range sizes, hence RAI is an unreliable method to compare abundance of species (Blount et al., 2021). Therefore, capture rates, in this case, were used to get an overall picture of the area regarding the presence of mammals and not as a proxy to compare abundance between sites or studies.

1.3. Results

The total number of days that a camera trap was active (*i.e.*, trap nights) was calculated for each camera trap location and for each survey period. The total survey effort was 3090 trap nights in 2015 and 1845 trap nights in 2016 (Table 1.1). Fourteen different mammal species, five different domestic animals (*i.e.*, dog, sheep, goat, cattle, horse), and humans were detected by camera traps in 2015 and 2016. Vehicle captures (e.g., trucks, tractors, private cars, official cars, heavy equipment) were the highest and made up 30% of all captures in 2015 and 54% of all captures in 2016 (Table 1.2). Vehicle captures occurred at 59 out of 160 camera traps in total. 410 of the photographs with an animal could not be identified, mostly due to blurred night shots. Stone and pine martens were grouped under *Martes* spp. because it was not possible to identify them at the species level based on the photographs.

Table 1.1. Camera trap (CT) stations and respective trap nights (TN).

	# CT stations set	**# CT stations included in the analyses**	**# TN included in the analyses**
June 2015 (Set 1)	12	9	383
July 2015 (Set 2)	40	37	1075
August 2015 (Set 3)	39	35	972
September 2015 (Set 4)	34	32	660
June-August 2016 (Set 5)	27	26	1065
September-October 2016 (Set 6)	21	21	780
2015 Total	125	113	3090
2016 Total	48	47	1945
Grand Total	173	160	4935

Table 1.2. Total number of captures per species, per year.

	2015 all	2015 filtered	2016 all	2016 filtered	Total all	Total filtered
Anatolian squirrel (*Sciurus anomalus*)	11	11	4	4	15	15
Badger (*Meles meles*)	41	41	18	18	59	59
Brown bear (*Ursus arctos*)	147	134	83	76	230	210
European hare (*Lepus europaeus*)	4	4	1	1	5	5
European lynx (*Lynx lynx*)	3	3	1	1	4	4
European wildcat (*Felis silvestris*)	175	173	113	112	288	285
Golden jackal (*Canis aureus*)	125	121	321	312	446	433
Grey wolf (*Canis lupus*)	91	84	25	24	116	108
Hedgehog (*Erinaceus concolor*)	51	50	18	17	69	67
Marten (*Martes* spp.)	259	256	73	69	332	325
Red deer (*Cervus elaphus*)	109	99	54	48	163	147
Red fox (*Vulpes vulpes*)	209	200	210	202	419	402
Roe deer (*Capreolus capreolus*)	216	204	106	99	322	303
Tortoise (*Testudo* sp.)	2	2	4	4	6	6
Wild boar (*Sus scrofa*)	147	133	88	88	235	221
Domestic dog	34	33	26	25	60	58
Domestic cat	0	0	2	2	2	2
Horse	3	2	3	3	6	5
Human	105	92	71	63	176	155
Vehicles	858	682	1703	1350	2561	2032
Livestock	28	23	11	11	39	34
Unidentified canid	123	118	195	191	317	308
Unidentified general	81	79	12	12	93	91
Total	2822	2544	3142	2732	5963	5275

In this study, camera trapping targeted medium and large mammals, although some random small mammals were captured by some of the camera traps, especially on forest paths. Since our camera trap design was set for capturing medium and large mammals, the other species were excluded from further statistical analysis. Capture rates for 12 mammal species were calculated based on trap nights for each year, for each area status, and for each camera trap location. European lynx had the lowest and golden jackal had the highest capture rates in total. The capture rate of the golden jackal was about four times higher in 2016 when compared to 2015. The capture rate of marten was prominently higher in 2015 and red fox in 2016. The capture rates of other target mammals between the years were not considerably different (Figure 1.4).

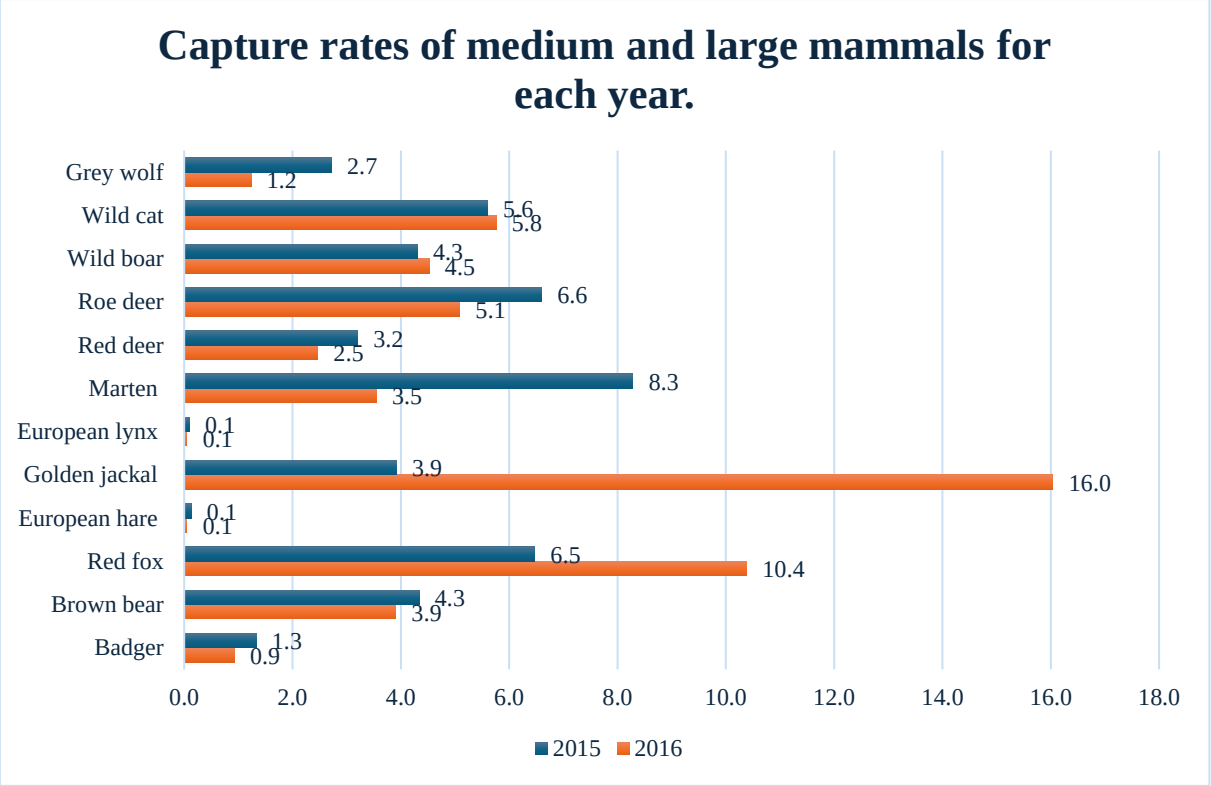


Figure 1.4. Capture rates of large-medium mammals for each year.

Camera traps were active for 2636 trap nights in YWLDA and 2299 trap nights in nonprotected areas. When species capture rates were compared based on the camera trap location, being either inside or outside of YWLDA, capture rate of red deer was about two times, roe deer about one and a half times, and marten was about two times higher in YWLDA. Red fox capture rate was about two times higher and badger capture rate was about two and a half times higher in non-protected areas. Brown bear capture rate was also slightly higher in non-protected areas (Figure 1.5). However, these rates do not indicate their abundance or site preferences since other factors might have influenced their detections, thus capture rates. Capture rates of other species did not show considerable differences between YWLDA and non-protected forest.

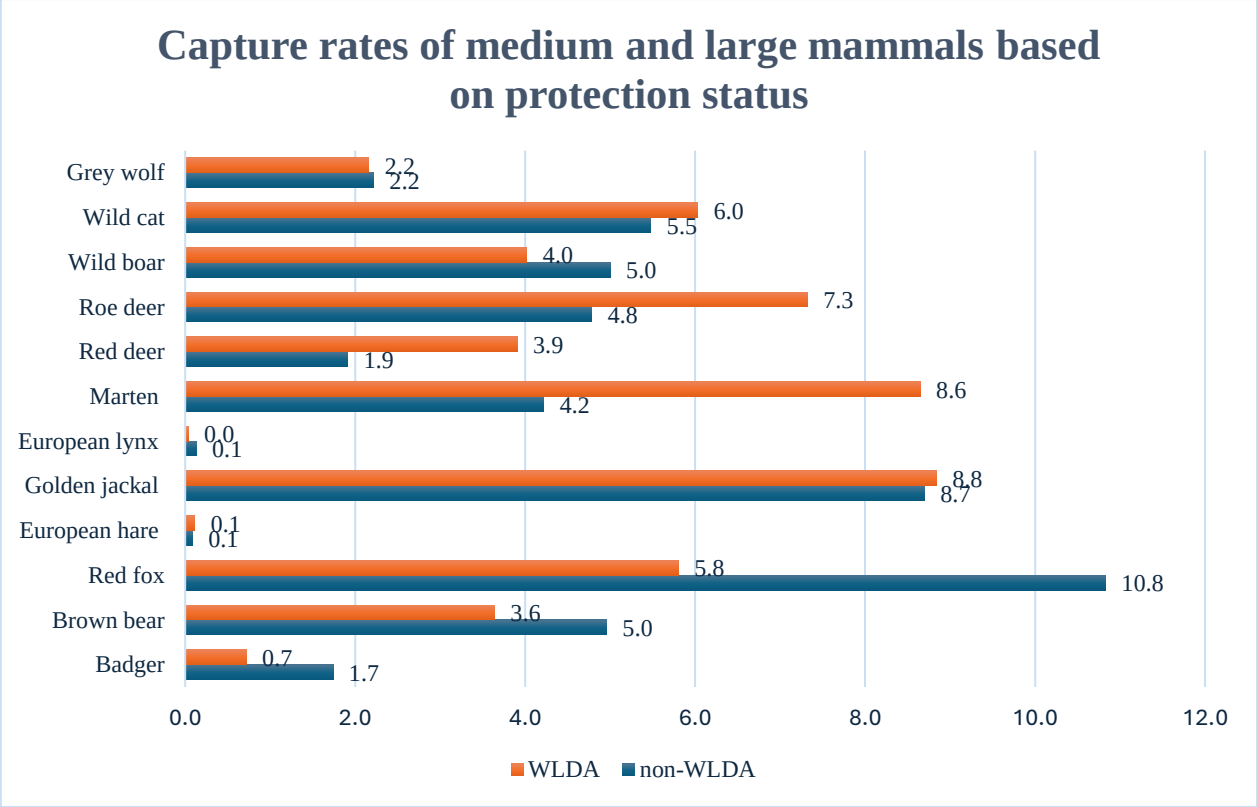


Figure 1.5. Capture rates of medium and large mammals based on protection status.

Camera traps were active for 2797 trap nights in the forest path and 2138 trap nights on the forest roads. When compared based on camera trap location (roads vs. paths), only red deer and roe deer capture rates were higher on the paths. European lynx was only detected on forest roads. Especially canid capture rates on roads were prominently higher compared to forest paths. In general, carnivores were mostly recorded on the forest roads, and ungulates were mostly recorded on the paths, except for wild boar (Figure 1.6).

European lynx was captured by the camera traps three times in 2015 and one time in 2016. Three records belonged to the non-protected area east of YWLDA, and one to the eastern YWLDA. As the photographs were not clear enough to identify the individuals by their spot patterns, we cannot be sure if these are the same individual or not (Figure 1.7). It was also not possible to identify the sex of these individuals.

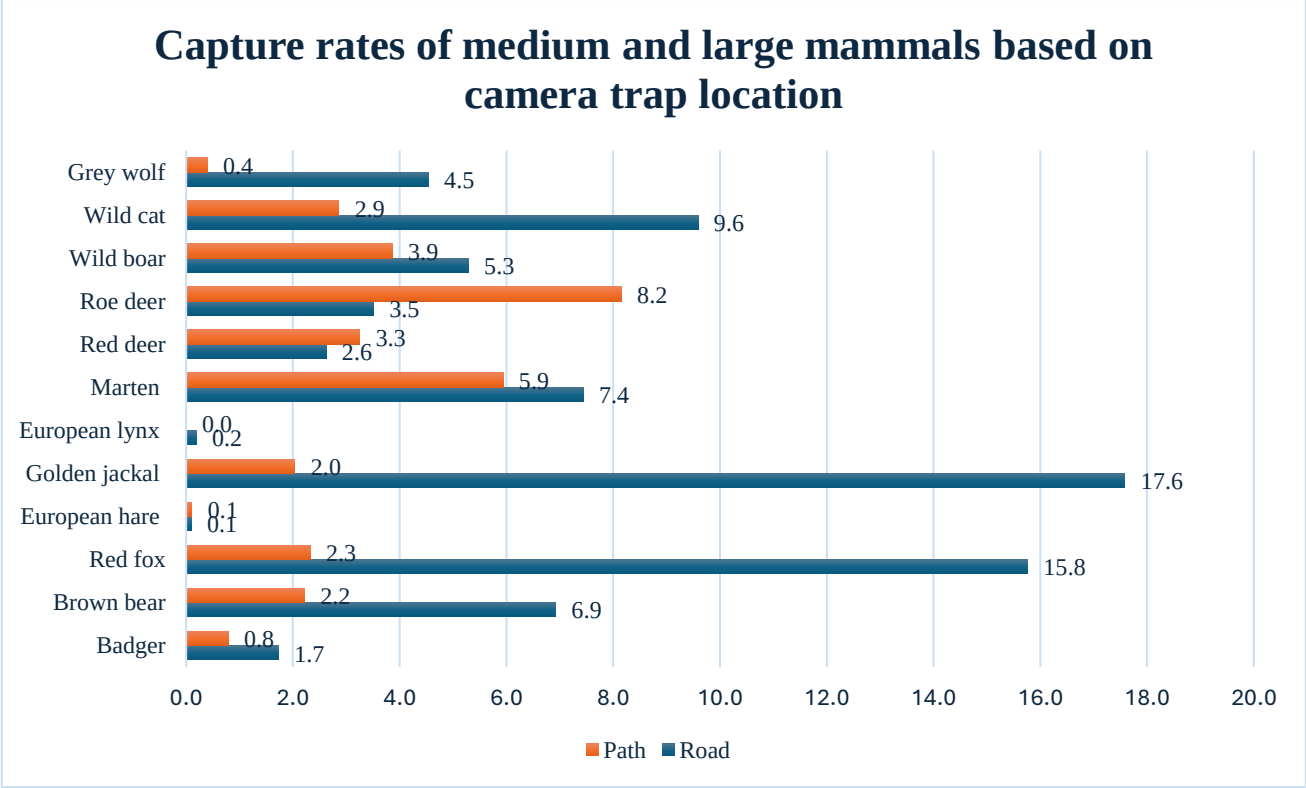


Figure 1.6. Capture rates of large-medium mammals based on camera trap location.



Figure 1.7. Lynx images that were captured in 2015 and 2016.

The distribution map of the 12 target mammal species based on the camera trap captures are shown in Figures 1.8 – 1.19. Capture frequencies are shown with black dots in different sizes, and camera traps with zero detection of related species are shown with a cross. Different colors indicate different camera trap surveys. Set 1-4 represent the camera traps from 2015, which were replaced monthly. Set 5 and 6 represent camera traps from 2016 which were replaced bimonthly. Details regarding the active camera trap nights and trap sets are given in Table 1.1. Although the size of the dots represents capture frequencies, we cannot safely assume higher abundance in the areas with larger dots. However, these maps can provide a perspective regarding the distribution of the species in the study area.

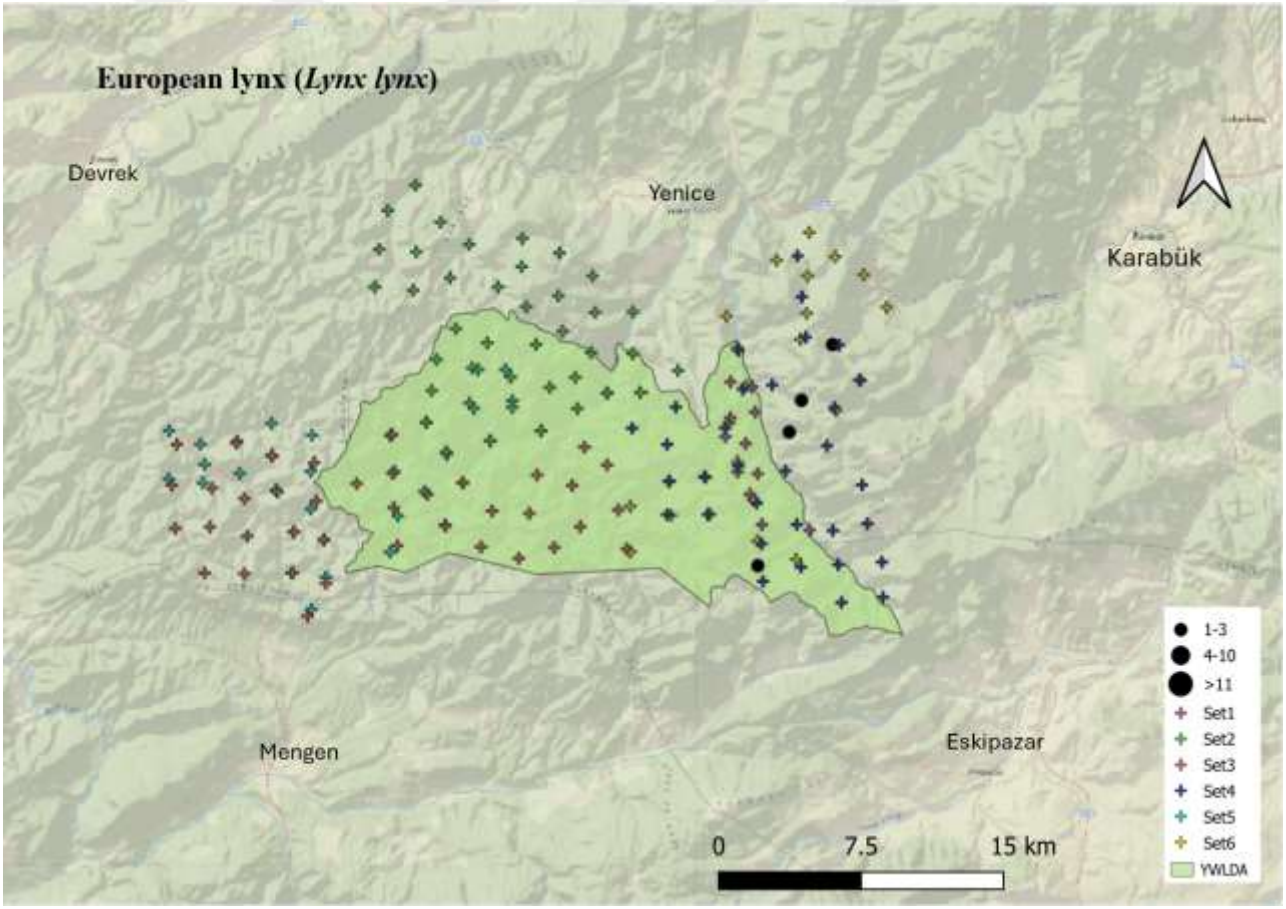


Figure 1.8. The camera traps with European lynx (*Lynx lynx*) records.

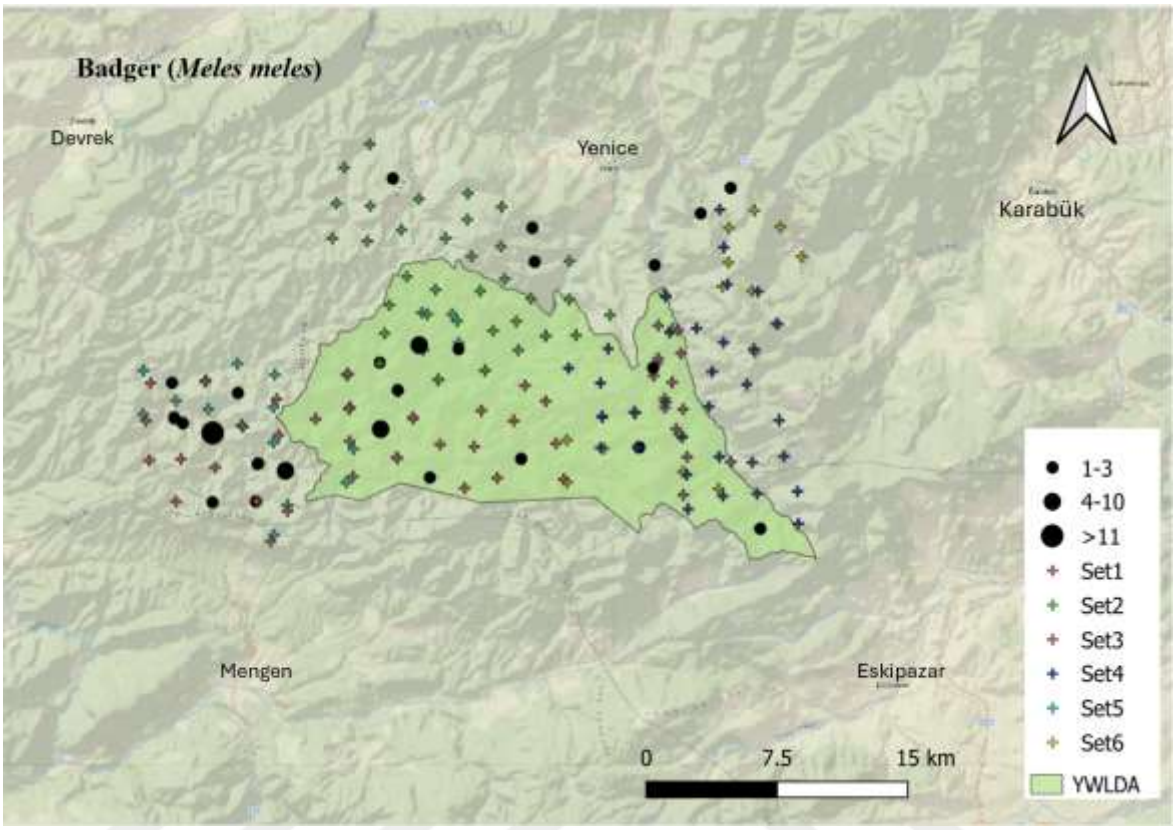


Figure 1.9. The camera traps with badger (*Meles meles*) records.

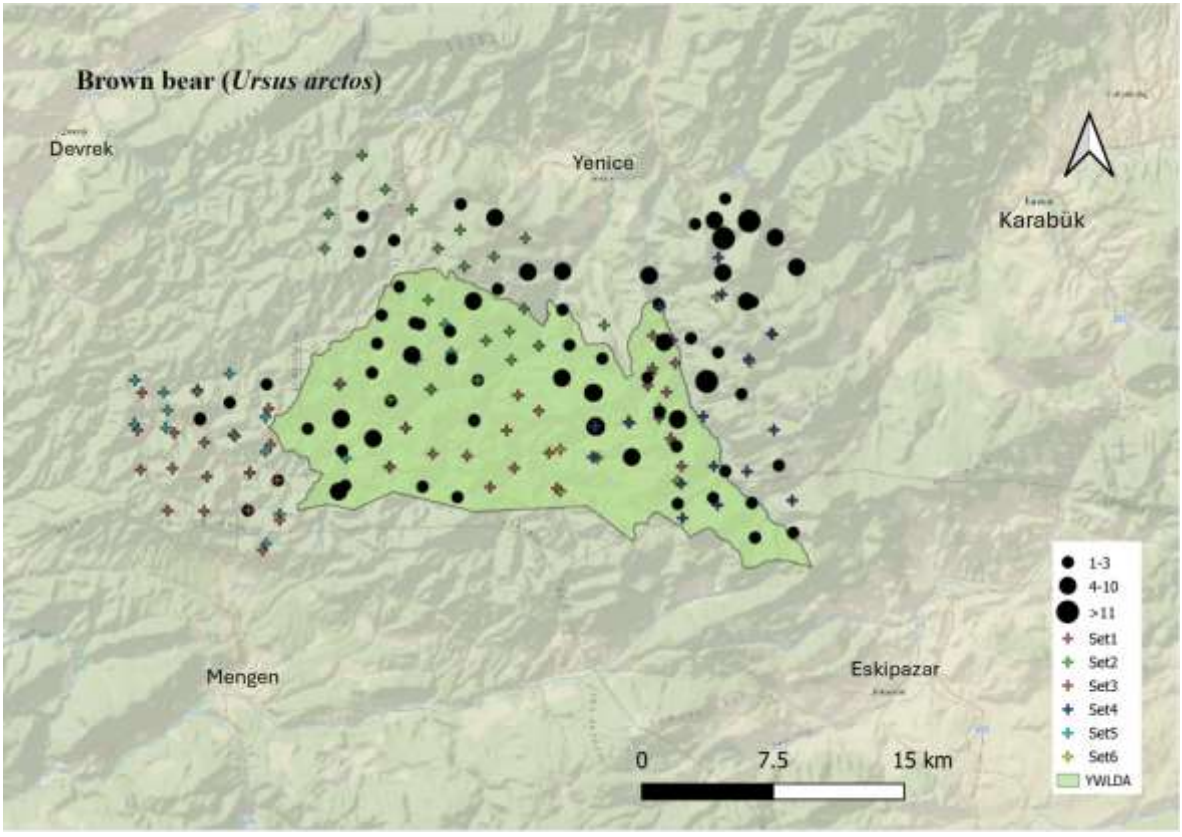


Figure 1.10. The camera traps with brown bear (*Ursus arctos*) records.

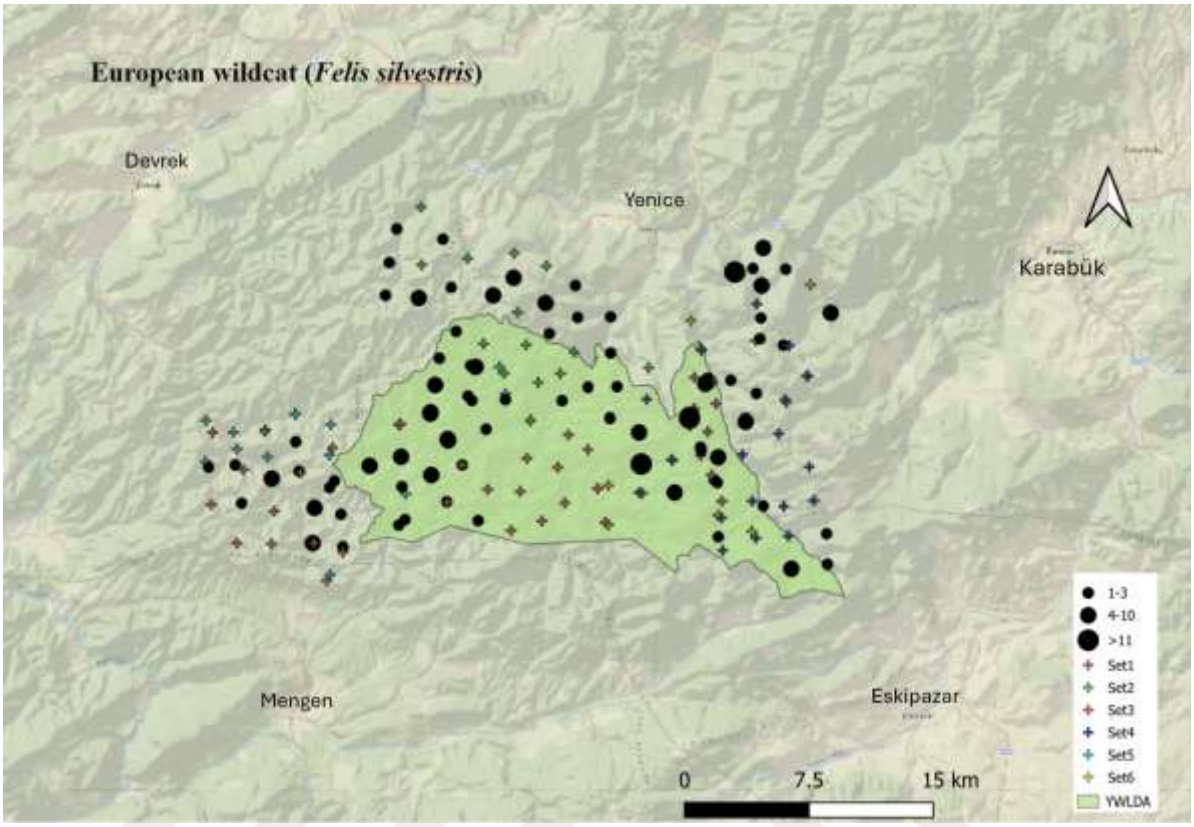


Figure 1.11. The camera traps with European wild cat (*Felis silvestris*) records.

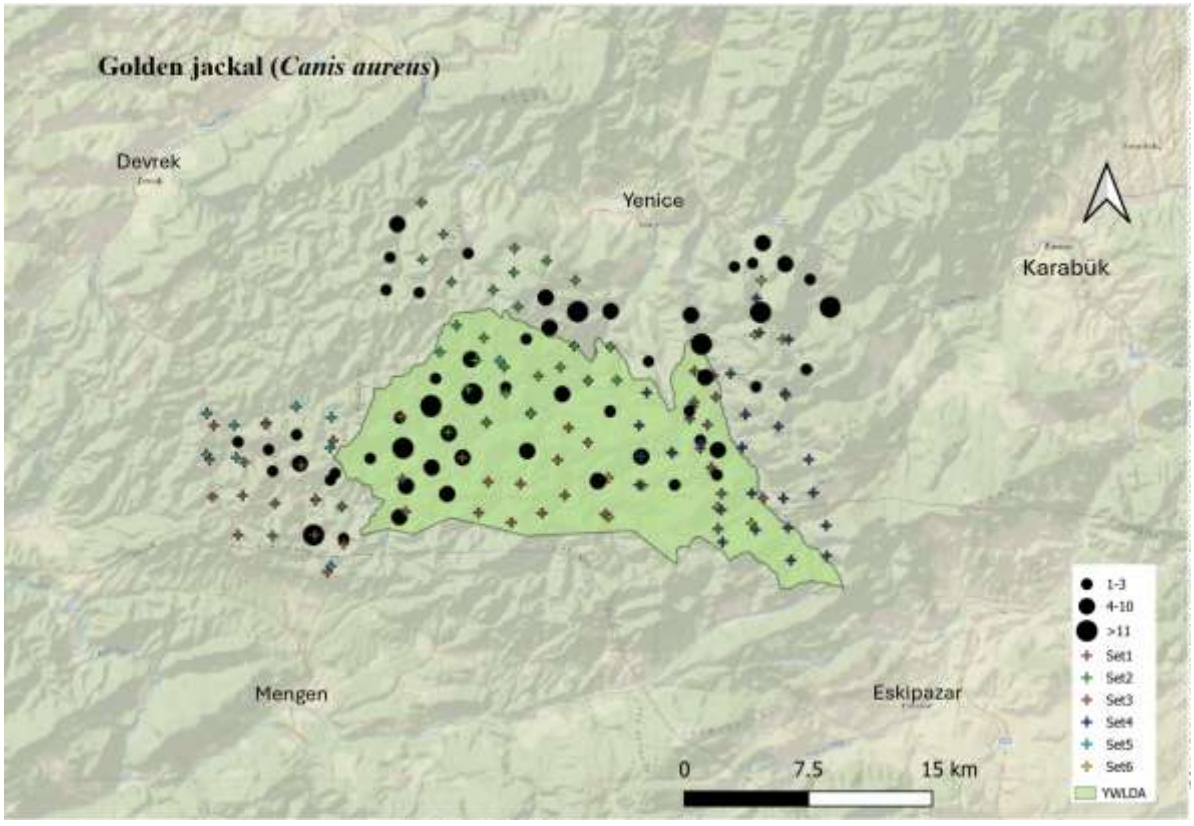


Figure 1.12. The camera traps with golden jackal (*Canis aureus*) records.

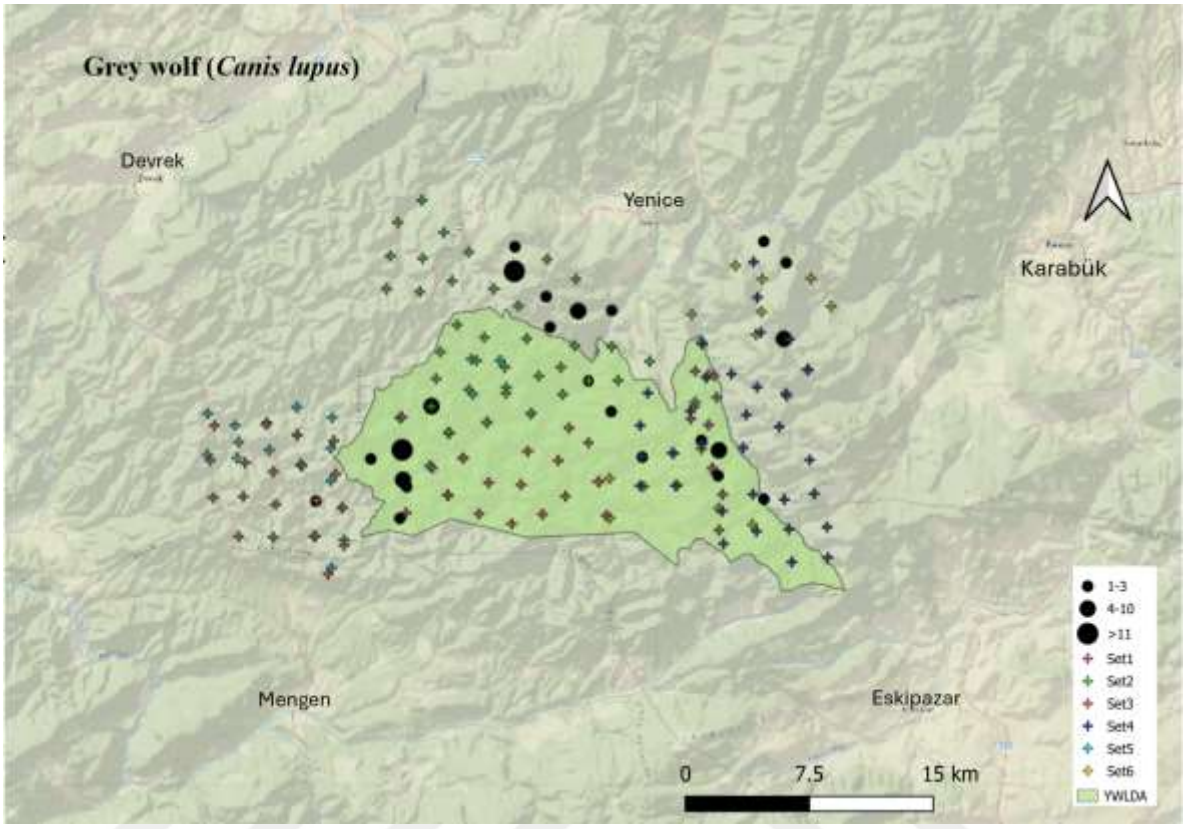


Figure 1.13. The camera traps with grey wolf (*Canis lupus*) records.

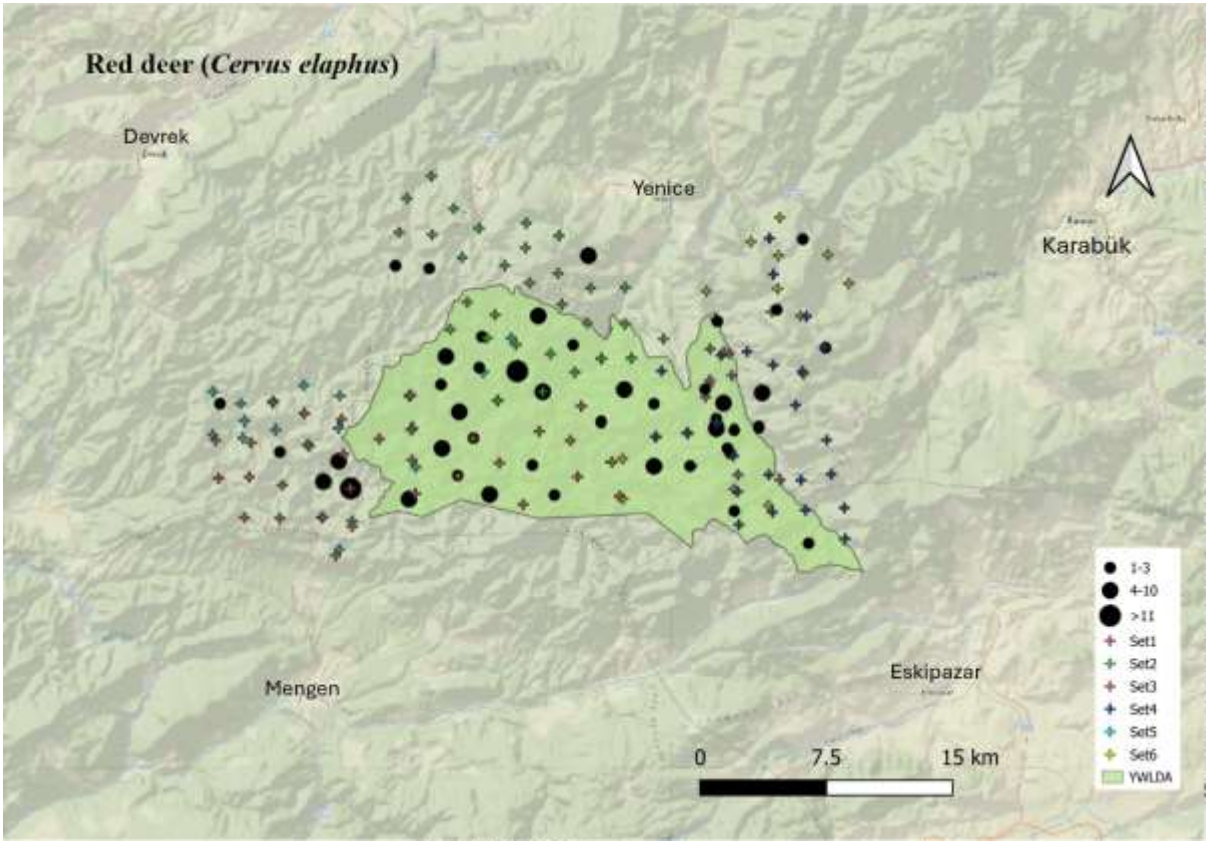


Figure 1.14. The camera traps with red deer (*Cervus elaphus*) records.

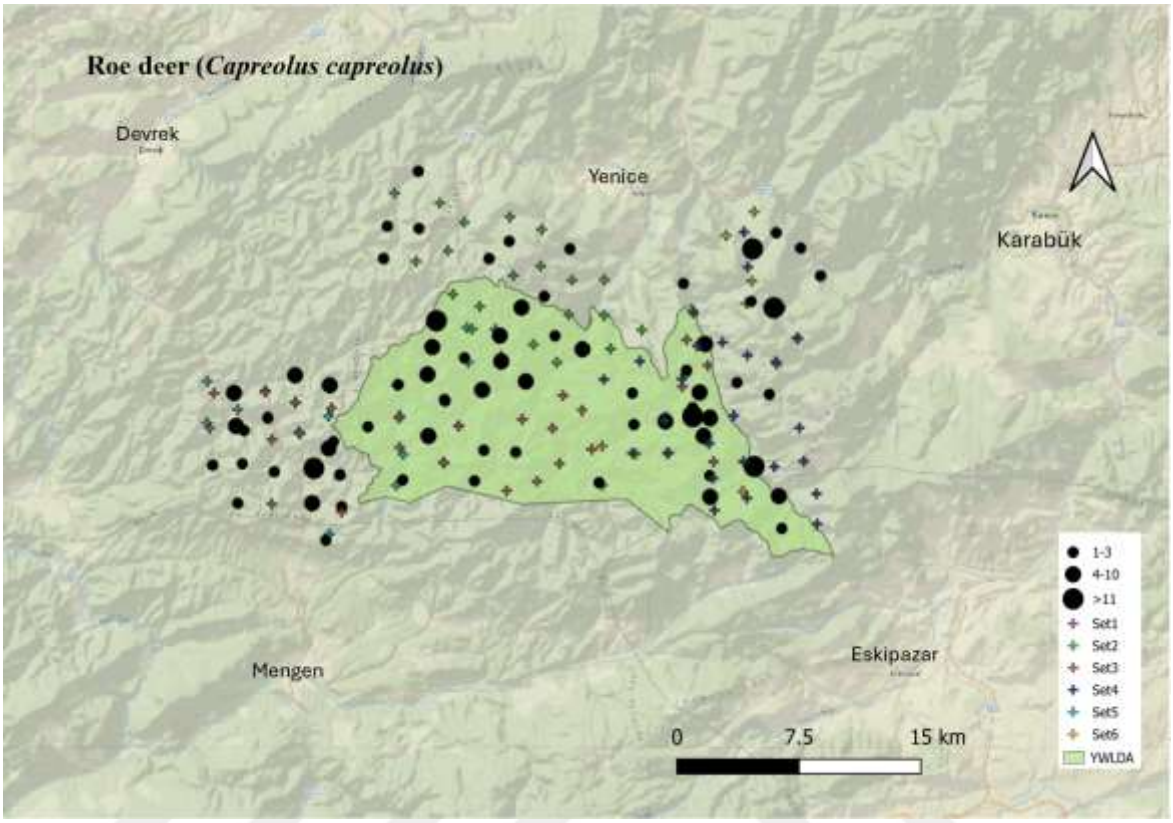


Figure 1.15. The camera traps with roe deer (*Capreolus capreolus*) records.

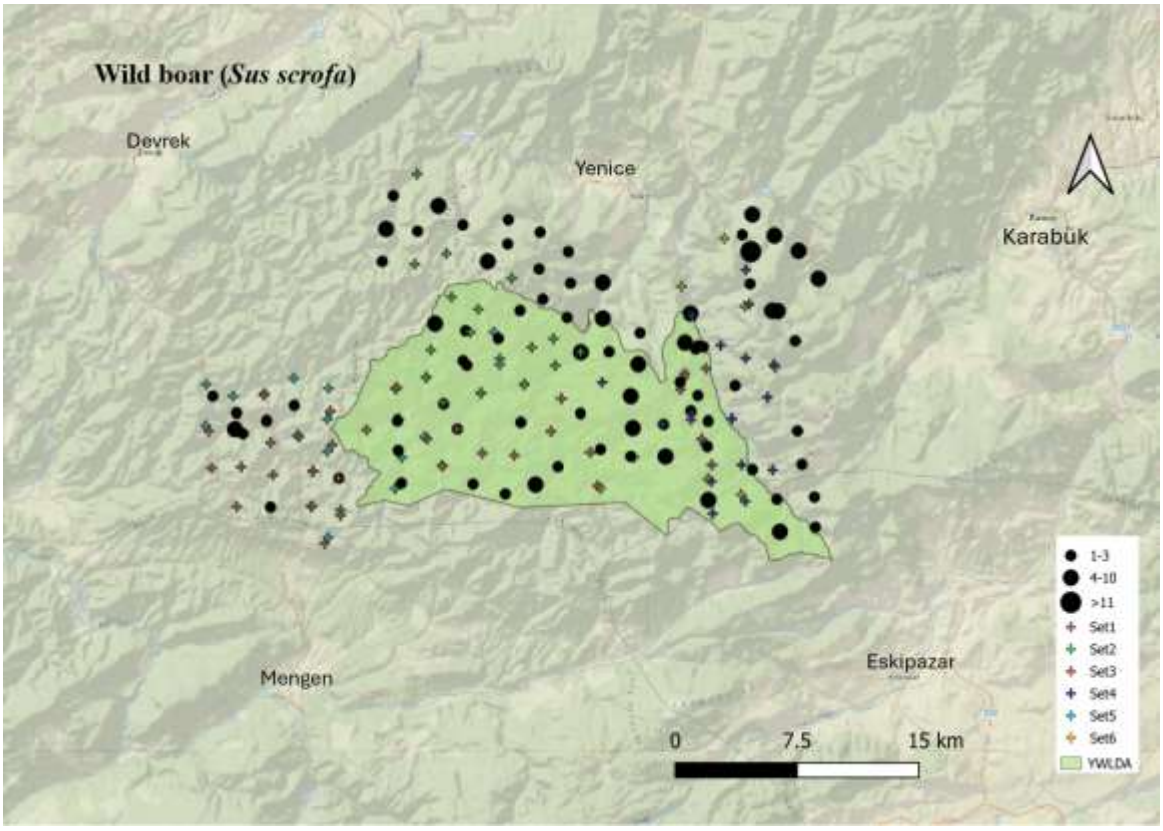


Figure 1.16. The camera traps with wild boar (*Sus scrofa*) records.

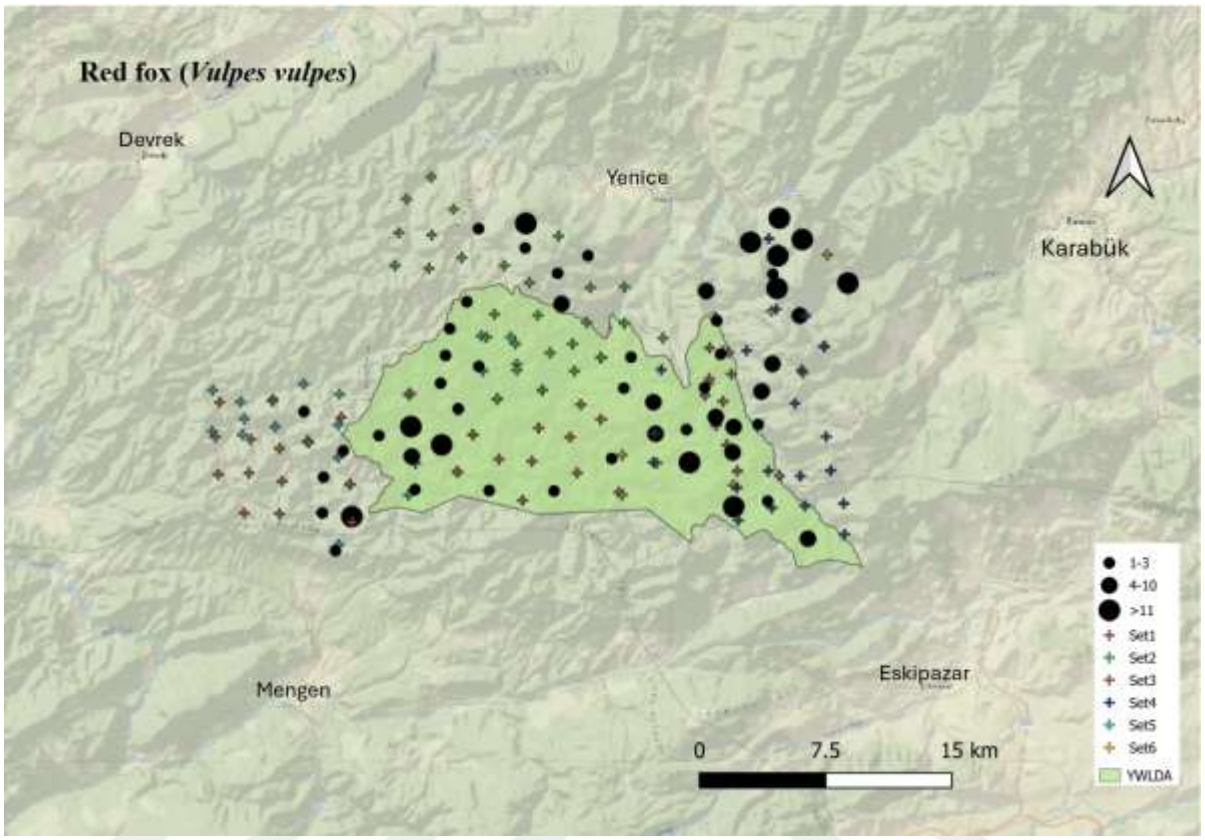


Figure 1.17. The camera traps with red fox (*Vulpes vulpes*) records.

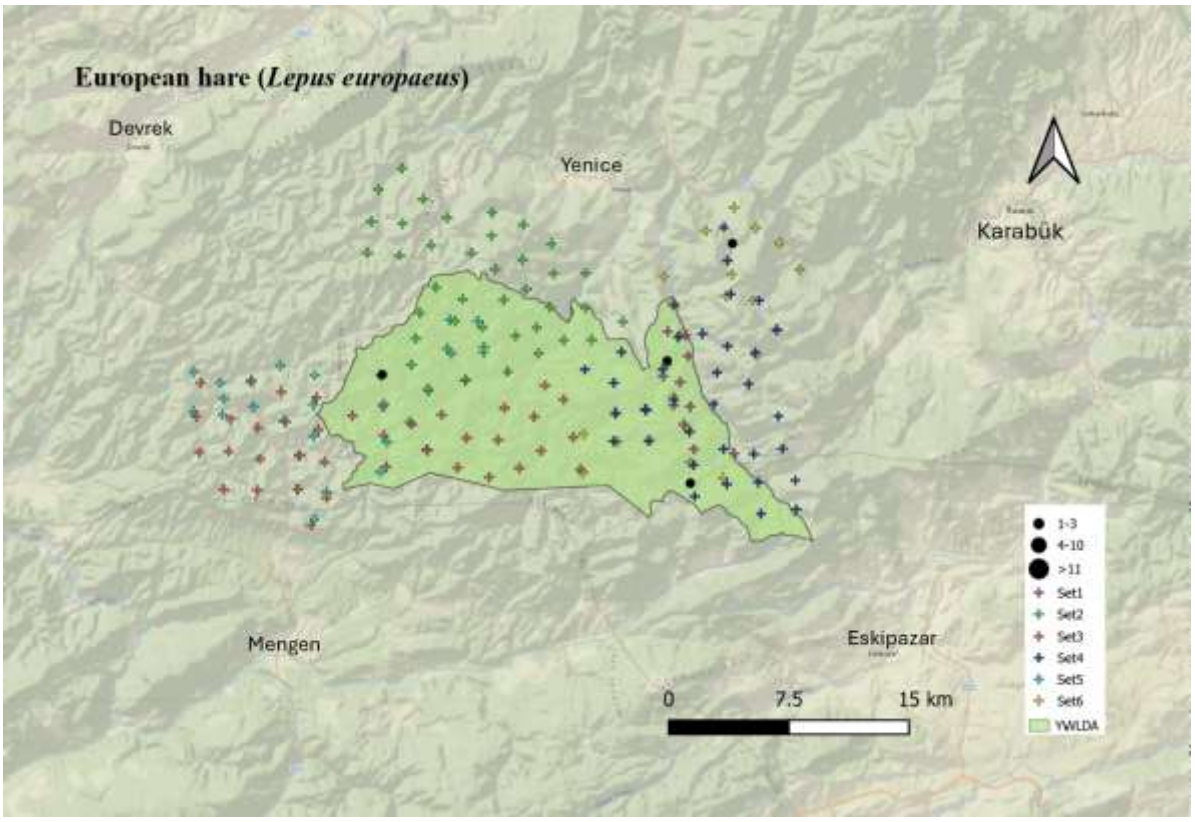


Figure 1.18. The camera traps with European hare (*Lepus europaeus*) records.

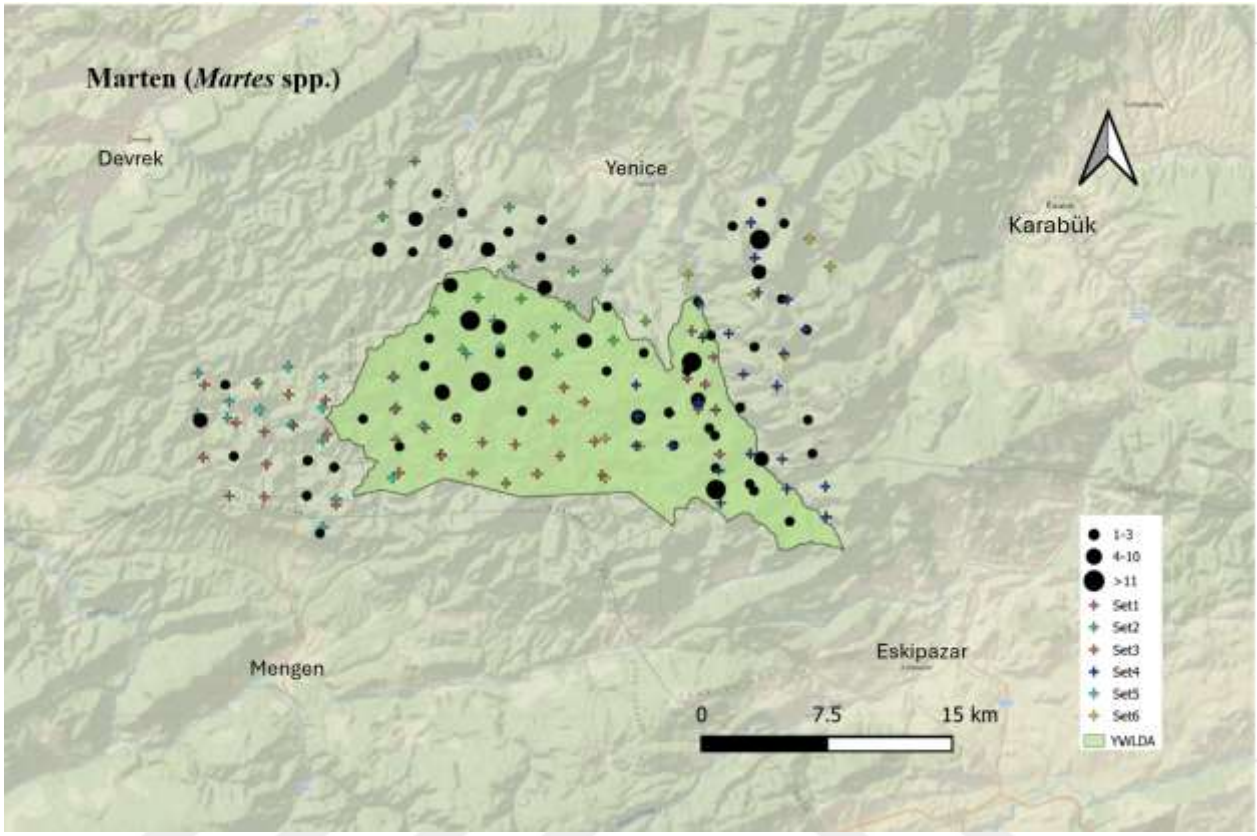


Figure 1.19. The camera traps with marten (*Martes* spp.) records.

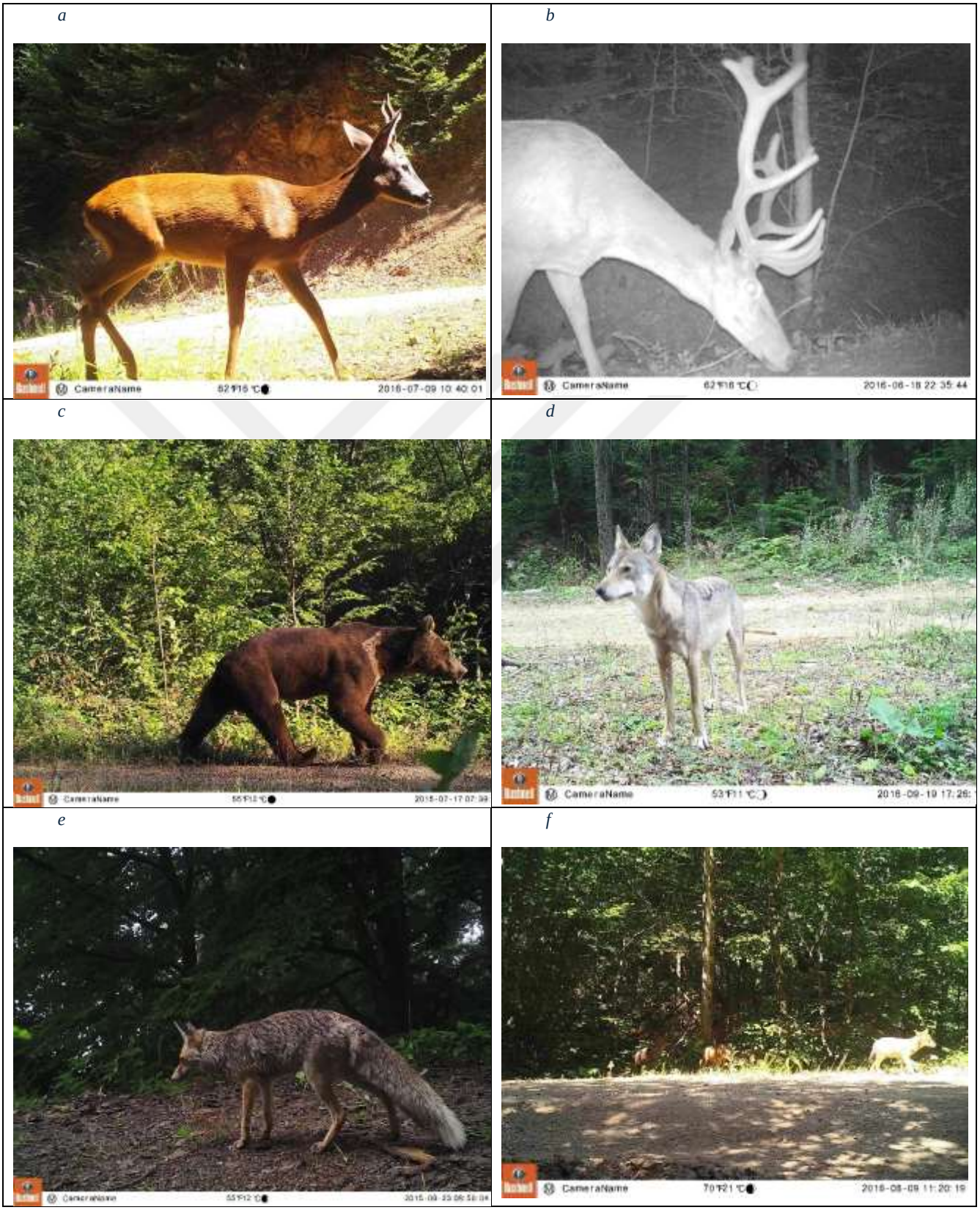


Figure 1.20. The medium and large mammals captured by the camera traps in the study area. a) Roe deer (*Capreolus capreolus*), b) Red deer (*Cervus elaphus*), c) Brown bear (*Ursus arctos*), d) Grey wolf (*Canis lupus*), e) Red fox (*Vulpes vulpes*), f) Golden jackal (*Canis aureus*).

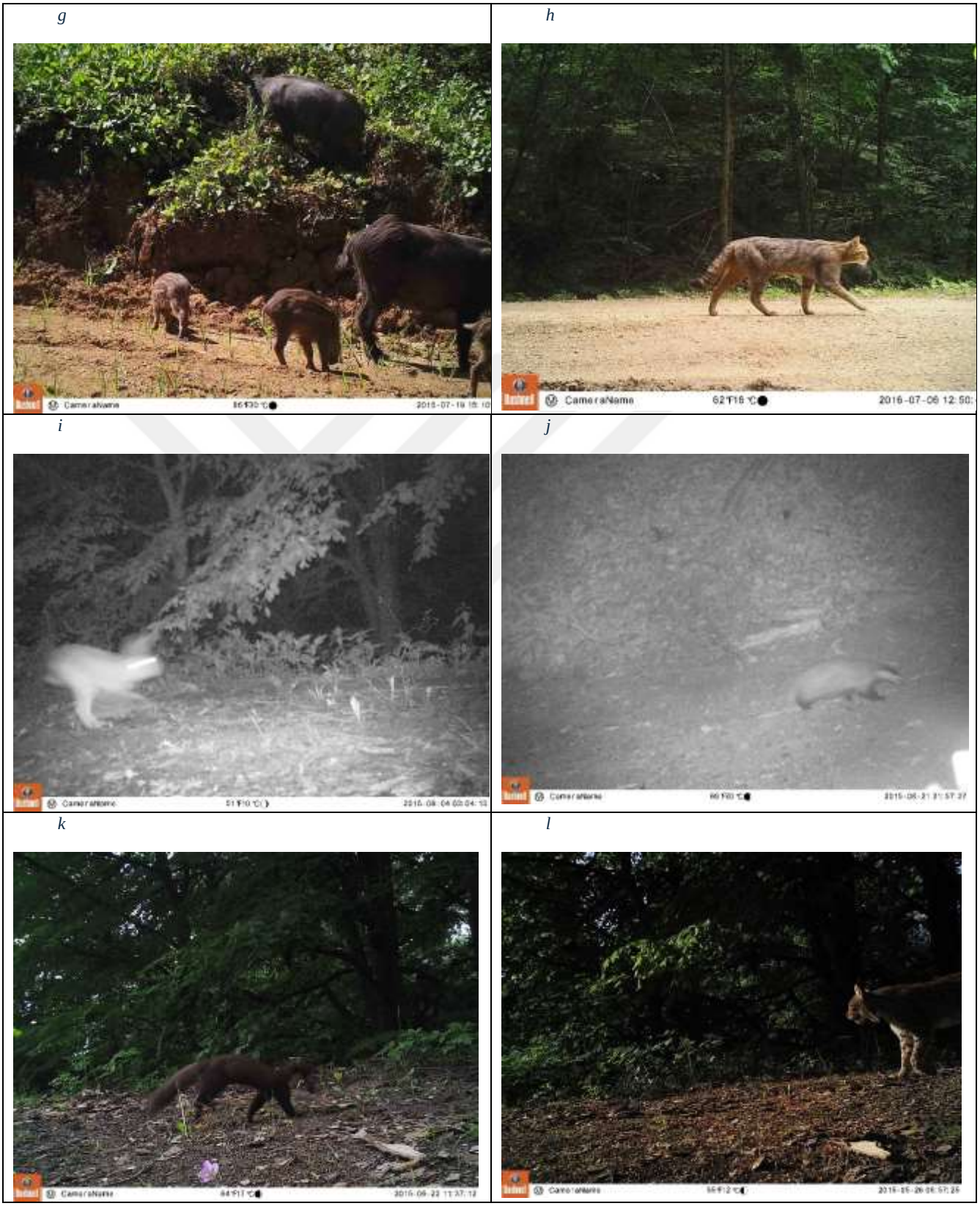


Figure 1.20. (cont). The medium and large mammals captured by the camera traps in the study area. g) Wild boar (*Sus scrofa*), h) European wild cat (*Felis sylvestris*), i) European hare (*Lepus europaeus*), j) European badger (*Meles meles*), k) Marten (*Martes* spp.), l) European lynx (*Lynx lynx*).

On many occasions hunting dogs and men with rifles were captured on the camera traps. Loggers who stay in the forest also carry rifles and it is not possible to tell whether these people had committed hunting in YWLDA or carrying rifles to feel safe as they claimed. No action of the hunt was photographed during the study, although gunshots were heard by the team during surveys. Hunting dogs were observed near sayvans, which are temporary housings of loggers in the forest, as mentioned above. Interviews with local people and members of the local hunting club showed that although there are licensed hunters that hunt permitted species in the forests surrounding YWLDA, which are officially state hunting grounds, there are also people who carry guns and shoot whatever they see in the forest disregarding the protection status of the area or the species (Figure 1.21).



Figure 1.21. Men with rifles in Yenice Wildlife Development Area.



Figure 1.22. Photos from field work.

a, b, c) Camera trap setting d) Scat collection e, f) Interviews with the local people.

1.4. Discussion

In this part of the study, we provided a general overview of Yenice Forest mammal fauna by listing the mammal biodiversity that could be detected by camera traps, and by examining the capture rates of medium and large mammals and their distributions in the study area. This was the first extensive camera trapping study in YWLDA and the surrounding forest and provided some insight into the frequency of human use of the forest, as well. Overall, the largest number of photographs were of vehicles, including trucks, tractors, private cars, official cars, and heavy equipment. This clearly shows the high level of human activity in this forest inhabited by protected species. High human activity, especially forestry activities with long-term human presence, noisy equipment, and logging trucks in the forest can disturb animals in the forest. In a camera trap study in Düzce, northeastern Anatolia, forest maintenance activities were reported to influence especially carnivores, and resulted in them leaving the area for long periods (Kahraman, 2019). According to this study, red fox and marten were the most affected species by the forest maintenance activities. In a study in Bolu, similar results were reported with the most affected species being wild boar and roe deer (Naiboğlu, 2016).

The camera traps also recorded many photographs of armed people, a clear sign of the threat to the wildlife of the Yenice Forest. According to Turkish Hunting Law (No. 4915), Wildlife Development Areas are areas where game and wild animals, and wildlife are protected, game animals are introduced, measures are taken to improve their habitats, and hunting is allowed, when necessary, within the framework of special hunting plans. In 2015 and 2016, no permits were issued for hunting within Yenice Wildlife Development Area. However, camera trap records and gunshots heard during the fieldwork show there has been poaching in the area. During site visits, conversations with locals revealed that people come to the forest to hunt, and because it is a remote, dense forest, poachers can make their way through it. The Local Directorate of Nature Protection and National Parks is based in Karabük and is about one hour drive from the WLDA. There are multiple entrances to the forest and there are no established security systems at the entrances or within the forest. Although being successful in some cases to catch the poachers in the act, the institution does not have enough personnel, vehicles, and budget to control this large forest by patrolling on a daily basis.

The presence of lynx in the area was uncertain at the onset of this study. Based on interviews with villagers, Can and Togan (2009) reported lynx as being present in the area. However, a previous camera

trap study carried out in Yaylacık Research Forest (about 50 km^2), which is a part of Yenice Forests, showed no evidence of lynx in the area (Can, 2008). A study in the east of Yenice WLEA, that covered neighboring forests including Kartdağ WLDA, Sökü WLDA, and Küre Mountains National Park with 186 camera trap stations did not report any lynx records (Soyumert, 2010). Another large-scale study in the western Black Sea region reported the presence of Eurasian lynx at Ilgaz, Gavurdağ and Elekdağ (Soyumert, Ertürk, Tavşanoğlu, 2019). Camera trap records of Eurasian lynx cubs in Ilgaz Wildlife Development Area show that the species is breeding in this WLDA (Soyumert, 2020). In Bolu, in the west of Yenice WLDA, two records of lynx were reported. Although the camera trap photos were not clear enough to examine the pattern of spots on their coats, these were suspected to be the same individual captured by two different cameras (Naiboğlu, 2016). A camera trap study in the neighboring forests in Zonguldak province did not report any lynx record (Çoğal, 2016). Another camera trap study in Ankara and Bolu reported 147 lynx captures and when adjusted with active camera trap nights, the capture rate was 1.46 (Akbaba, 2016). In our study, with only four camera trap records, it is questionable whether the lynx has dispersed into the area for a short time or is breeding in the area. The photographs were not clear enough to determine whether these were the same individual or different individuals. Intensive camera trapping combined with sign surveys and genetic sampling in this part of the study area may help determine if there is a breeding population in the area. Similarly, the European hare, which is one of the main preys of the European lynx, was detected only five times in total. Can (2008) reported only one record from Yaylacık Research Forest, which is a part of Yenice Forest. European lynx presence was found to be significantly related with European hare presence in the western Black Sea Region, though not with roe deer. Also, the daily activity overlap of European lynx with European hare was higher (Δ_4 = 0.81) than that with roe deer (Δ_4= 0.62) (Soyumert et al., 2019). In Ankara-Bolu study where lynx capture rate was 1.46, capture rate of European hare was 10.58 (Akbaba, 2016). Mengüllüoğlu (2019) reported that the main prey of lynx was European hare based on fecal analysis in Ankara, Antalya and Artvin populations. Considering the results of these studies, the low number of European lynx in our study area could be related to low number of European hare in the area.

Capture rates of the targeted mammals were compared based on year, camera trap location, and area status to provide an overall picture. Although we can make rough inferences, capture frequencies or capture rates do not indicate the abundance of species in the study area. For example, the capture rate of the golden jackal was considerably higher in 2016. However, this does not necessarily mean that the relative abundance of the species was higher that year. The number of captures of this species seems to

have inflated at three cameras with 46, 56, and 79 captures during the survey period. This may be due to the location of the camera traps, which may be close to their dens or on their frequently used paths, and these may be the same individuals. Similarly, the capture rate of the red fox was prominently higher in non-protected areas compared to YWLDA possibly due to three camera trap location where the species was recorded 36, 37, and 39 times. No statistical analysis was carried out in comparing these capture rates as several factors including detection probabilities, and other environmental factors such as elevation, slope, land cover, etc. can influence these rates. These variables are examined in Chapter 4 at the community level for the targeted species and in Chapter 2 for brown bears in particular.

However, a pattern can be observed from the capture rates of predator and prey species being opposite on forest paths and roads. Carnivores used roads more frequently than paths in the forest. On the other hand, ungulates used forest paths more frequently compared to roads. Wild boar was an exception to this pattern with similar capture rates for both locations. Predator avoidance, human activity, and related perceived risk of hunting, lack of forest cover, and food resources may influence prey species' avoidance of roads. Especially canid capture rates on roads were prominently higher compared to forest paths. Canid's preference for roads was reported elsewhere (Raiter, Hobbs, Possingham, Valentine, & Prober, 2018; Kautz et al., 2021). Carnivores move faster and use energy more efficiently and navigate easier when they use roads (Dickie, McNay, Sutherland, Cody, & Avgar, 2019). Roads also increase the hunting success of predator species (James, & Stuart-Smith, 2000). Wolves can travel three times faster on the roads compared to the natural forest. This increases net daily movement rates, and hence their search rate (Dickie, Serrouya, McNay, & Boutin, 2016). Carnivores, in turn, avoid roads during the day when human activity is highest (Kautz et al., 2021,). This temporal avoidance can also turn into spatial avoidance in case of high human activities, such as forestry work that span longer periods (Keten, 2019; Naiboğlu, 2016). A study in Bolu and Düzce, northwestern Anatolia, showed that wolves avoided the roads with heavy traffic and forestry activities (Tokmak, & Ambarlı, 2018). Similarly, another study in Kastamonu, northwestern Anatolia, reported that wolves and other large mammals avoided areas with high road densities and settlements (Ugış, 2023). As a result, when present in the area, predators seem to prefer roads to forest paths. Our results are in line with another camera trap study that was carried out in northwestern Turkey, except for red deer preferring paths and badgers preferring roads in our study area (Soyumert, 2010).

Capture rates of medium and large mammals between Yenice WLDA and surrounding forest were also compared to make a rough inference. As mentioned before, WLDAs are areas where game and wild animals are protected, game animals are introduced, and measures are taken to improve their habitats. Targeted protected species for Yenice WLDA are brown bear, roe deer and red deer. Roe deer and red deer capture rates were higher in YWLDA (7.3 vs 4.8 and 3.9 vs 1.9, respectively) and slightly lower for brown bear (3.6 vs. 5). In this respect, YWLDA seems to have achieved its target for two species. However, reliable estimates of abundance are necessary to support this conclusion. When compared based on the existence of medium and large mammal species, YWLDA and the surrounding forest did not show any differences. A study in neighboring Kartdağ and Sökü WDLAs (Soyumert, 2010), did not show higher capture rates regarding the target protected species compared to non-protected areas. On contrary, red deer and wolf capture rates were higher in unprotected Kurtgirmez area compared to Sökü WLDA.

A previous study by Can (2008) in Yaylacık Research Forest, a part of Yenice Forest, reported the camera trap capture rates of marten (0.66 vs 6.58 in our study), wolf (0.75 vs. 2.18 in our study), brown bear (1.08 vs 4.25 in our study), wild cat (1.75 vs 5.77 in our study), roe deer (2.83 vs. 6.14 in our study), red fox (2.83 vs. 8.15 in our study) and wild boar (12.66 vs. 4.48 in our study) as target species. In addition to these target species, one jackal, four badgers, and one brown hare were also recorded in Yaylacık Research Forest. Camera trap studies carried out near our study area, in northwestern Anatolia has reported different patterns regarding capture frequencies. A previous study by Soyumert (2010) reported that wild boar and roe deer were the most frequently captured species by camera traps, and golden jackal and European badger were the least frequently captured species in seven study areas in Bartın, Kastamonu and Karabük. In Ballıdağ, Kastamonu, medium and large mammals with the highest capture rates were roe deer, hare, and fox while badger had the lowest capture rates. In Kurtgirmez, Kastamonu, the species with the highest capture rates were roe deer, wolf, and fox and the lowest was red deer (Ugış, 2023). In Sarıçam Forest Management Chiefdom, Düzce, roe deer, wild boar, red fox and brown bear were the most frequently captured species (Naiboğlu, 2016). In Düzce province, the species with highest capture frequencies were golden jackal, marten and red fox in 33 different segments and these frequencies varied between the segments (Algan, 2019).

These findings were compared to provide an overview of the capture rates of earlier research in the same biome, in the neighboring areas, however as mentioned previously, detection probabilities vary in

time, space, between camera trap settings, and species. The capture rates are not a proxy of abundance and differences do not necessarily reflect a difference in the abundance of these species. Although these areas are a part of the northwestern Anatolia forests, capture rates varied among the species in different areas and also within our study area. In order to make more reliable estimates of abundance or habitat preferences of these species, more advanced analyses are needed to be carried out adjusting for the factors affecting detection and taking into account the variables that may have influenced the occurrence. In this study, an umbrella species. the brown bear, was selected for further analysis. Additionally, community level analyses were carried out to provide a broader understanding of the medium and large mammal community in Yenice Forests.

2. HABITAT USE OF THE BROWN BEAR IN YENİCE FOREST

2.1. Introduction

Various forms of human activity and global climate change have been described as major factors that increase the likelihood of extinction in the wild (Parmesan & Yohe, 2003). Research on species distribution patterns, biodiversity, and defining the interaction between animals and their environment has emerged as a major area of interest in ecology and conservation due to these pressures and the consequent changes in populations and communities (Brncic, Amarasekaran, McKenna, Mundry, & Kühl, 2015). Designing appropriate management plans and effective conservation measures, such as monitoring populations (Keane et al., 2012), prioritizing key areas (Wilson, McBride, Bode, & Possingham, 2006), evaluating the efficiency of protected area networks (Bruner, Gullison, Rice, & Da Fonseca, 2001), and determining the efficacy of conservation interventions (Tranquilli et al., 2012), requires information on species distributions, species-habitat relationships, and biodiversity patterns.

Occupancy modeling is a valuable method for extensive monitoring programs, to analyze species distributions, to identify habitat selection, meta-population dynamics, species interactions, and species richness. It provides valuable information when demographic or abundance estimates are not feasible (Bailey, Simons, & Pollock, 2004). It is also less labor-intensive and cheaper, and larger areas can be studied. In addition, it is more reliable than traditional methods because traditional occurrence estimations, such as logistic regression, assume perfect detection of the organism, and even moderate amounts of false absences can bias parameter estimates (Rota, Fletcher Jr., Dorazio, & Betts, 2009). Line transect survey or camera trapping survey data with presence and absence information can be used for occupancy modeling. Occupancy modeling proposed by Mackenzie et al. (2002) accounts for imperfect detection, which is estimated from patterns of presence-absence data from multiple surveys at each site. This model requires independent replicate surveys to have been carried out at several sampling sites. The model assumptions are as follows:

1. Site closure assumption: It is assumed that the occupancy status of a site does not change during the sampling period. It remains closed to immigration, emigration, births, and deaths during the period of the study.

2. Spatial independence: It is assumed that the presence or absence of a species at one site is independent of the presence or absence at the other sites.
3. Constant probability of occurrence: The model assumes that the probability of occurrence of a species is constant across the sites. Incorporating covariates that might influence occupancy probabilities adjusts for heterogeneity in occupancy probabilities.
4. Homogeneous detection probability: This assumption states that species has an equal chance of being detected across the sites. Incorporating covariates that might influence detection probabilities adjusts for heterogeneity in detection probabilities.
5. No false positives: The model assumes that there will be no detections if a species is not present at a site.

The site closure assumption is challenging to fulfill when studying wild populations due to the absence of clear borders. To address this, researchers employ 'temporal closure' by conducting surveys during short periods, so immigration or emigration is not expected. Also, the spatial independence assumption might be compromised if camera spacing is smaller than the species home range size. This is particularly problematic when studying animals with extensive home ranges such as brown bears, as meeting this assumption becomes difficult. To avoid reduced detection probability and gather information on habitat variables relevant to their presence, camera traps cannot be spaced too far apart. MacKenzie et al. (2005) propose that this assumption can be relaxed if the species' movements between sites are random. In this case, model results can be interpreted based on the proportion of sites used by the species rather than the proportion of sites occupied.

2.1.1. Brown Bear

The brown bear (*Ursus arctos*) has the largest worldwide distribution among other bear species and is found in North America, Europe, and Asia. However, habitat loss, habitat fragmentation and destruction, and poaching have reduced their historical distributions (Swenson, Taberlet, & Bellemain, 2011). Its European populations consist of small and fragmented populations in Western Europe and the Mediterranean region, and they are critically endangered (McLellan, Proctor, Huber, & Michel, 2016).

Habitat fragmentation, habitat loss, and increasing human impact, such as poaching, have reduced populations of large carnivores globally (Woodroffe, 2000), and the brown bear as the largest carnivore in Europe has been extirpated from human-dominated landscapes (Zedrosser, Dahle, Swenson, & Gerstl,

2001; Kopatz et al., 2012). Remaining and disconnected small populations heavily depend on conservation efforts for their survival (Swenson, Gerstl, Dahle, & Zedrosser, 2000). After becoming legally protected by several international agreements such as the Bern Convention, CITES (Convention on International Trade in Endangered Species of the Wild Fauna and Flora), and the EC Habitats Directive, bear populations started to increase in some areas (Zedrosser, Steyaert, Gossow, & Swenson, 2011). However, urbanization, recreation and road development, resource extraction, deforestation, and intensive agriculture are still threatening European brown bear populations (Breitenmoser, 1998; Zedrosser et al., 2001).

In Turkey, although having wide historical distributions, their numbers have declined, and their distributions have shrunk in parallel to available suitable habitats. According to the IUCN Red List of Threatened Species, they are listed as endangered (EN) and critically endangered (CR) in most of their regional distributions, and similarly, local populations in Turkey are considered vulnerable (VU) (e.g., Yenice Forest), endangered (EN) (western Anatolia and Taurus Mountains), and critically endangered (CR) (Datça) (McLellan, Proctor, Huber, & Michel, 2017).

2.1.2. Habitat use of the brown bear

Habitat use of brown bears highly varies across its range. Several studies show that habitat use of the brown bear varies seasonally and between sexes and age groups. Various techniques were used in these studies, from radio or GPS-GSM collars to camera traps and direct observations of the animals or their tracks.

2.1.2.1. Season. In the Yellow Stone National Park brown bears avoid human structures and main roads, especially during summer and autumn (Mattson, Knight, & Blanchard, 1987). In Denali National Park, an observational study reported that brown bear habitat use changed seasonally with the phenological development of different berry species in the area (Stelmock, & Dean, 1986). Another study in southwest Alaska reported that spatial distribution of food resources is more influential on habitat use of brown bears than temporal availability (Mangipane et al., 2018). 56 grizzly bears collared between 1979 and 1995 in southeast British Columbia also showed seasonal variation in habitat use. According to the MANOVA model that used sex, age class, and season as factors, only season had a significant effect on grizzly habitat use ($p < 0.001$). There were two groups of bears, a group was resident in the mountains

and a group was migrating between elevations seasonally. In summer, both groups selected areas burnt 50-70 years ago. In autumn the most popular areas were riparian habitats, followed by forest and open forest areas (McLellan, & Hovey, 2001).

2.1.2.2. Demographics. In Alberta, males and females use different aspects of the mountains in spring. In late summer, areas used by males and females overlap mostly, and three habitat variables differ between sexes: elevation, aspect, and slope. In autumn, the overlap of used areas reduces, and two variables differ between sexes: forest edge and elevation (Wielgus, & Bunnel, 1994). Another study with radio collars (1979-1985) found that adult males avoid the areas within 100 m of roads, but females and cubs avoid these areas less frequently. As adult males may tend to kill cubs and yearlings, habitats near roads might be safer for sub-adult brown bears (McLellan, & Shackleton, 1988). Sexual differences in habitat use are usually interpreted as female avoidance of males to protect their cubs (Mattson et al., 1987; McLellan, & Shackleton, 1988). In Yellowstone National Park, dominant males stay in the core areas pushing females and sub-adult males closer to human structures (Mattson et al., 1987). In Denali National Park, based on scans of 34 fixed plots suggested grizzly bear habitat use patterns changed based on family age (age of the cubs with the mother), years, and seasons (Darling, 1987).

2.1.2.3. Land cover. A study based on GPS-GSM collars reported that bears select different habitats for foraging and resting (Moe, Kindberg, Jansson, & Swenson, 2007). According to this study, coniferous habitat is mostly selected during resting and avoided during foraging. In a study based on radio telemetry and bear signs (1985-1988) Cantabrian brown bear bears selected beech (*Fagus sylvatica*), and oak forests (*Quercus* sp.) in greater proportion to their availability. In general, they avoided roads and settlements. The most popular habitats they used were complexes of deciduous forests, shrubs, and grassland (1100 -1400 m elevation) (Clevenger, Purroy, & Campos, 1997). Cantabrian and Pyrenean brown bears' habitat selection was similar at the landscape level. They both preferred areas with forest connectivity, hard mast-producing forests with low levels of human disturbance (Martin et al., 2012). Mateo Sánchez, Cushma and Saura (2013) found landscape composition and human disturbance were the most influential factors in brown bear habitat use in Spain and habitat selection was driven by environmental factors at different scales. The variables in the habitat suitability model were the most predictive at different scales, e.g., "building density" at 16 km scale, "canopy closure" at 1 km scale; "percentage of the landscape covered by mixed forest" at 32 km scale; "percentage of the landscape covered by deciduous forest" at 8 km scale. Ecological niche factor analysis of brown bears in Greece

based on GPS and radio telemetry data combined with sign surveys (1997-2002) show that brown bears in northern Pindos mostly prefer agricultural areas and forest edges (Mertzanis et al., 2008). In the central Apennine Mountains of Italy, the brown bear occurrence was found to be positively related to average elevation and deciduous forest, and negatively related to vineyard-olive groves and shrublands (Posillico, Meriggi, Pagnin, Lovari, & Russo, 2004). Agricultural land-cover showed a positive relationship with brown bear occurrence in Lahaul Valley, India (Sharief et al., 2020). Brown bear habitat use is also influenced by landscape heterogeneity: brown bears occupy larger areas when landscape homogeneity is higher, possibly in search of a wider variety of food sources (Mangipane et al., 2018). A study at Zagros Mountains reported that distance to village, distance to road, slope, and elevation were the most important variables that influence brown bear habitat use (Almasieh, Rouhi, & Kaboodvandpour, 2019). Another study at the Zagros Mountains reported that distance to protected areas, the maximum temperature in the warmest month, the roughness of the landscape, forest density, and annual precipitation were the most important factors that influence brown bear distributions (Ashrafzadeh, Shahbazinasab, Mohammadi, & Penteriani, 2023).

2.1.2.4. Human disturbance. Human density had a negative influence on brown bear occupancy in the Pyrenees, France (Piédallu et al., 2016). Brown bears prefer areas far from intensive human activities and high human densities (Martin et al., 2010; Long, Donovan, MacKay, Zielinski, & Buzas, 2010). In some cases, habituation to humans (Wheat, & Wilmers, 2016) and intraspecific interactions such as males dominating core habitats and females and cubs being pushed to human-intensive areas (Steyaert et al., 2016) reverse this avoidance pattern. In the Italian Alps, brown bear detectability increased with distance to the settlements, but no significant influence was found on the occurrence of the species (Oberosler, Groff, Iemma, Pedrini, & Rovero, 2017). A study in Bieszczady mountains, Poland reported that human avoidance of brown bears (estimated using distance to road, village, and forest edge) was only evident in spring which could also be a result of the diet choice of brown bears in that region in spring (Frąckowiak, Theuerkauf, Pirga, & Gula, 2014). It is important to note that the specific effects of human disturbance on brown bear occupancy can vary depending on the context, intensity, and duration of the disturbance, as well as the characteristics of the bear population and habitat.

In Turkey brown bears are mainly distributed in the Black Sea Region and Eastern Anatolia, and in low numbers in the Mediterranean region (Ambarlı, Ertürk, & Soyumert, 2016). A study in Yusufeli, Artvin (Ambarlı, 2012) reported that brown bears mostly use croplands, in agricultural fields. They also

used evergreen forests, mixed forests, shrublands, and open spaces in forests as secondary habitats. A study carried out in Akseki, Antalya reported that based on the Maxent method, environmental variables that influence brown bear occurrence were ruggedness index, elevation, slope, vegetation index, roughness index, and solar illumination index (Suel, 2019). Brown bears showed seasonal variation in using northern and southern aspects and the species was recorded in all elevation ranges studied in northeastern Anatolia (Soyumert, 2010). In Zonguldak province, brown bear capture rates increased with increasing distance to settlement, distance to roads, and elevation (Çoğal, 2016). Distance to road was not influential on camera trap capture rates of brown bears and they mostly preferred culverts (menfez in Turkish), compared to viaduct and underpasses when crossing the Ankara-Bolu highway. Collared brown bears in this study area preferred coniferous forest and steppe-coniferous or steppe-shrub vegetations (Sayar, 2022). A study from north-eastern Turkey showed differences in habitat use between sedentary and roaming brown bears. During the sedentary phase, brown bear preferred forested areas with higher slopes and in close proximity to roads and during the roaming phase they preferred forested areas, and they were less responsive to the other environmental factors tested (Cozzi et al., 2016). A previous study carried out in Yaylacık Research Forest, a 50 km^2 part of Yenice Forests, reported the presence and diel activity patterns of brown bears, but provided no analysis on the habitat use of the species (Can, 2008).

2.1.3. Aims of this Chapter

The brown bear is a keystone and umbrella species, and successful management of its population would also help the conservation of the species inhabiting the same habitat. This translates into very diverse plant and animal species considering their home ranges (e.g. 20 km^2- 315 km^2, Sayar, 2022). They also have important functions in the ecosystem such as serving as seed dispersers (Garcia-Rodriguez et al. 2021). For instance, the germination rates of fruit seeds (*Malus sylvestris* and *Pyrus elaeagnifolia)* from bear feces were found higher than the seeds collected from the same sites (Tavşanoğlu, Kazancı, Soyumert, Ertürk, & Değirmenci, 2021). Besides, brown bear diggings in search of roots were found to be more productive and richer in ammonium and nitrate. Lilies (*Erythronium grandiflorum*) naturally grown on these diggings had higher nitrogen and water-soluble carbohydrate and produced twice more seeds compared to the ones in the control areas (Tardiff, & Standford, 1998). The wounds on the trees made by brown bears were shown to provide sites for insects and birds to feed and breed (Zyśk-Gorczyńska, Jakubiec, & Wuczyński, 2015). The brown bear is an omnivorous species

and a top predator. They help regulate prey populations in their ranges and their absence can lead to an increase in prey populations and hence an overexploitation of primary resources (Palazon, 2017). However, their need for large areas and low reproduction rates make their populations vulnerable. Therefore, understanding the ecological needs of the species is important to make evidence-based decisions for the management of the areas in their distributional range. The brown bear is also included among the target species to be protected in Yenice WLDA. Considering its importance in the ecosystem and the need for information on brown bear ecology in this area we focused on this species specifically.

In this chapter, we aimed to estimate the environmental and human-related factors that influence the habitat use of brown bears in Yenice WLDA and surrounding unprotected forests. We also tested if the WLDA status has an influence on brown bear distribution. Our observation in the study area showed that these two areas are not different from each other regarding human activities, spatial structure, and vegetation cover. Therefore, our hypothesis is that WLDA status does not have an influence on the use of area by brown bears.

2.2. Methods

We used occupancy modeling which was introduced by MacKenzie et al. (2002). According to this model, multiple site visits are required to estimate the probability of detection and occupancy. The presence of a species is coded as 1 and indicates the occurrence of that species at a site, and the species is detected by an observer/tool. The absence of a species is coded as 0 and indicates that the species does not occur at that site OR the species was not detected by an observer/tool. This means that non-detection of the species does not necessarily mean it is absent. Probabilities of detection and occupancy are estimated from the multiple presence-absence data using logistic regression with a logit link function. The logistic regression equation for the occupancy model can be represented as:

$$\text{logit}(\Psi) = \beta 0 + \beta 1x1 + \beta 2x2 + ... + \beta nxn \quad (2.1)$$

where, Ψ denotes the probability a site is occupied by the species, $\beta 0$ denotes the intercept, and $\beta 1$, $\beta 2$,…, βn denote the coefficients of the site-level covariates (x1, x2, ..., xn) that influence occupancy.

The logistic regression equation for the detection model can be represented as:

$$\text{logit}(p) = \alpha 0 + \alpha 1z1 + \alpha 2z2 + ... + \alpha mzm \quad (2.2)$$

where, *p* denotes the probability of detecting the species in site, α0 denotes the intercept, and α1, α2, ..., αm denote the coefficients of the site-level covariates (z1, z2, ..., zm) that influence detection.

2.2.1. Data Organization

The data organization for occupancy analysis from the overall camera trap data included the following steps:

- Bear occurrence data were extracted from the overall camera trap data. The detection history table was prepared by using '1' for detection, and '0' for non-detection of a bear.
- A data matrix showing active camera trap days was prepared. Camera traps that worked for less than five days per month were removed from the table.
- The date matrix and detection history tables were combined to show bear detections only within the active camera trap periods. Non-active days were coded as 'NA'.
- Data were collated to form five-day survey periods, as camera traps that operated for less than five days in a survey period would have had a smaller chance of detecting bears. Survey periods of one or two days were categorized as NA, and survey periods of three or four days were weighed with 0.6 and 0.8, respectively.

2.2.2. Variables

We assessed the importance of 21 anthropogenic and environmental covariates for brown bear occupancy and/or detection (Table 2.1). These include land use variables, spatial variables, and vehicle capture rates at the camera trap sites. We calculated the environmental and spatial variables at five different scales since ecological patterns are scale-dependent and scale is important to determine species-habitat relationships (Niedballa, Sollmann, Mohamed, Bender, & Wilting, 2015; Mateo Sánchez, Cushman, & Saura, 2013).

Table 2.1. Site and survey covariates evaluated in the study.

Covariate	Value	Calculated based on:
Slope	Mean	Circular area with radii 1, 2, 4, 8, 16 km
Elevation	Mean	Circular area with radii 1, 2, 4, 8, 16 km
Agriculture	Percent land cover	Circular area with radii 1, 2, 4, 8, 16 km
Forest	Percent land cover	Circular area with radii 1, 2, 4, 8, 16 km
Scrub and herbaceous vegetation	Percent land cover	Circular area with radii 1, 2, 4, 8, 16 km
Open spaces	Percent land cover	Circular area with radii 1, 2, 4, 8, 16 km
Urban	Percent land cover	Circular area with radii 1, 2, 4, 8, 16 km
Contagion (CONTAG)	Landscape metrics	Circular area with radii 1, 2, 4, 8, 16 km
Contrast weighted edge density (CWED)	Landscape metrics	Circular area with radii 1, 2, 4, 8, 16 km
Patch Density (PD)	Landscape metrics	Circular area with radii 1, 2, 4, 8, 16 km
Shannon Diversity Index (SHDI)	Landscape metrics	Circular area with radii 1, 2, 4, 8, 16 km
Vehicle capture rate	ratio	Number of vehicles captured/trap nights x 100
Distance to river	meters	Euclidian distance to the nearest river in meters
Distance to road	meters	Euclidian distance to the nearest road in meters
Distance to settlement	meters	Euclidian distance to the nearest settlement in meters
Distance to WLDA border	meters	Euclidian distance to the WLDA border in meters
WLDA/Non-WLDA	1 and 0	1 and 0, within WLDA or outside WLDA
Road/Path	1 and 0	Camera trap location, on the road or path
Forest type (3 covariates) **a. Broad leaf** **b. Mixed** **c. Coniferous**	Percent land cover	Circular area with radii 1 and 2 km.

Contagion (CONTAG) measures the extent to which patch types are aggregated or clumped. Contrast weighted edge density (CWED) calculates the edge density between patch types. Patch Density (PD) is a function of the number of patches and total landscape area. Shannon Diversity Index (SHDI) calculates the diversity of patches in the landscape.

The elevation raster was downloaded from Earth Data using QGIS Desktop 3.6.3 with Grass 7.6.1 (QGIS.org, 2021) with the SRTM downloader plug-in (www.urs.earthdata.nasa.gov). The slope raster was prepared based on the elevation raster using the 'raster terrain analysis' function. The Corine 2018 layer data were downloaded from Copernicus (https://land.copernicus.eu/pan-european/corine-land-cover/clc2018). Elevation and slope mean values for different scales were calculated using QGIS Desktop 3.6.3 with Grass 7.6.1, r.neighbours function.

Agriculture, forest, scrub, and herbaceous vegetation, open spaces, and urban and forest type data were extracted from the Corine 2018 layer using QGIS Desktop 3.6.3, with Grass 7.6.1, with the r.reclass function. Five different scales (circular windows with radii of 1, 2, 4, 8, and 16 km) were calculated for each covariate, as they were found to be predictive for different covariates (Mateo Sanchez et al., 2013). This approach involves assigning the value within a raster pixel by calculating the mean of the surrounding pixels depending on the scale selected. Agriculture, forest, scrub, and herbaceous vegetation, open spaces, urban, forest type, contagion (CONTAG), contrast weighted edge density (CWED), Patch Density (PD), Shannon Diversity Index (SHDI), and class metrics of percentage of landscape (PLAND) values for different scales were calculated using Fragstats v. 4.2.1 (McGarigal et al., 2012).

Distance to river, distance to road, distance to settlement, and distance to WLDA border variables were calculated using the QGIS Desktop 3.6.3 Grass 7.6.1 using the NNJoin plug-in. For distance to WLDA variable, negative values were assigned for the camera traps within WLDA. All distance variables were standardized to z-scores. The vehicle capture rate was calculated as the percentage of camera trap capture based on the total number of trap nights for each camera trap (CR = [number of vehicles captured/trap nights] x 100). The elevation and slope data were standardized to z-scores. For edge density calculations, a matrix design (Table 2.2) was used with different weighting scores for edges between different habitat types, in order to take into account the transition between habitat types not as sharp across all types. Covariates with highly skewed distributions were transformed into categorical variables.

Table 2.2. Edge matrix for land use variables.

	Urban	**Agriculture**	**Open Spaces**	**Scrub&herbaceous vegetation**	**Forest**	**Water bodies**
Urban	0	0	0	0	0	0
Agriculture	0	0	0.25	0.5	1	1
Open Spaces	0	0.25	0	0.5	1	1
Scrub and herbaceous vegetation	0	0.5	0.5	0	0.75	1
Forest	0	1	1	0.75	0	1
Water bodies	0	1	1	1	1	0

2.2.3. Occupancy Modelling

Occupancy models were run using R v. 3.5.2 (R Core Team, 2018) 'unmarked' package that uses a maximum likelihood-based approach. Royle-Nichols (RN) method with maximum likelihood approach was used for all models. Unlike the single-season occupancy models of MacKenzie et al. (2002), RN models take the relation between detection probability and abundance into account (Royle and Nichols 2003). These models allow estimation of abundance based on the presence-absence data without the need to mark individuals in a population. This method relates the detection probability of an animal to the number of animals detected at each site. Variation in the abundance of an animal can lead to heterogeneity in the detection probability. In this method, a heterogeneous detection probability model is used to estimate the abundance distributions. Occupancy probability, (psi, "the probability that at least one individual is available for detection at the site") and detection probability (p) can be derived from an RN model, where N (relative abundance) and r (probability of detecting an individual) are estimated.

The 'year' covariate was used to evaluate the effect of the year on the abundance and detection probabilities. A two-stage model-building approach was used to reduce the number of possible models. The most suitable model for detection probability was developed, keeping abundance constant and running all possible combinations for detection probability.

First, the factors that might affect the detection probability were evaluated. Traffic, camera trap location (road/path), and year were used in the model, with N held constant. Likelihood ratio testing was used to select between the top two models. Using fixed r(.), different scales of covariates of N(.) were tested against the null model, to identify the best-performing scale of each covariate for their inclusion into further analysis. Pearson's correlations were calculated to exclude the correlated variables. A global model was run using the variables that proved to contribute to the model. The goodness of fit (GoF) of the global model was tested using Pearson's chi-squared and Freeman-Tukey chi-squared tests. The RN models were run using all possible combinations of covariates in the global model. Model averaging was performed by selecting the top models with <2.5 ΔAIC weight and the average of all beta coefficients was calculated.

2.3. Results

Null RN models (occuRN) performed better than null standard occupancy (occu) models (for example AIC 789 vs. 796, for Royle-Nichols and standard models, respectively). The year covariate was used to evaluate the effect of the year on the abundance and detection probabilities. There were no significant differences between the 2015 and 2016 survey periods, and the data from the two years were combined for further analysis. Factors that might affect the detection probability were evaluated. Vehicle capture rate (VehicleCR), camera trap location (road/path), and year were used in the model, holding N constant (Table 2.3).

Table 2.3. Test for covariates that might affect detection probability based on Royle-Nichols abundance estimation models.

	#Par	AIC	ΔAIC	AIC wt	Cmlt wt
N(.)r(Cam_location)	3.00	781.32	0.00	0.32	0.32
N(.)r(Cam_location+year)	4.00	782.33	1.00	0.20	0.52
N(.)r(VehicleCR)	6.00	782.39	1.06	0.19	0.71
N(.)r(VehicleCR+year)	7.00	782.84	1.51	0.15	0.86
N(.)r(VehicleCR+Cam_location)	7.00	784.30	2.98	0.07	0.93
N(.)r(VehicleCR+Cam_location+year)	8.00	784.78	3.46	0.06	0.99
N(.)r(year)	3.00	788.93	7.60	0.01	0.99
N(.)r(.)	2.00	789.52	8.19	0.01	1.00

#Par: Number of parameters, AIC: Akaike Information Criterion score, ΔAIC: The difference in AIC score when compared to the best model, AICwt: AIC weight, relative likelihood of the model, Cmltwt: Cumulative weight, The sum of the AIC weights.

The camera trap location was the most informative covariate for bear detection (r) based on AIC values. Therefore, this variable was used for further analyses. Using camera trap location as a covariate of r(.), different scales of covariates of N(.) were tested against the null model to identify the best-performing scale of each covariate for their inclusion in further analyses. The covariate scales that performed better than the null model are shown in Table 2.4.

Table 2.4. Royle-Nichols abundance estimation model list of all the selected covariates

	#Par	AIC	ΔAIC	AIC wt	Cmlt wt
N(Pland_OpenSpaces _16km)r(CameraLocation)	4	771.71	0.00	0.71	0.71
N(Slope_16km)r(CameraLocation)	4	774.99	3.28	0.14	0.85
N(Pland_Water_4km)r(CameraLocation)	4	776.85	5.14	0.05	0.90
N(Elevation_16km)r(CameraLocation)	4	777.45	5.75	0.04	0.94
N(Pland_Forest_8km)r(CameraLocation)	4	778.22	6.51	0.03	0.97
N(cwed_8km)r(CameraLocation)	4	779.89	8.18	0.01	0.98
N(pd_4km)r(CameraLocation)	4	780.66	8.95	0.01	0.99
N(contag_8km)r(CameraLocation)	4	780.78	9.07	0.01	0.99
N(.)r(CameraLocation)	3	781.32	9.62	0.00	1.00

Pland: Percentage of landscape, CWED: Contrast weighted edge density, contag: contagion, #Par: Number of parameters, AIC: Akaike Information Criterion score, ΔAIC: The difference in AIC score when compared to the best model, AICwt: AIC weight, relative likelihood of the model, Cmltwt: Cumulative weight, The sum of the AIC weights.

Based on the likelihood ratio test, pd_4km (p = 0.10), contag_8km (p = 0.11), and cwed_8km (p= 0.06) were removed from further analyses. The other covariates that were tested, such as forest type, vehicle capture rate, distance to settlement etc. (Table 2.1) did not contribute to the model. Univariate Royle-Nichols abundance estimation models based on informative covariates are given in APPENDIX A. Table A1 – A5.

The global model with the final variables showed that brown bear relative abundance increased with increasing mean slope and mean elevation within 16 km radius, and 1-25 percent land with water bodies within 4 km radius. Relative abundance of brown bear decreased with increasing percent of land (from 0 to 1-25%) with open spaces within 16 km radius, and increasing percent land (from 51-75% to 76-100%) of forest within 8 km radius. However, the influences of OpenSpace_16 km and forest_8 km were highly variable with relatively high standard errors. Detection probability decreased at the forest paths (Table 2.5). Response curves of the continuous variables elevation 16 km and slope 16 km are given in Figure 2.1.

Table 2.5. Global Royle-Nichols abundance estimation model with final covariates.

Abundance:	**Estimate**	**SE**	**z**	**P(>|z|)**
Intercept	0.52	0.45	1.17	0.24
slope_16km	0.30	0.15	2.04	0.04
elev_16km	0.44	0.20	2.18	0.03
1-25%_OpenSpace_16km	-0.04	0.36	-0.11	0.92
76-100%_forest_8km	-0.62	0.41	-1.49	0.14
1-25%_water_4km	0.69	0.40	1.74	0.08
Detection:	**Estimate**	**SE**	**z**	**P(>|z|)**
Intercept	-1.42	0.23	-6.08	<0.01
Cam_place(path)	-0.84	0.26	-3.17	<0.01
AIC: 767.77				

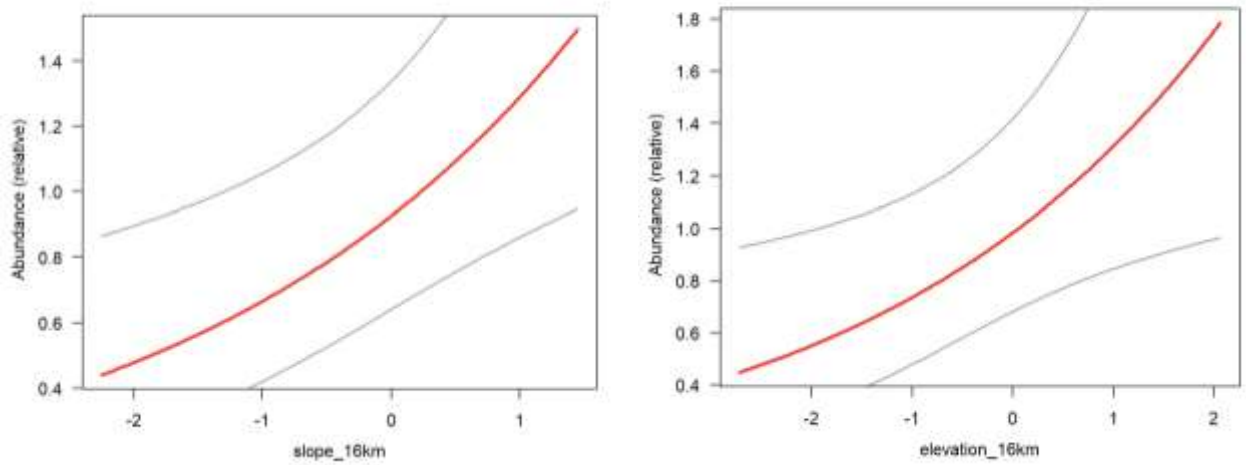


Figure 2.1. Response curves for Slope 16km and Elevation 16km.

The goodness of fit (GoF) of the global model was tested based on the sum of squared errors, Pearson's chi-squared test, and the Freeman-Tukey chi-squared test. For all tests, the p-value was higher than 0.05, indicating a good fit (Table 2.6).

Table 2.6. Goodness of fit test of the most complex model.

	t0	**mean**	**SE**	**P**
SSE	120	-0.39	9.52	0.53
Chisq	859	-58.53	39.59	0.95
freemanTukey	173	1.77	11.61	0.45

The RN models were run using all possible combinations of covariates (Appendix B). Model averaging was performed by selecting the top models with <2.5 ΔAIC weight and the average of all beta coefficients was calculated (Figure 2.2).

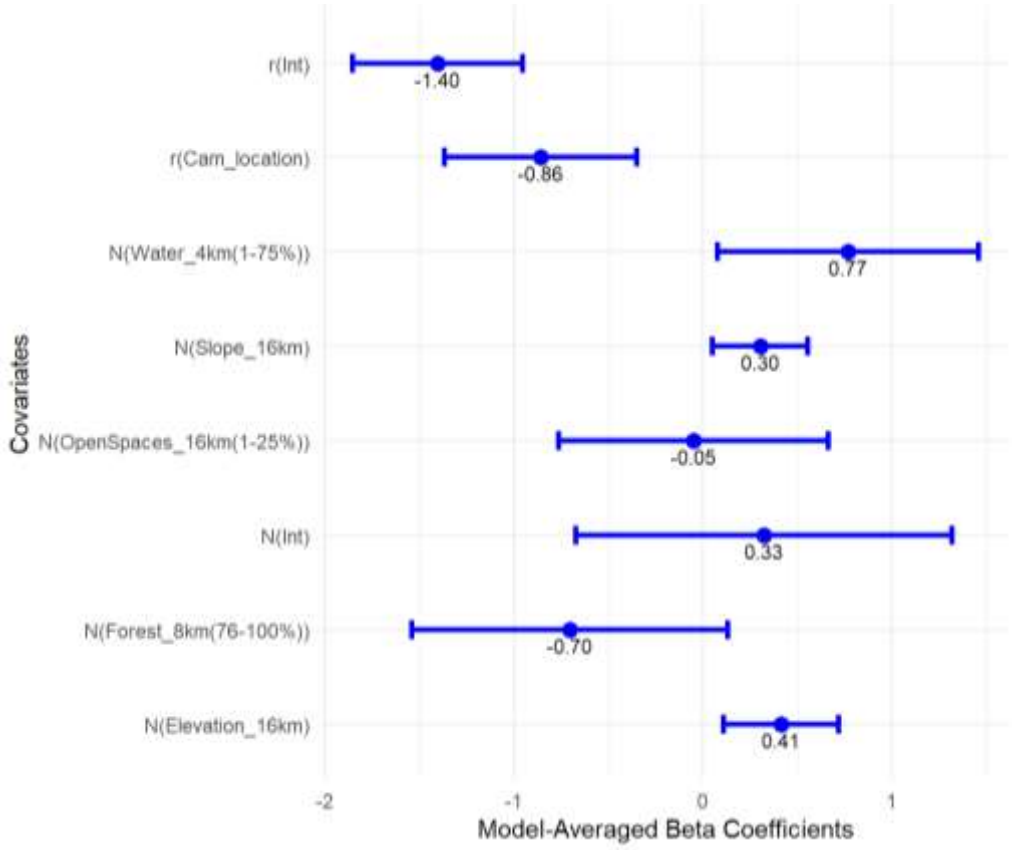


Figure 2.2. Beta coefficients of covariates based on model averaging.

2.4. Discussion

We examined the relationship between brown bear habitat use and environmental factors/human disturbance using Royle & Nichols occupancy modeling based on maximum likelihood estimation, taking detection probabilities into account. Open spaces, slope and elevation within 16 km, water resources within 4 km, and forest within 8 km were the most important covariates that have a significant influence on brown bear habitat use in Yenice Forest. Detection probability was significantly influenced by the location of camera traps. These results are consistent with the literature where slope (in Zagros, Almasieh et al., 2019; in Antalya, Suel, 2019; in Eastern Carpathians Štofík, & Saniga, 2012) elevation (in Zagros, Almasieh et al., 2019; in British Columbia, McLellan & Hovey, 2001; in Spain, Clevenger et

al., 1997; in Italy, Posillico et al., 2004, in Antalya, Suel, 2019; Ugış, 2023), forest cover (in British Columbia, McLellan & Hovey, 2001; in Spain, Clevenger et al., 1997 and Martin et al., 2012 and Mateo Sánchez et al. 2013; in Italy, Posillico et al., 2004; Ugış, 2023; Kalleci, 2023), open spaces (in Artvin, Ambarlı, 2012; in British Columbia, McLellan, & Hovey, 2001) and water bodies (Bartzokas et al., 2023; Ugış, 2023; Kalleci, 2023) were found to influence brown bear habitat use and camera trap location has an influence on detection probability of mammal species (Soyumert, 2010; Burton et al., 2012; Srbek-Araujo, & Chiarello, 2013; Boron et al., 2019).

We tested the covariates at six different scales (1km, 2km, 4km, 8km, 16km) and eight covariates had significant effects at different scales, highlighting the importance of scale selection when handling environmental factors, especially for the mammals that have large home ranges. This was consistent with the studies of Mateo Sanchez et al. (2013) suggesting that predictive ability of the environmental variables changes significantly depending on the scale they are measured. They reported that unscaled models were weaker, less discriminant, and led to different predictions of the factors that were important for brown bear habitat use.

Suel (2019) reported that in the western Mediterranean region, brown bears preferred forests with open areas and avoided dense forests. Almasieh et al. (2019) also reported that brown bears preferred the areas between forests and rangelands at high elevations. A combination of forest, shrub, and grassland was the preferred habitat for brown bears in the Cantabrian Mountains (Clevenger, Purroy, & Pelton, 1992). Similarly, collared brown bears near Ankara-Bolu highway mostly used coniferous forests and open areas mixed with coniferous trees and shrubs (Sayar, 2022). Although in our study, open spaces within 16 km and forests within 8 km covariates contributed to the model with high standard deviations, the contribution of these covariates is consistent with these studies. A mix of open spaces and forests can be advantageous for broader foraging opportunities.

In some studies in the neighboring forests, tree type was found to be an important factor in brown bear habitat use. A study in Ilgaz and Gavurdağ WLDAs in Kastamonu reported brown bears preferred broad-leaved and coniferous forests (Kalleci, 2023). According to another study in Kastamonu, in Ballıdağ brown bears mostly preferred coniferous forests, and in Kurtgirmez they mostly preferred broad-leaved forests and degraded tree stands (Ugış, 2023). A study in Bartın (Horasan, 2018) reported that brown bears mostly used the areas with broad-leaved forests with fir and beech trees. In northeastern

Anatolia, Artvin, brown bear was most frequently recorded in broad-leaved forests (Özyanık, 2019). In our study, forest types evaluated in three categories as coniferous, broad-leaved, and mixed forest, did not influence brown bear habitat use. It is important to note that the above-mentioned studies evaluated habitat preference based on presence data only. Therefore, these analyses do not account for detection probabilities. This, and other environmental factors might have contributed to the variations in brown bears' forest type preference between the study areas.

In the neighboring province of Zonguldak, there was a positive relationship between brown bear capture rates and elevation (Çoğal, 2016). In Kastamonu, habitat suitability analysis showed that brown bears preferred the areas with 1000-1500m elevation (Ugış, 2023). In Bartın brown bears were most frequently captured between 1400-1500m (Horasan, 2018), and in Eskişehir the most frequent captures were between 900-1000m elevations (Özay, 2019). Slope did not have an influence on the capture rate of brown bears in several study sites in northwestern Anatolia (Soyumert, 2010) and in Kastamonu (Ugış, 2023; Kalleci, 2023). In our study, elevation and slope were evaluated as continuous variables and not categorized into subgroups. In addition, elevation and slope were not evaluated based on their values at camera trap locations, considering the species use larger areas than the point it was detected. Among the six different scales, slope and elevation within 16km radius was found to significantly contribute to the RN model. Increasing slope and increasing elevation within 16km radius had positive influences on brown bear abundance. It was shown before that brown bear, especially females with cubs prefer areas with high slopes (Zarzo-Arias et al., 2019; Dahle, & Swenson, 2003; Martin et al., 2010). Elevational trends seem to be related to foraging opportunities at preferred elevations (McLellan, & Hovey, 2001) and human settlements at lower elevations (Almasieh et al., 2019; Clevenger et al., 1997).

Unlike the other studies that report the avoidance of human disturbance by brown bears (Piédallu et al., 2019; Preatoni et al., 2005; Kobler, & Adamic, 2000; Çoğal, 2016; Ugış, 2023) covariates related to possible human disturbance such as distance to settlements, percent of urban land, and traffic (vehicles captured by camera traps) did not have an influence on brown bear habitat use in Yenice forest. One reason for this might be that our study area is already far from crowded settlements/urban areas and the bears may not be intimidated by small villages consisting of a few houses. During site visits we gathered information regarding human-bear interactions and some locals reported bears intruding on their gardens and exploiting their crops (mostly beans and fruits). In this case, these small settlements could be perceived as an attraction rather than a threat by brown bears. According to the model, the bears also did

not avoid roads with high traffic (logging trucks) within the forest. In many parts of the forest, these roads are the main paths for large carnivores because of steep slopes and thick understory in the Yenice forest. In this case, temporal partitioning of space use by bears and humans can be observed as reported in Chapter 3 - Diel Activity Patterns of Brown Bears in Yenice Forest. Based on diel activity pattern analysis of our camera trap data, brown bears in general showed a crepuscular activity pattern with peaks at dusk and dawn. The diurnal activity of brown bears was very low and human activities were mostly confined to day hours. The overall activity overlap between humans and brown bears was low.

One of the main aims of the study was to evaluate the influence of YWLDA status on brown bear distribution in Yenice Forest. Several studies reported that protected areas do not provide shelter for protected species. In Kenya, a comparison of mammal population declines between protected areas and surrounding non-protected areas showed trends were similar between the areas, and protected areas were not providing safety for the protected species (Western, Russell, & Cuthill, 2009). Another study on African national parks reported a 59% decline in large mammal populations between 1970 and 2005 (Craigie et al., 2010). Without clear borders and limiting human activities, many protected areas can be considered as 'Paper parks', which are protected on paper but in reality, provide little to no protection. In our study, there was no difference between the areas inside and around (outside) YWLDA regarding brown bear habitat use. Distance to the YWLDA border and YWLDA/non-YWLDA covariates did not contribute to the model. This was not surprising considering the intensive logging activity in YWLDA that is carried on throughout the year. Human activity in the forest as tested with vehicle capture rate did not contribute to the model. On the other hand, logging activities involve foresters living in the forest in sayvans and loud forestry work which can be captured by camera traps. Therefore although vehicle capture rates did not influence the bear occurrence significantly, the factors related to logging activity might have contributed to this similarity between protected and unprotected areas. A study in Düzce (Kahraman, 2019) and another study in Bolu (Naiboğlu, 2016) reported that forestry activities force the mammal species to leave the area for long periods. Sıkdokur et al. (2023) reported that human activity in and around protected areas leads to increasing human-bear conflict and is responsible for 12% of all human-bear incidents in Turkey, and suggested that logging, collecting, livestock grazing, and recreational activities should be limited to reduce or prevent brown bear and human mortality. In addition, illegal hunters with rifles and hunting dogs were captured by several camera traps. Basically, although hunting is not legal in the area, illegal hunting might also exert an influence on this "protected"

population similar to the surrounding unprotected areas. In addition to these factors, other environmental variables might play a role in the similarity of brown bear abundance in two areas.

In our study, since we used camera traps and individual identification of bears was not possible, we cannot comment on the differences in habitat use between the individuals. Studies have shown that there is an inter-individual variation in brown bear habitat preferences, although some main characteristics such as seeking forest cover are common preferences (Cozzi et al., 2016; Ambarlı, 2012). Further studies that account for individuals would contribute to our understanding of the brown bears in Yenice Forest. These may involve invasive techniques such as GPS collaring or noninvasive techniques such as genetic barcoding via scat and hair samples. Also, long-term monitoring of the brown bears in the area using camera traps would be informative on determining population trends in time.

3. DIEL ACTIVITY PATTERNS OF BROWN BEAR

3.1. Introduction

3.1.1. Daily Activity Patterns

Daily activity patterns, the distribution of activity over the course of a day, are a major characteristic of animal behavior with significant implications for a variety of ecological and physiological processes (Vazquez, Rowcliffe, Spoelstra, & Jansen, 2019). Daily activity patterns can be classified as nocturnal, diurnal, crepuscular, or cathemeral (Bennie, Duffy, Inger, & Gaston, 2014). These patterns reflect a complicated balance between foraging, predator avoidance, resting, competition, and social behaviors as a result of adaptation to environmental change throughout the day (Kronfeld-Schor, Bloch, & Schwartz, 2013). Animals can change their activity patterns in response to a variety of environmental conditions, including anthropogenic stress. Therefore, daily activity patterns can be seen as an indicator of environmental stress (Roth, 1983) and can be used to assess the degree of anthropogenic effects on the ecosystem (Vazquez et al., 2019).

Anthropogenic activities that cause changes in daily activity patterns can lead to physiological stress and impair the reproduction and survival of animals (Bourgoin et al., 2011). Human disturbance is defined as "any human activity that changes the contemporaneous behavior or physiology of one or more individuals" (Nisbet, 2000, as cited in Martin et al., 2010). Many wildlife species have similar reactions to humans and the threat of predation. Animals may increase vigilance at the cost of other fitness-enhancing activities (Duchesne et al., 2000), spend energy to flee and reduce time in areas that might otherwise provide vital resources (Beal, & Monaghan, 2004; González, Arroyo, Margalida, Sánchez, & Oria, 2006), or avoid the risk entirely by foraging in possibly sub-optimal habitats (Frid & Dill, 2002). When no other suitable habitats exist, animals may shift their activity budgets and related energetic costs rather than being displaced (Gill et al., 2001). If disturbance results in decreased food intake or higher energy expenditure, it might result in decreased bodily condition and decreased reproductive success, which could have a population-level impact (Gonzales et al., 2006; Shively, Alldredge, & Phillips, 2005). Hence, understanding the activity patterns of a species is critical for developing successful conservation and management strategies for species (Frey et al., 2017).

3.1.2. Daily Activity Pattern of the Brown Bear

The brown bear activity patterns were reported to be crepuscular (in Kastamonu, Ugış, 2023; in Bartın, Horasan, 2018; in Scandinavia, Moe, Kindberg, Jansson, & Swenson, 2007; in Alberta, Munro, Nielsen, Price, Stenhouse, & Boyce, 2006), nocturnal (in Zonguldak, Çoğal, 2016; in Sakarya, Töke, 2023; in northwestern Anatolia, Soyumert, 2010; in Spain, Parres et al., 2020; in Croatia and Slovenia, Kaczensky et al. 2006; in Sweden Ordiz et al. 2014) and diurnal (in British Columbia, MacHutchon et al., 1998; in Russia Seryodkin, Kostyria, Goodrich, & Miquelle, 2013). Since good visual acuity can increase efficient foraging (berries, salmon etc.) daytime is the most convenient period for foraging by brown bears (MacHutchon et al., 1998). However, other environmental factors and intraspecific interactions can contribute to regulation of diel activity patterns. Several factors were found to influence the daily activity of brown bears including availability of food resources, demographics, and human disturbance.

3.1.2.1. Demographics. Younger brown bears show more nocturnal activity in Scandinavia (Hertel, Swenson, & Bischof, 2017) and more diurnal activity in North America (MacHutchon et al., 1998). Female brown bears are more diurnal compared to males, probably to avoid conflict or infanticide by males (Schwartz, Cain, Podruzny, Cherry, & Frattaroli, 2010; Steyaert et al. 2013; Hertel, Swenson, & Bischof, 2017; Parres et al., 2020). Among 57 cases of intraspecific killing, 44% of the victims were cubs of the year and 18% were adult females (McLellan, 1994, in Schwartz et al., 2010). While opting for nocturnal activity is effective to avoid humans, subadult males, females and cubs seem to show diurnal activity to avoid conflict with adult males (Parres et al., 2020).

3.1.2.2. Human disturbance. Higher road densities and human developments were found to be related with nocturnal activity of brown bears in Sweden (Hertel et al., 2017; Ordiz et al., 2016; Schwartz et al., 2010). Especially becoming more nocturnal after hunting season starts and after encounters with humans may indicate nocturnal behavior is a response to human disturbance (Ordiz et al., 2012; Ordiz et al., 2013). In North America, while brown bears are reported to be more diurnal, diel activity of the brown bear shifted to becoming nocturnal and crepuscular when intensity of human activities increased (MacHutchon et al., 1998; Olson, Squibb, & Gilbert, 1998; Gibeau, Clevenger, Herrero, & Wierzchowski, 2002; Rode, Farley, & Robbins, 2006). For instance, brown bears switched their habitat use spatially and temporally in presence of humans near Kulik River, Alaska (Smith, 2002). In Brooks

River, Alaska, variation was observed in response to human disturbance among bears. Habituated bears did not change their activity in human presence during salmon season, however, non-habituated brown bears delayed their foraging by more than 17 days (Olson, Gilbert, & Squibb, 1997). Diurnal activity of brown bears in Italian Alps decreased significantly at sites with higher human activity (Oberosler, Groff, Iemma, Pedrini, & Rovero, 2017). The same pattern was observed for brown bear populations in Russia (Seryodkin, Kostyria, Goodrich, & Miquelle, 2013). A recent global study examining activity patterns of mammals in relation to human activities showed that carnivores showed highest decreases in activity and highest increases in nocturnality with the increase of human activities (Burton et al., 2024).

3.1.2.3. Food resources. Presence of good foraging habitat is related with diurnal activity of brown bears (Hertel et al., 2017). In Sweden, brown bears tend to show more diurnal activity during the period they feed on berries, and more nocturnal and crepuscular activity in the seasons they feed on reindeer and moose calves (Ordiz, Saebø, Kindberg, Swenson, & Støen, 2016). Similarly, in Kamchatka, brown bears show high activity during the day during berry and salmon seasons (Revenko, 1993 in Seryodkin et al., 2013). Insufficient food resources can also have an influence on diel activity patterns. Animals in search of food tend to be active both day and night to increase their chances of finding food (Zavatsky, 1993, in Seryodkin et al., 2013).

3.1.3. Aims of this Chapter

In this chapter we aim to investigate the diel activity pattern of brown bears in Yenice forest, to provide insight into their ecological needs and preferences, and to understand the diel activity behavior of other carnivores and ungulates in the study area to see if there is temporal partitioning in their habitat use. Although many factors can contribute to the daily activity patterns of brown bears, such as demographics and availability of food resources, literature also suggests a relationship between brown bear diel activity patterns and human disturbance. It is suggested that brown bears tend to be more nocturnal or crepuscular to avoid human disturbance and risks associated with human presence. Yenice WLDA and surrounding hunting grounds are open to human activities and there are year-round logging activities and legal/illegal hunting. Therefore, we hypothesize that brown bears in Yenice Forest would show mostly crepuscular and nocturnal activity. We also hypothesize that there is no difference between the activity patterns of brown bears in Yenice WLDA and surrounding state hunting grounds.

3.2. Methods

3.2.1. Data Preparation

Activity pattern analysis was based on the time a photo was captured by camera traps. For this analysis, total data were used instead of filtered data with independent captures because filtered data do not reflect the continuous behavior of the animal and can lead to bias in the activity pattern estimates (Peral, Landman, & Kerley, 2022). Overall activity patterns were estimated for each species, pooling the data from all camera trap locations. Human activity was represented by vehicle activity in the area, and individual human captures, being very low (176) compared to vehicles (2561), were not included because the effects of a vehicle *versus* a human would not be identical. Camera traps were set starting on 20 May 2015 and very little data were collected in this month. Therefore, May was excluded from the monthly analysis of activity. The capture time of each individual was transformed into a radian using the formula: Time in radians = (hour/24) + (minute/60/24).

Daily activity overlap between brown bears and vehicles was evaluated by categorizing data at five levels, considering the potential changes in the activity patterns due to environmental and temporal factors.

- overall overlap: Data from all camera trap locations and all periods were combined.
- overlap in different months: Data were grouped by months and analyzed for each month.
- overlap in WLDA and non-WLDA areas: data were grouped based on camera trap locations and analyzed for each area.
- overlap in different seasons: Data were grouped as summer (June-August) and autumn (September-November), the seasons that camera traps were active during the year.
- overlap at forest paths and roads: Data were grouped based on camera trap locations, either located on roads or forest trails.

Daily activity overlap of brown bears with canids and humans was evaluated at two levels.

- Overall overlap: Wolf, jackal, and fox were grouped together as canid species and this group was compared to brown bears and humans.
- Species by species: Wolf, jackal, and fox activity overlap with brown bear was analyzed separately.

Daily activity overlap of brown bears with ungulates, wild boar, roe deer, and red deer was evaluated at two levels.

- Overall overlap: Brown bear activity was compared with each species individually using total data.
- Seasonal overlap: Brown bear activity was compared with each species individually using seasonal data as summer (June-August) and autumn (September-November).

3.2.2. Analysis of Activity Overlap

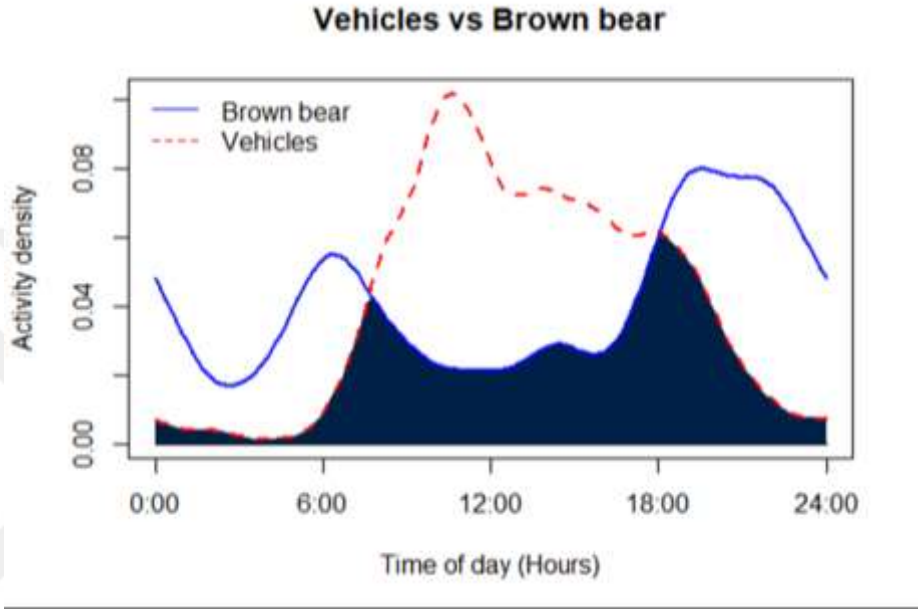
Activity patterns of brown bear, human, canid species (grey wolf, golden jackal, red fox) and ungulate species (wild boar, red deer, and roe deer) were estimated using kernel density analysis in R v 4.1.2 (R Development Core Team 2021) with Overlap package v 0.3.4 (Ridout, & Linkie 2009). Overlap between two distributions was evaluated using the coefficient of overlapping Δ, varying from zero (no overlap) to 1 (complete overlap) (Ridout, & Linkie, 2009). Δ is defined as "the area under the curve which is formed by taking the minimum of the two density functions at each time point" (Linkie, & Ridout, 2011). Since the sampling size is larger than 75 for each species, Dhat4 (Δ_4) estimator was applied (Meredith, & Ridout, 2017). The vehicle capture rate was lower on the forest roads and Dhat1 (Δ_1) estimator was applied in this case. Two species were considered to have had a strong overlap in daily activity patterns if $\Delta_4 > 0.75$, a moderate overlap if $0.5 > \Delta_4 \geq 0.75$, and a low overlap if $\Delta_4 \leq 0.5$ (Monterroso et al., 2014). The activity overlap's confidence intervals (CI) were calculated by 1000 bootstrap samples of probability density functions (Meredith, & Ridout, 2017).

3.3. Results

In this study, during a total of 4935 trap nights, 2561 vehicles, 230 bear, 116 grey wolf, 446 jackal, 419 fox, 235 wild boar, 163 red deer, and 322 roe deer were recorded as independent camera trap photos. May data was excluded from seasonal and monthly analysis since there were only a few records from the camera traps set at the end of May. Trap nights and sampling durations are provided at the end of the graphs for each occasion.

Based on the kernel density estimates, brown bears in general showed a crepuscular activity pattern with peaks at dusk and dawn. The diurnal activity of brown bears was very low, and human activities

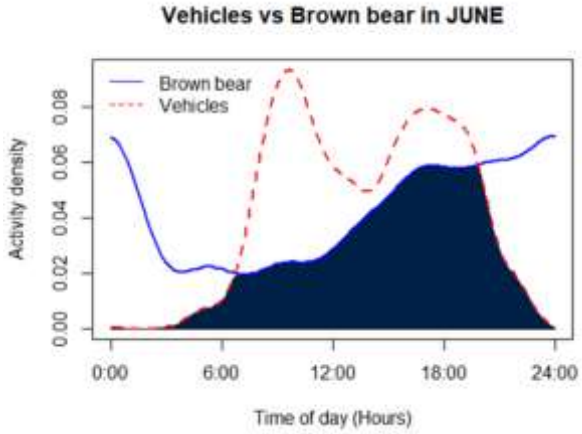
were mostly confined to day hours. Bears were also active during the nighttime with reduced activity before dawn. Overall activity overlap between humans and brown bears was low (Δ_4 = 0.53) according to the classification suggested by Monterroso et al. (2014) (Figure 3.1).



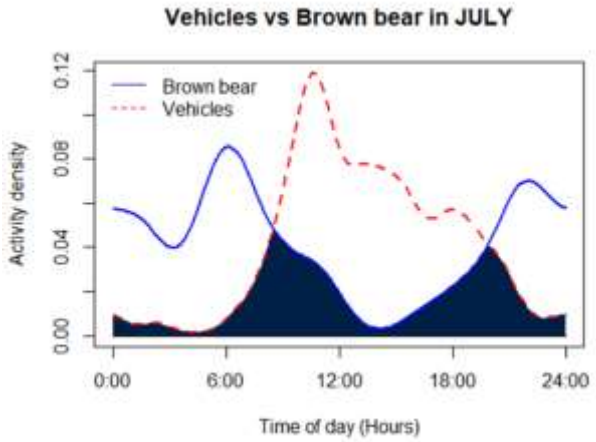
Δ_4 = 0.53, $N_{brownbear}$: 230, $N_{Vehicles}$: 2561

Figure 3.1. Kernel density plot showing daily activity overlap of human and brown bear based on collective data. The coefficient of overlap equals the area below the curves, shaded blue.

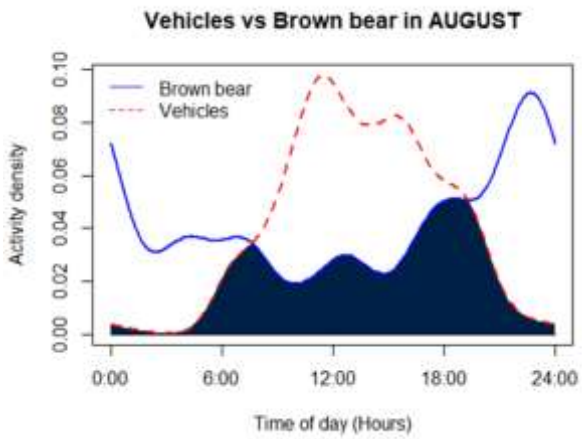
Analyses based on months, seasons, camera trap locations (forest road/trail), protected area status (wildlife Development area/hunting ground) showed similar results. In September and October evening activity peaks were stronger, showing an increase in activity densities in the evening. In June and August, bimodal peaks were not observed, and activity was higher at night. The lowest activity overlap between brown bear and human was observed in July (Δ_4 =0.40) (Figure 3.2). In June and in September, activity overlap between brown bears and humans was moderate (Δ_4 = 0.62 and 0.55, respectively), higher than the other months which showed low overlap (0.51 in Aug, and 0.49 in Oct) (Figure 3.2). When the data were classified based on seasons, activity overlap between brown bears and human was moderate in autumn (Δ_4 = 0.54) with a peak in the evening, and it was low in summer (Δ_4 = 0.45) (Figure 3.3).



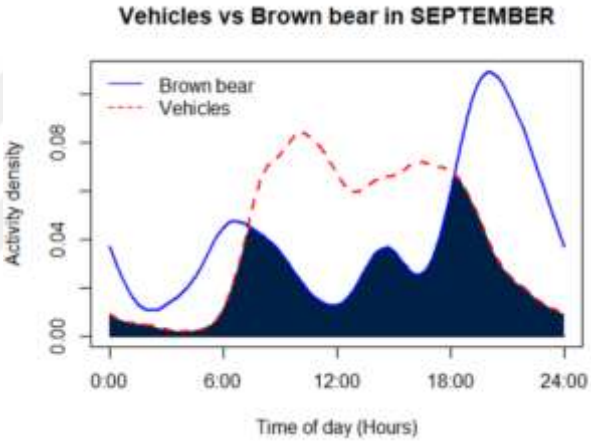
$\Delta_1 = 0.62$, $N_{brownbear}$: 12, $N_{Vehicle}$: 200



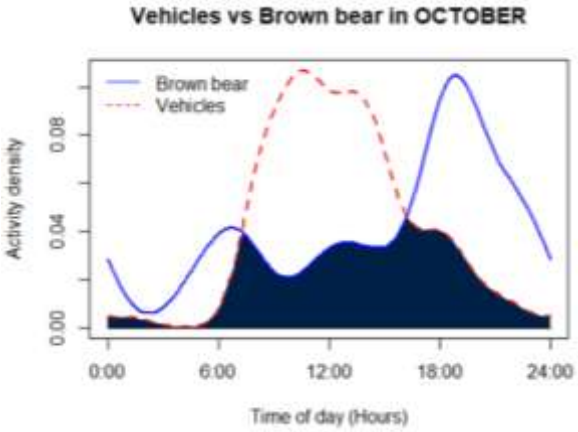
$\Delta_1 = 0.40$, $N_{brownbear}$: 50, $N_{Vehicle}$: 736



$\Delta_1 = 0.51$, $N_{brownbear}$: 29, $N_{Vehicle}$: 387

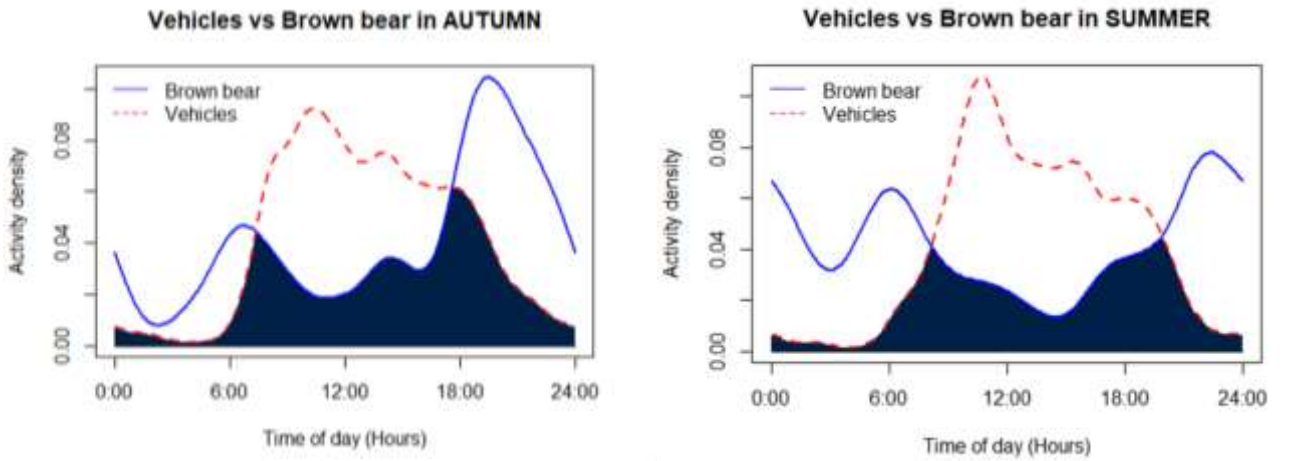


$\Delta_4 = 0.55$, $N_{brownbear}$: 89, $N_{Vehicle}$: 858



$\Delta_1 = 0.49$, $N_{brownbear}$: 44, $N_{Vehicle}$: 337

Figure 3.2. Kernel density plot showing daily activity overlap of human and brown bear for each month.

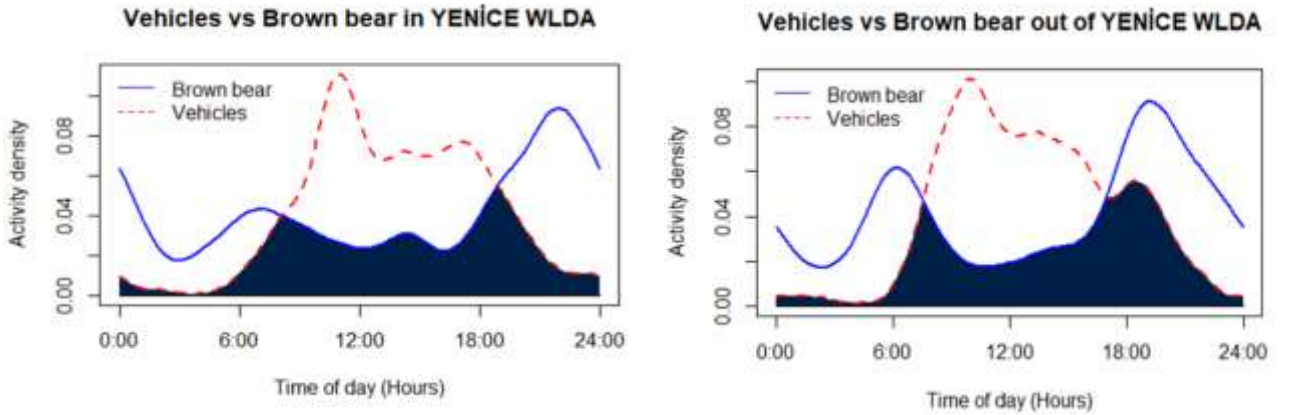


$\Delta_4 = 0.54$, $N_{brownbear}$: 135, $N_{Vehicle}$: 1219

$\Delta_4 = 0.45$, $N_{brownbear}$: 91, $N_{Vehicle}$: 1323

Figure 3.3. Kernel density plots showing daily activity overlap of human and brown bear in summer and autumn.

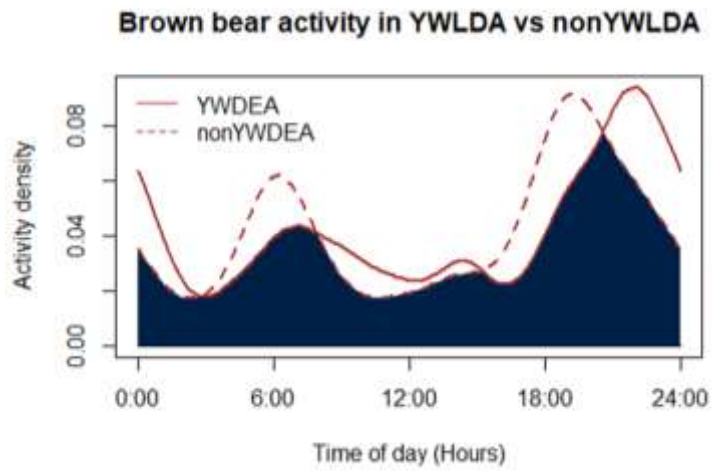
Brown bear and human activity overlap in Yenice WLDA and surrounding forest was similar ($\Delta_4 = 0.52$ and $\Delta_4 = 0.51$, respectively (Figure 3.4). Activity pattern of brown bears in these two areas were quite similar with peaks at nearly same times during the day and showed high overlap ($\Delta_4 = 0.81$) (Figure 3.5).



$\Delta_4 = 0.52$, $N_{brownbear}$: 102, $N_{vehicle}$: 1155

$\Delta_4 = 0.51$, $N_{brownbear}$: 128, $N_{vehicle}$: 1406

Figure 3.4. Kernel density plots showing daily activity overlap of human and brown bear in Yenice Wildlife Development Area and surrounding forest (nonYWLDA).



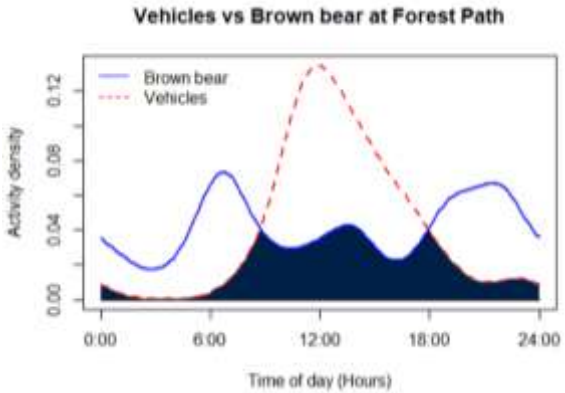
$\Delta_4 = 0.81$, N_{YWLDA}: 102, $N_{nonYWLDA}$: 128

Figure 3.5. Kernel density plots showing daily activity overlap of brown bear in YWLDA and surrounding hunting grounds (nonYWLDA).

Activity overlap between brown bear and vehicles were moderate on the forest roads ($\Delta_4 = 0.51$) and low on the forest paths $\Delta_1 = 0.45$. Only 24 vehicles were captured on the forest paths while 2537 of the vehicles were captured on the forest roads. Since the vehicle capture rate was lower than 75 on the forest roads and Dhat1 (Δ_1) estimator was applied (Figure 3.6). Activity overlap of brown bears on forest roads and forest paths highly overlapped (Figure 3.7).

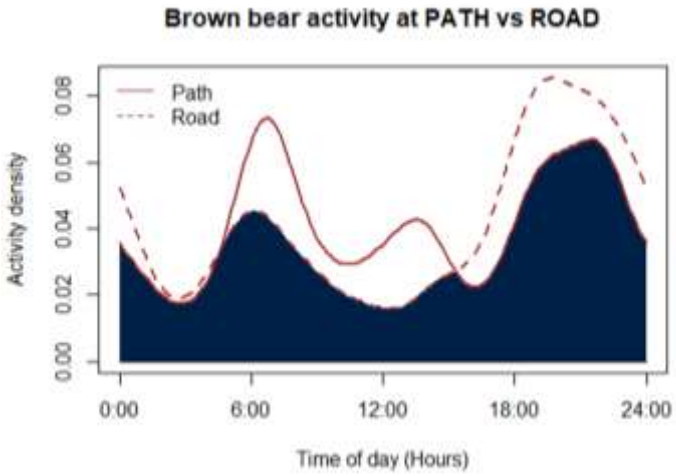


$\Delta_4 = 0.51$, $N_{brownbear}$: 163, $N_{vehicle}$: 2537



$\Delta_1 = 0.45$, $N_{brownbear}$: 67, $N_{vehicle}$: 24

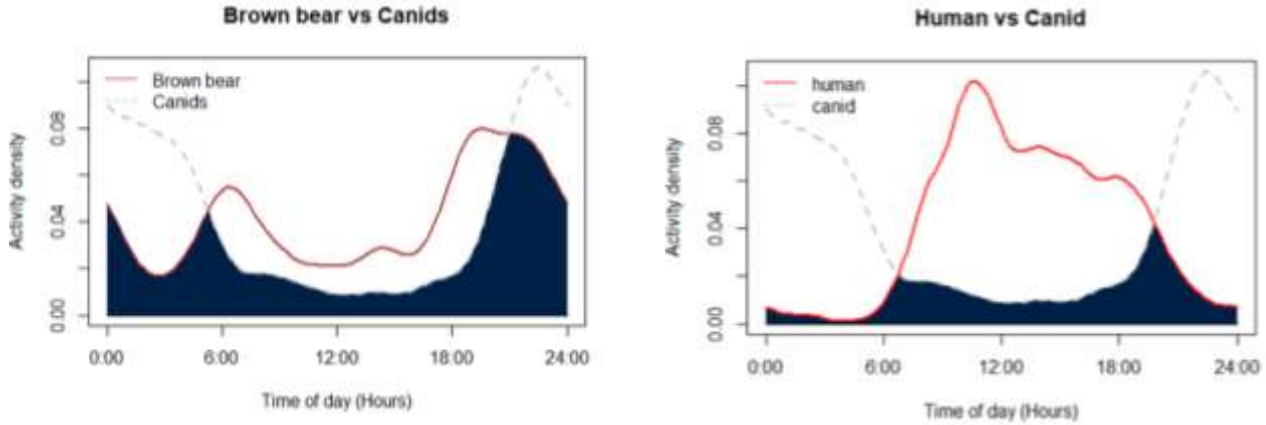
Figure 3.6. Kernel density plots showing daily activity overlap of human and brown bear in forest roads and forest paths.



$\Delta_4 = 0.82$, N_{road}: 163, N_{path}: 67

Figure 3.7. Kernel density plots showing daily activity overlap of brown bear in forest roads and forest paths.

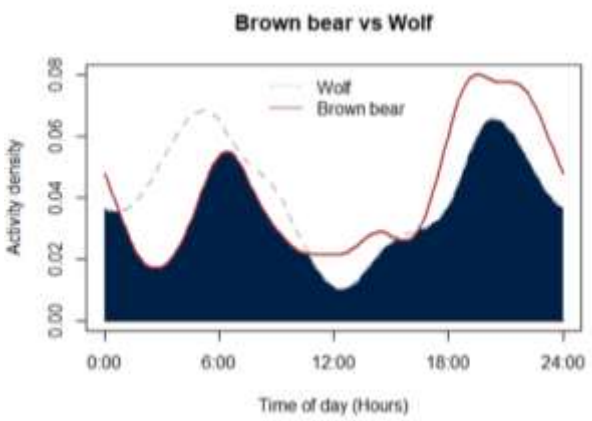
Canid activity levels were minimum during the day when human activities were high. Wolves showed more nocturnal and crepuscular activity pattern similar to brown bears with peaks at dusk and dawn. Jackals and foxes showed a nocturnal activity pattern. Temporal overlap between wolves and brown bears was high ($\Delta_4 = 0.82$). Brown bear showed moderate overlap of activity with jackal ($\Delta_4 = 0.62$) and fox ($\Delta_4 = 0.62$). Overall activity overlap between vehicles and canids were low ($\Delta_4 = 0.29$) (Figure 3.8).



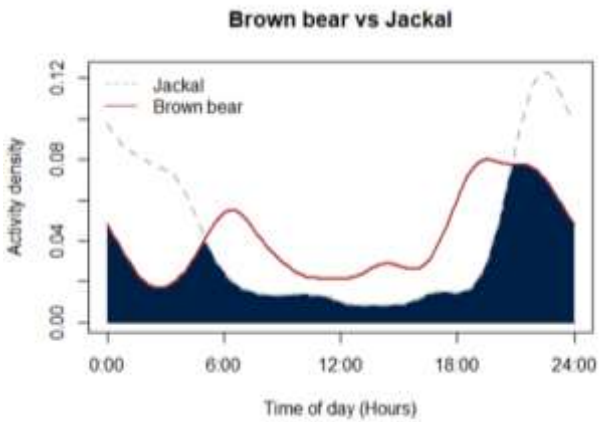
$\Delta_4 = 0.65$, $N_{brownbear}$: 230, N_{canids}: 981

$\Delta_4 = 0.29$, $N_{vehicles}$: 2561, N_{canids}: 981

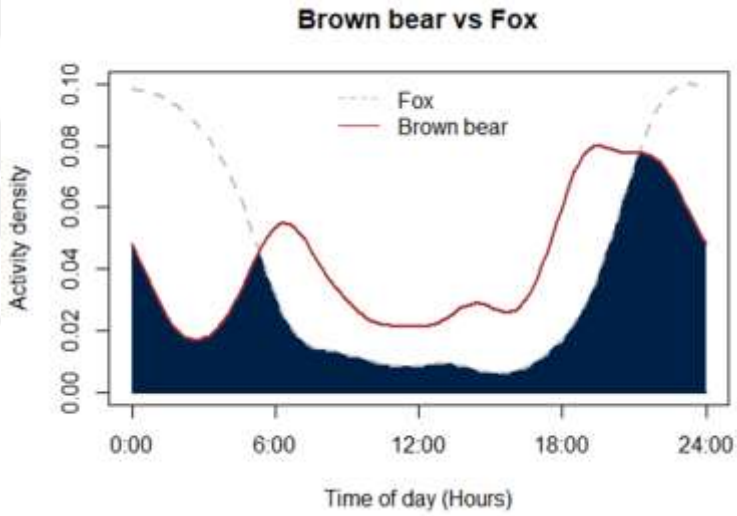
Figure 3.8. Kernel density plots showing daily activity overlap of human, brown bear and canids.



Δ_4 = 0.82, $N_{brownbear}$: 230, N_{wolf}: 116



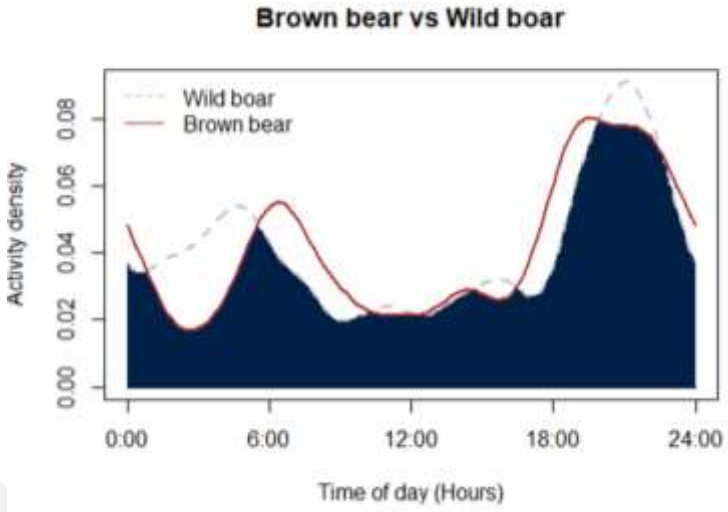
Δ_4 = 0.62, $N_{brownbear}$: 230, N_{jackal}: 446



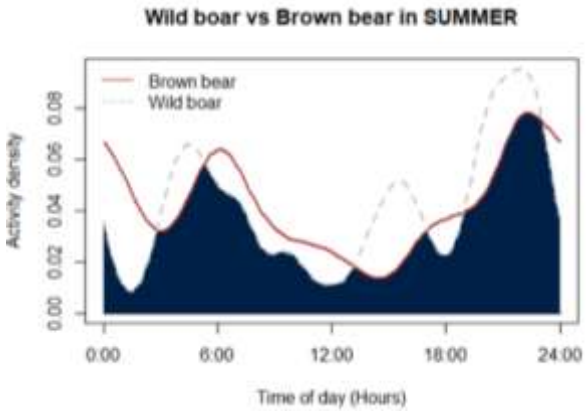
Δ_4 = 0.62, $N_{brownbear}$: 230, N_{fox}: 419

Figure 3.8.(cont.) Kernel density plots showing daily activity overlap of human, brown bear and canids.

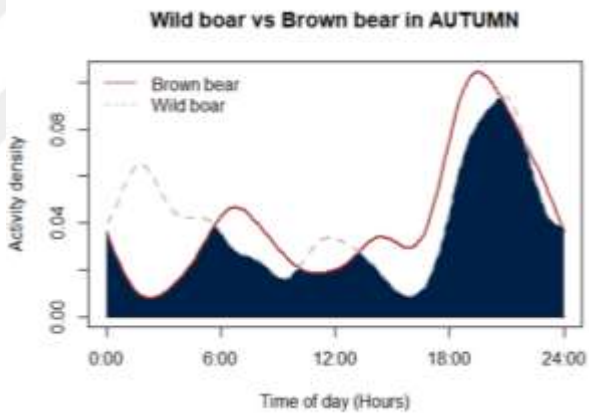
Wild boar was active day and night with peak in early morning and in the evening, and showed high overlap in general (Δ_4 = 0.87) with brown bear. Overlap between these species was also high in summer (Δ_4 = 0.78) and autumn (Δ_4 = 0.78). In summer, morning and evening peaks of wild boar are followed by the activity peaks of brown bear. In autumn, both species showed the highest activity in the evening nearly at the same hours. On the contrary brown bear activity increased while wild boar activity decreased during late night-early morning hours (Figure 3.9).



Δ_4 = 0.87, $N_{brownbear}$: 230, $N_{wildboar}$: 235



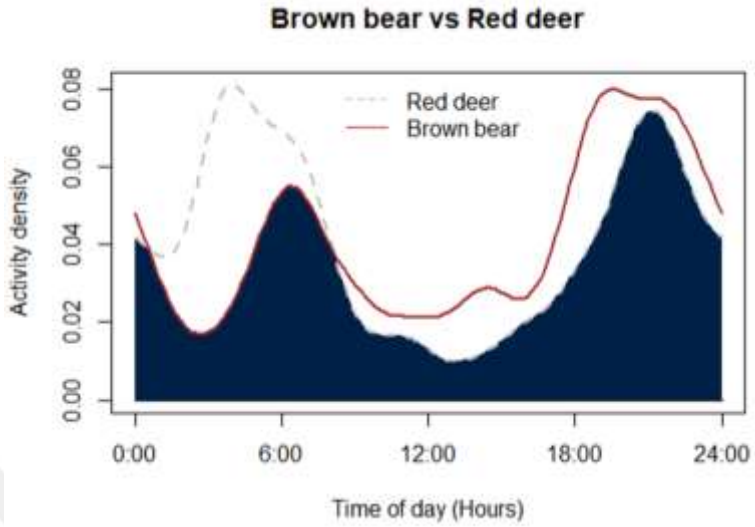
Δ_4 = 0.78, $N_{brownbear}$: 91, $N_{wildboar}$: 115



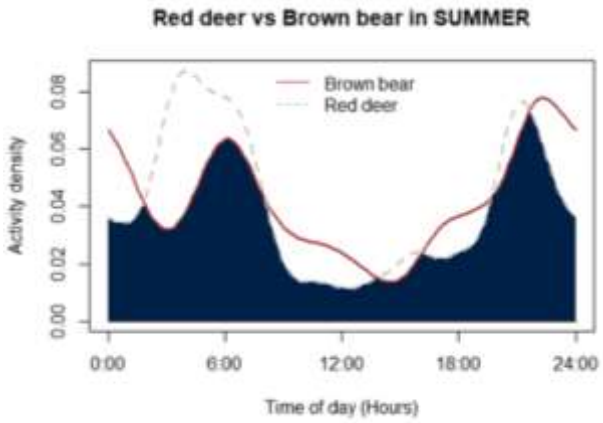
Δ_4 = 0.78, $N_{brownbear}$: 135, $N_{wildboar}$: 113

Figure 3.9. Kernel density plots showing daily activity overlap of wild boar and brown bear.

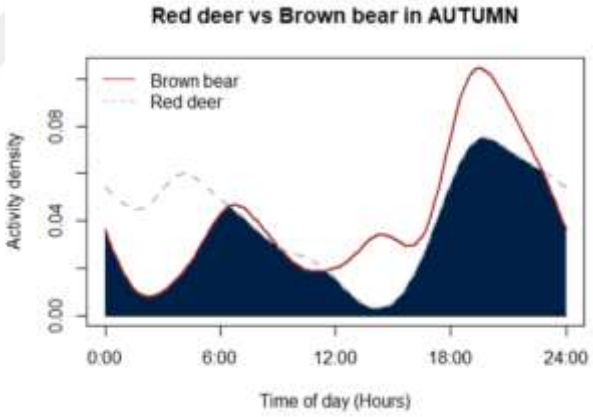
Red deer and brown bear diel activity showed similar patterns with reduced activity during the day and peaks at dusk and dawn. Brown bear activity peak followed red reed peak in the morning in general and in both seasons. In autumn both species showed higher peaks in the evening hours compared to the morning peaks. Activity overlap between red deer and brown bear was high (total Δ_4 = 0.80, summer Δ_4 = 0.81, autumn Δ_4 = 0.78) (Figure 3.10).



Δ_4 = 0.80, $N_{brownbear}$: 230, $N_{reddeer}$: 163



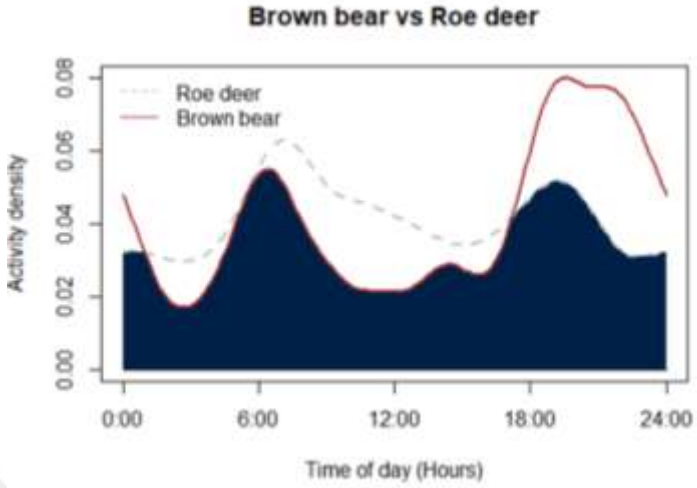
Δ_4 = 0.81, $N_{brownbear}$: 91, $N_{reddeer}$: 116



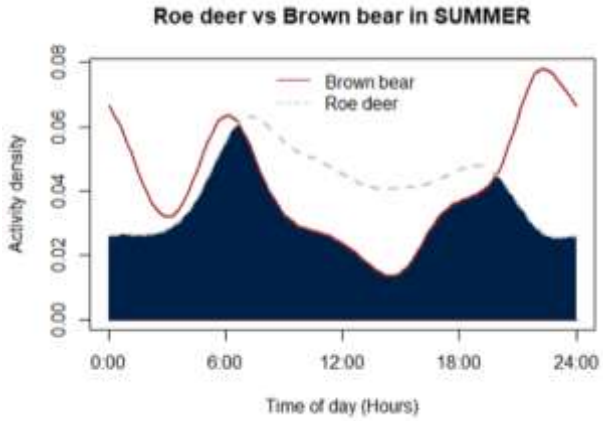
Δ_4 = 0.78, $N_{brownbear}$: 135, $N_{reddeer}$: 46

Figure 3.10. Kernel density plots showing daily activity overlap of red deer and brown bear.

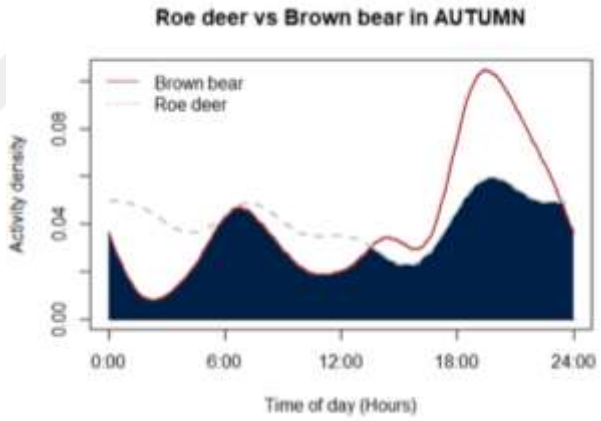
Roe deer and brown bear activity peaks were also similar in the morning, with roe deer showing higher activity during day hours when compared to brown bear. Especially in summer roe deer activity was higher during the day compared to autumn. Activity overlap between red deer and brown bear was high (total Δ_4 = 0.80, summer Δ_4 = 0.76, autumn Δ_4 = 0.78) (Figure 3.11).



Δ_4 = 0.80, $N_{brownbear}$: 230, $N_{roedeer}$: 322



Δ_4 = 0.76, $N_{brownbear}$: 91, $N_{roedeer}$: 228



Δ_4 = 0.78, $N_{brownbear}$: 135, $N_{roedeer}$: 77

Figure 3.11. Kernel density plots showing daily activity overlap of wild boar and brown bear.

3.4. Discussion

In this chapter, we analyzed daily activity patterns of brown bears and estimated the overlap of brown bear diel activities with vehicles and other carnivores (grey wolf, golden jackal, and red fox) in the study area. Brown bears showed mostly crepuscular and nocturnal activity, reducing their activities during daytime, which is the period human activities take place in the area. This pattern is consistent with the studies in Turkey (Töke, 2023; Horasan, 2018; Çoğal, 2016; Ugış, 2023; Soyumert, 2010; Ambarlı

2012) and in Europe (in Spain, Parres et al., 2020; in Scandinavia, Moe, Kindberg, Jansson, & Swenson, 2007; in Slovenia and Croatia, Kaczensky et al., 2006; in Sweden, Ordiz et al., 2014). Yenice WLDA and surrounding forests are managed by Karabük Directorate of Forestry and there is year-round logging activity in the forest. The public also visit the forest for collecting mushrooms and berries, hiking, and other activities. Human activities are mostly confined to daytime. Several studies report brown bears shift their activity patterns to be more nocturnal when/where human activity is high (see 3.1.2.2. Human disturbance). Although brown bear activity is very low during daytime when vehicle capture frequencies are high and increases upon the decrease of the latter (Figure 3.1), we do not have definitive proof of a causal relationship between vehicle capture frequencies and activity pattern of brown bears in Yenice Forest. Unfortunately, in Yenice Forest there are no strictly protected forest patches where human entrance is not allowed and that are larger than the home range of a brown bear. Therefore, comparison of brown bear activity between an undisturbed forest and a forest with human activities was not possible in our study area. The activity overlap between brown bears and vehicles were almost the same when Yenice WLDA ($\Delta_4 = 0.52$) and surrounding forests ($\Delta_4 = 0.51$) were compared which is in accordance with our hypothesis.

When we compared the activity patterns of brown bears and overlap with vehicle capture frequencies between seasons and between months, we did not observe a prominent difference. In the months of September and October, evening activity peaks were stronger, showing an increase in activity densities in the evening. At the end of summer and in autumn, bears enter the hyperphagia period and look for food most of the day to store as much fat as possible before hibernation. During this period, they may need to be active during the day (Ordiz et al., 2016). In our study area, this search for food seems to take place at night with an increase in activity densities after 17:00 - 18:00 and reduce to minimum by 02:00 – 03:00. In the months of June and August, bimodal peaks were not observed, and activity was higher at night. The lowest activity overlap between brown bear and vehicles was observed in July (Δ_4 =0.40) (Figure 3.3). The highest level of vehicle activity was observed in July and September (Figure 3.12). It is known that in summer, the daily activity pattern of brown bears is influenced by precipitation, cloud cover, and temperature (Seryodkin et al., 2013). Similarly, the brown bear in Yenice Forest may have opted for resting on hot summer days with high human activity and kept being active despite high human activity because of the high need for foraging in autumn before hibernation and better weather conditions for being active.

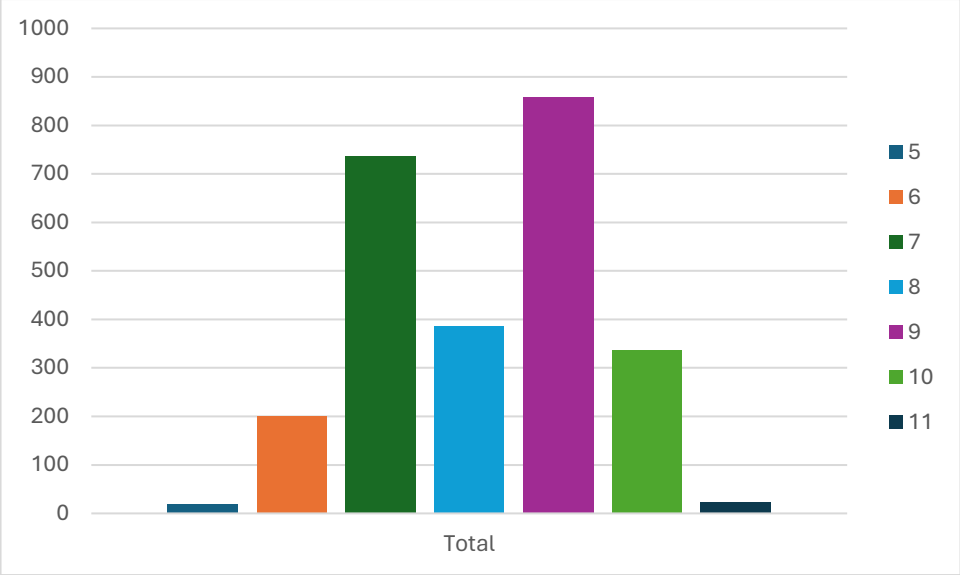


Figure 3.12. Number of vehicles detected by camera traps per month.

According to the literature, brown bears in other parts of the world do not change their activity patterns significantly between the seasons, however the density of activities changes during the day. For instance, Seryodkin et al. (2013) reported that activity patterns of brown bears in Russian Far East do not change seasonally, however, there are variations in activity levels and activity peaks during the day. They also reported that during rainy and cloudy days brown bears were more active during the day and less active at night, and they tended to rest in the shade in sunny weather. In Spain, brown bears showed a crepuscular pattern with lower daytime activity in spring and higher daytime activity in autumn (Clevenger, Purroy, & Pelton, 1990). On the other hand, season did not have a significant influence of the activity pattern of brown bears in Croatia and Slovenia (Kaczensky et al., 2006). In Denali National Park, Alaska, brown bears increased the time they spent active during the day in late summer. However, this difference was not at a significant level (Stelmock, & Dean, 1986). Exceptionally, brown bears show higher diurnal activity in spring and summer and switch to crepuscular and nocturnal activity in autumn in Valdai Upland, Russia (Ogurtsov et al., 2018).

The activity overlap of brown bears and vehicles on the forest roads ($\Delta_4 = 0.51$) was higher than the forest paths ($\Delta_1 = 0.45$). Increase of overlap on the forest roads occurred during evening hours. Vehicle captures on the forest paths were confined to day hours and there were only 24 vehicle captures on the forest roads. In general, brown bears avoid roads especially when there are human activities and reduce their activities during the day in response to human activities in areas of higher road density (Ordiz et al.

2016). However, bears can habituate to people and vehicles when they find the activity predictable and do not perceive a threat (McLellan, 1990). It is also reported that bears use roads more often at night, probably using darkness as a cover (McLellan, & Shackleton 1988). In a similar vein, in Yenice Forest, habituation or using darkness as a cover can be the reasons of this higher overlap of activity of vehicles and brown bears on the forest roads.

In our study area, brown bears showed a high activity overlap with grey wolf (Δ_4 = 0.82) and moderate overlap with golden jackal and red fox (Δ_4 = 0.61, 0.62, respectively). Moderate to high activity overlap between brown bear and wolf was observed elsewhere (in Russia, Δ_4 = 0.80, Ogurtsov et al., 2018; in Greece, Δ_4 = 0.70, Petridou, Benson, Gimenez, & Kati, 2023). Similar to our study area, in Poland, wolves were active throughout the day with the highest activity at dusk and dawn when they mostly hunt their preys (Theuerkauf et al., 2003). Nocturnal activity of wolves was found related to domestic animal diet and high road densities in a meta-analysis (Theuerkauf, 2009). In Kastamonu wolf was mostly nocturnal and diel activity pattern was similar with roe deer and red deer, and especially with the wild boar (Ertürk, 2017). A study comparing diel activity patterns of wolves between a human restricted area and control areas with human activities report that wolves become more active during the day in human restricted areas compared to the control areas, with 78% increase in camera trap capture rates and a daily activity peak between 10:00-12:00 (Martínez-Abraín et al., 2023). Comparable to brown bears, grey wolves seem to avoid human presence in our study area temporally. However, there can be other factors such as predation, temperature etc. and based on our analysis we cannot make inferences regarding the reason for these observed patterns. In general canid species, being mostly nocturnal, showed a very low overlap of activity with humans (Δ_4 = 0.28).

Wild boar diel activity is mostly nocturnal in the neighboring forests (Zonguldak, Çoğal, 2016; Kastamonu, Ertürk, 2017 and Ugış, 2023; Ankara, Mengüllüoğlu, 2010; Düzce, Kahraman, 2019). Roe deer was mostly diurnal with peaks in the morning and evening (northwestern Anatolia, Soyumert, 2010; Kastamonu, Ertürk, 2017; Ugış, 2023; Bartın, Horasan, 2018; in Zonguldak, Çoğal, 2016). Red deer was cathemeral with a peak in the early morning in Ankara (Mengüllüoğlu, 2010) with peaks in the early morning and evening in Kastamonu (Ertürk, 2017; Ugış, 2023). In our study area, wild boar was active day and night with peaks in the early morning and the evening. Red deer and brown bear diel activity showed similar patterns with reduced activity during the day and peaks at dusk and dawn. Roe deer and brown bear activity peaks were also similar in the morning, with roe deer showing higher activity during

day hours compared to brown bear. The highest activity overlap of brown bear was observed with wild boar ($\Delta_4 = 0.87$). Brown bear showed high overlap with also roe deer and red deer (for both, $\Delta_4 = 0.80$). The activity overlaps with these species were similar in summer and autumn (wild boar: $\Delta_4 = 0.87$ both seasons; red deer $\Delta_4 = 0.81$ in summer and $\Delta_4 = 0.78$ in autumn; roe deer: $\Delta_4 = 0.76$ in summer and $\Delta_4 = 0.78$ in autumn).

In Zonguldak, golden jackal and red fox mostly showed nocturnal activity and based on the six wolf records wolves were active throughout the day (Çoğal, 2016). Wolf, fox, and jackal showed mostly nocturnal activity in Ankara (Mengüllüoğlu, 2010). In Kastamonu, wolves were mostly nocturnal (Ertürk, 2017). Another study in Kastamonu reported fox was mostly nocturnal, wolf was active all day with peak at dusk and dawn, and jackal showed higher activity between 1:00 – 2:00 and 10:00 – 12:00 (Ugış, 2023). A study in northwestern Anatolia, including study sites in Kastamonu, Bartın and Karabük found that golden jackal and red fox were mostly nocturnal and wolf was cathemeral (Soyumert, 2010). In Sakarya, wolves showed cathemeral activity (Töke, 2023). In our study area, wolf showed reduced activity during the mid-day, and golden jackal and red fox showed low activity during day hours. The low activity of canids during the day may be related to warm temperatures during the day (Theuerkauf et al., 2003), activity patterns of their prey (Ikeda et al., 2016), higher hunting success during the crepuscular period (Theuerkauf, 2009), and adaptation to human avoidance due to heavy human persecution in history (Theuerkauf et al., 2003). However, it is not possible to make inferences regarding which factors drove this low day-hour activity pattern based on our analysis. Jackals and foxes showed similar diel activity patterns and to avoid competition and aggression preferred spatial partitioning and differential use of habitats in Italy (Toretta et al., 2021). An experimental study showed that foxes do not avoid areas with the scent of jackals but avoid those areas when jackals are present (Scheinin, Yom-Tov, Motro, & Geffen, 2006). Similarly in our study area, no evidence for temporal avoidance was observed between these species.

4. COMMUNITY OCCUPANCY MODELING, SPECIES RICHNESS AND INTERACTION OF THE BROWN BEAR WITH THE OTHER MAMMALS IN YENİCE FOREST

4.1. Introduction

4.1.1. Community Occupancy Modeling

Species and community occupancy estimations provide an effective approach for assessing the status and vulnerability of the species and communities (Cavada et al., 2019). Understanding the relationship between species and their environment and how human induced or natural changes are related to their habitat use is essential to develop successful management strategies (Zipkin et al., 2012). Community occupancy and distribution models can also contribute to conservation efforts by providing the information to evaluate whether planned conservation actions are beneficial to the species and to the community (Kery, & Royle, 2008; Zipkin et al., 2010). These models can help to evaluate trade-offs in the expected occurrences of species in association with different conservation actions (Suarez-Rubio & Thomlinson, 2009). Growing concern regarding the effects of global change on community composition is making studies at the community level increasingly necessary (Devarajan et al., 2020). However, although studies on focal species are abundant, communities are relatively understudied.

Similar to single species occupancy models, repeated sampling of multiple locations allows estimating community occupancy in case of imperfect detection (Dorazio, & Royle, 2005). By presuming that all parameters originate from a common distribution, community models retain the capacity to estimate species-specific parameters while sharing information across species. This distribution is then controlled by hyperparameters, which represent patterns or processes at the community level; such model formulation is comparable to including a random effect at the species level (Sollmann et al., 2021). A key advantage of community models is the capacity to derive more accurate parameter estimates for relatively rarely observed species (Sollmann et al., 2021).

4.1.2. Species Richness Estimation

Community occupancy and species richness estimation are fundamental for conservation and management of animal communities. Species richness is the most commonly used metric of biodiversity (Purvis, & Hector 2000). It is frequently used in monitoring programs and is among the most important criteria for selection of natural reserves (Howard et al., 2000). However, species richness estimations may not be reliable when variability in detection rates across species and environmental characteristics are not taken into account (Boulinier et al., 1998). Species richness can be estimated via community occupancy modeling using multiple species presence/absence data from repeated surveys, while accounting for imperfect detection (Guillera-Arroita et al., 2019). Community occupancy models can also estimate the effect of habitat and human disturbance on species richness and species distributions in a community (Cavada et al., 2019). In community occupancy models, collective community data informs occupancy probabilities of all species including rare species. Therefore, community and species-specific occurrence estimates can be more precise (Zipkin et al., 2010). In this type of model, habitat and sampling covariates that influence detection and occupancy can be included, and species richness estimates can be obtained while accounting for variation among species (Russel et al., 2009).

4.1.3. Modeling Species Interactions

Occurrence of a species can be influenced by environmental factors and presence of other species (Rota et al., 2016). The occurrence of predators may have an impact on the habitat preference of prey species (Willems, & Hill, 2009). Additionally, lesser competitors may be prevented from occupying a habitat due to competitive exclusion (Wang et al., 2015), and the presence of another species may make habitat use easier (Veblen, 2012).

The single-season occupancy models can be expanded to include interactions between two or more species. Modeling approach of Rota et al. (2016) assumes that the latent occupancy state is determined by a multivariate Bernoulli random variable, which allows for stable construction of covariate models for species co-occurrence without any initial assumptions of asymmetric interactions. Additionally, it provides specific conditions for independence between species without needing additional parameters, unlike the ‘Species Interaction Factor’ method proposed by MacKenzie, Bailey, and Nichols (2004), and allows determination of pairwise covariance/correlation using basic probability theory. This modeling

approach enables ecologists to model the probability of multiple species occupying the same site as a function of covariates.

4.1.4. Aims of this Chapter

The brown bear is an umbrella species which has large home ranges and protecting this species and its habitat helps in protecting other species inhabiting its habitat. However, species richness and community composition, and the environmental factors that influence them are needed to be evaluated to maintain communities and ecosystems (Zipkin et al., 2010). This chapter aims to make community level analyses of medium and large mammals in Yenice Forest by:

- estimating the environmental and human related factors that influence occupancy of 12 large and medium mammals inhabiting Yenice Forest,
- estimating species richness considering the variability in species detection due to environmental factors,
- comparing species richness with the brown bear occupancy to see if there is a relationship between two, and
- evaluating the interactions between the brown bear and the other apex predator wolf, and three herbivorous species: wild boar, red deer, and roe deer.

4.2. Methods

4.2.1. Data Preparation

The species included in the analysis were grey wolf (*Canis lupus*), Eurasian lynx (*Lynx lynx*), brown bear (*Ursus arctos*), golden jackal (*Canis aureus*), red fox (*Vulpes vulpes*), European wildcat (*Felis silvestris*), roe deer (*Capreolus capreolus*), red deer (*Cervus elaphus*), wild boar (*Sus scrofa*), and badger (*Meles meles*). Due to the nature of the camera traps, the animals with smaller body size may not be detected and therefore smaller mammals such as weasel (*Mustela nivalis*), squirrel *(Sciurus anomalus)*, and marten *(Martes* spp.*)* were not included in the analysis, even though they were detected by camera traps in the area.

Detection history tables were prepared for each species for 160 camera trap locations in total from 2015 and 2016, using Python 3.10 programming language. Species occurrence data were extracted from the overall camera trap data and was coded with '1' for detection, and '0' for non-detection of the respective species. Camera trap inactive days were coded as NA. Data was collated to form five-day survey periods. Total number of detections for 10 species at each location were calculated to create a community data set.

A species accumulation curve was built using Vegan (v 2.4-2) package's *specaccum* function using the community data set to determine whether the sampling effort was sufficient. 19 human related, spatial and environmental covariates were used for modeling. Environmental covariates were analyzed at five different scales (1km, 2km, 4km, 8km, 16km) (see Chapter 2, Table 2.1).

4.2.2. Community Occupancy Model

Community occupancy models were run using R v. 4.2.3 (R Core Team, 2022) *unmarked* package. Royle & Nichols (RN) method which takes into account the relation between detection probability (r) and abundance (N) was used to fit the models (Royle, & Nichols, 2003). Firstly, factors that might affect detection probability were evaluated. Vehicle CR, camera trap location (road/path), and year were used in the model, holding N constant. Likelihood ratio testing was used to select between the top two models via R v. 4.2.3 (R Core Team, 2022) *lmtest* package. Then, using fixed r, different scales of covariates of N were tested against the null model, in order to identify the best performing scale of each covariate for their inclusion into the further analysis. Models with best performing covariates were tested against the simplest model using a likelihood ratio test. Complex model was selected when *P* value was below 0.05 and the covariates in the selected models were used for further modeling.

RN models were run using all covariates that contributed to the model and listed according to their AIC values. Since the resulting number of covariates was too high for the model to converge given the species occurrence data available, only the top five covariates (with lowest AIC values at the univariate model selection stage) were selected for building a multivariate model (global model). Goodness of fit of the global model was tested with goodness-of-fit test of MacKenzie and Bailey (2004) for occupancy models based on Pearson's chi-square statistics, using *mb.gof.test* function in the *AICcmodavg* package (Mazerolle, 2023). Model was considered a good fit when $P > 0.1$ and overdispersion parameter c-hat

value was close to 1. *Dredge* function was used to produce a set of models using the combinations of the covariates in the global model. A set of high performing models (delta AICc < 2.5) were selected for a model averaged prediction which was made using *model.avg* function.

4.2.3. Species Richness and Correlation with Relative Bear Abundance

Brown bears were excluded from species richness estimations for comparison. The species included were grey wolf (*Canis lupus*), Eurasian lynx (*Lynx lynx*), golden jackal (*Canis aureus*), red fox (*Vulpes vulpes*), European wildcat (*Felis silvestris*), roe deer (*Capreolus capreolus*), red deer (*Cervus elaphus*), wild boar (*Sus scrofa*), badger (*Meles meles*), brown hare (*Lepus europaeus*), and marten (*Martes* spp.). Abundance (N) values from Royle & Nichols models were transformed into occupancy values (psi). Species richness was calculated by adding species occupancy values of 12 species at each camera trap location. The data were log-transformed for normal distribution. Pearson correlation between species richness and brown bear occupancy was tested using R v. 4.2.3 (R Core Team, 2022). Brown bear occupancy and species richness relationship model was built based on linear regression in R v. 4.2.3 (R Core Team, 2022).

4.2.4. Species Interactions Occupancy Modeling

The multispecies occupancy models were fit using multispecies occupancy modeling suggested by Rota et al. (2016), which is a modification of single species occupancy modeling of MacKenzie et al. (2002) for two or more species. The models were fit in R v. 4.2.3 (R Core Team, 2022) using *unmarked* package and *occuMulti* function. Occupancy probability of multiple species were estimated in two scenarios: assuming independence among species and assuming dependence among species. Dependent pattern indicates one species more likely occurs at an area where the other species occurs, and independent pattern assumes no dependence between occurrences of the two or more species. Brown bear occurrence was estimated in presence of one of the apex predators, wolf, and in presence of herbivorous species roe deer, red deer, and wild boar. The co-occurrences of the three canid species were also estimated.

4.3. Results

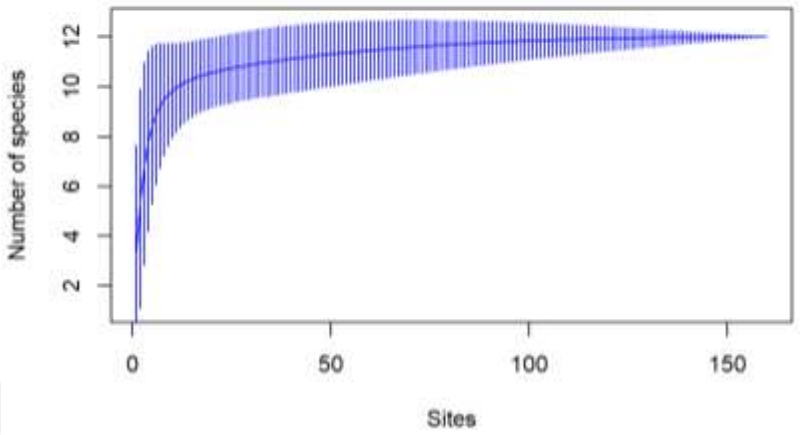
4.3.1. Data Preparation

The camera traps were active for 3090 trap-nights in 2015 and 1945 trap-nights in 2016. Successive photographs of the same species at a camera trap were considered independent captures when they were >15 min. apart. Total number of independent captures for the 12 species of interest was 2502. The least frequently captured species was *Lynx lynx* with four unique detections and the most frequently captured species was *Canis aureus* with 433 independent captures throughout the study (Table 4.1).

Table 4.1. Number of unique detections of ten species from camera surveys.

Species	#Independent captures
Badger, *Meles meles*	59
Brown bear, *Ursus arctos*	210
Red fox, *Vulpes vulpes*	402
Golden jackal, *Canis aureus*	433
Eurasian lynx, *Lynx lynx*	4
European hare, *Lepus europaeus*	5
Marten, *Martes* spp.	325
Red deer, *Cervus elaphus*	147
Roe deer, *Capreolus capreolus*	303
Wildboar, *Sus scrofa*	221
European wildcat, *Felis silvestris*	285
Grey wolf, *Canis lupus*	108
Total	2502

The species accumulation curve reached a plateau, indicating the adequacy of the survey design (160 sites) to effectively survey the study area's large mammal community (12 species) (Figure 4.1).



Error bars indicate 90% prediction intervals.

Figure 4.1. Predicted species accumulation curve for 12 mammals in the study area.

4.3.2. Community Occupancy Modeling

Factors that might affect detection probability (r) were evaluated. Vehicle CR, camera trap location (road/path) and year were used in the model, holding N constant (Table 4.2).

Table 4.2. Test of covariates that might affect detection probability based on Royle-Nichols abundance estimation models.

	#Par	AIC	ΔAIC	AIC wt	Cum wt
N(Species)r(Species+VehicleCR+CamLoc+Year)	30	7195.91	0.00	0.73	0.73
N(Species)r(Species +VehicleCR+ CamLoc)	29	7199.10	3.20	0.15	0.88
N(Species)r(Species+VehicleCR+Year)	29	7199.88	3.98	0.10	0.98
N(Species)r(Species+VehicleCR)	28	7203.68	7.77	0.02	0.99
N(Species)r(Species+CamLoc)	26	7205.33	9.42	0.01	1.00
N(Species)r(Species+CamLoc+Year)	25	7206.74	10.83	0.00	1.00
N(Species)r(Species+Year)	25	7239.68	43.77	0.00	1.00
N(Species)r(Species)	24	7240.57	44.66	0.00	1.00

#Par: Number of parameters, AIC: Akaike Information Criterion score, ΔAIC: The difference in AIC score when compared to the best model, AICwt: AIC weight, relative likelihood of the model, Cumwt: Cumulative weight, The sum of the AIC weights.

Vehicle CR, camera trap location and year together were the most important covariates that affected r based on the AIC values. Null model was built as N(Species)r(Species+VehicleCR+CamLoc+Year). Models were run using one site covariate at a time and for each different scale (1km, 2km, 4km, 8km,

16km) and best performing scales and the covariates that contributes the model (with lower AIC compared to the null model) were selected. Likelihood ratio test was also used to support this contribution.

Human-related variables such as Urban_8km, Vehicle CR; a spatial variable slope_16km; and land-cover variables such as Shannon Diversity Index_8km, Water_4km, Forest_8km, Open Spaces_2km, Agriculture_8km, contrast weighed edge density_8km, contagion_8km, patch density_4km, percent land coniferous forest_1km were the covariates that contributed to the model and help explain the variance in medium and large mammal abundances in our study area. Univarite Royle-Nichols community abundance models with fixed covariates for detection probability are listed in Table 4.3.

Table 4.3. Royle-Nichols community abundance models with the covariates that contributed to the null model with fixed covariates for detection probability.

model	#Par	AIC	ΔAIC	AIC wt	Cum wt
N(Vehicle CR)r(Species+VehicleCR+CamLoc+Year)	34	7166.19	0.00	0.99	0.99
N(Slope_16km)r(Species+VehicleCR+CamLoc+Year)	31	7176.06	9.88	0.01	1.00
N(Water_4km)r(Species+VehicleCR+CamLoc+Year)	31	7179.95	13.76	0.00	1.00
N(Cwed_8km)r(Species+VehicleCR+CamLoc+Year)	31	7182.22	16.03	0.00	1.00
N(Contag_8km)r(Species+VehicleCR+CamLoc+Year)	31	7184.28	18.09	0.00	1.00
N(Shdi_8km)r(Species+VehicleCR+CamLoc+Year)	33	7186.12	19.93	0.00	1.00
N(Forest_8km)r(Species+VehicleCR+CamLoc+Year)	31	7189.45	23.27	0.00	1.00
N(Pd_4km)r(Species+VehicleCR+CamLoc+Year)	31	7189.97	23.79	0.00	1.00
N(Conifer_1km)r(Species+VehicleCR+CamLoc+Year)	32	7190.94	24.76	0.00	1.00
N(Urban_8km)r(Species+VehicleCR+CamLoc+Year)	31	7193.14	26.95	0.00	1.00
N(OpenSpace_2km)r(Species+VehicleCR+CamLoc+Year)	31	7193.84	27.66	0.00	1.00
N(Agriculture_8km)r(Species+VehicleCR+CamLoc+Year)	31	7193.95	27.76	0.00	1.00
N(Species)r(Species+VehicleCR+CamLoc+Year)	30	7195.91	29.72	0.00	1.00

Vehicle CR covariate was the most important common covariate that influence abundance of all 12 mammal species evaluated in the area. Model results for each covariate are given in Appendix B. Table B.1- B.13. Since the number of covariates were too high for a reliable modelling process considering the size of the species occurrence data, vehicle CR, slope_16km, cwed_4km, water_4km and contag_8km as covariates with lowest AIC values were selected for global Royle-Nichols community abundance model (Table 4.4).

Table 4.4. Global Royle-Nichols community abundance model with top five covariates.

Abundance:	Estimate	SE	z	P(>|z|)
(Intercept)	-1.54	0.33	-4.66	0.00
Species(bear)	1.32	0.30	4.36	0.00
Species(fox)	0.73	0.28	2.63	0.01
Species(hare)	2.97	2.15	1.38	0.17
Species(jackal)	0.83	0.28	3.01	0.00
Species(lynx)	2.95	2.44	1.21	0.23
Species(marten)	1.08	0.28	3.88	0.00
Species(reddeer)	0.51	0.29	1.78	0.07
Species(roedeer)	1.15	0.27	4.22	0.00
Species(wildboar)	1.63	0.33	4.92	0.00
Species(wildcat)	1.65	0.31	5.38	0.00
Species(wolf)	-0.12	0.33	-0.35	0.73
Vehicle CR(1-100)	0.63	0.12	5.19	0.00
Vehicle CR(101-300)	0.45	0.25	1.77	0.08
Vehicle CR(301-600)	0.62	0.25	2.53	0.01
Vehicle CR(601-1000)	0.37	0.35	1.06	0.29
Slope_16km	0.15	0.04	3.67	0.00
Cwed_4km(1-2.5)	-0.01	0.19	-0.03	0.98
Cwed_4km(2.6-5)	0.00	0.26	-0.02	0.99
Cwed_4km(5.1-7.5)	0.23	0.52	0.45	0.65
Water_4km(1-25%)	0.29	0.17	1.74	0.08
Contag_8km(76-100%)	-0.10	0.13	-0.77	0.44
Detection:	**Estimate**	**SE**	**z**	**P(>|z|)**
(Intercept)	-1.35	0.29	-4.59	0.00
Species(bear)	-0.13	0.34	-0.39	0.70
Species(fox)	0.90	0.30	2.98	0.00
Species(hare)	-5.71	2.21	-2.59	0.01
Species(jackal)	0.65	0.30	2.15	0.03
Species(lynx)	-5.69	2.49	-2.29	0.02
Species(marten)	0.14	0.31	0.45	0.65
Species(reddeer)	0.29	0.32	0.93	0.36
Species(roedeer)	0.27	0.31	0.88	0.38
Species(wildboar)	-0.51	0.37	-1.38	0.17
Species(wildcat)	-0.20	0.34	-0.59	0.55
Species(wolf)	0.20	0.35	0.58	0.56
Vehicle CR(1-100)	-0.14	0.15	-0.90	0.37
Vehicle CR(101-300)	-0.13	0.28	-0.45	0.65
Vehicle CR(301-600)	0.10	0.28	0.34	0.73
Vehicle CR(601-1000)	0.47	0.33	1.42	0.16
CamLoc(path)	-0.21	0.12	-1.71	0.09
Year (2016)	-0.26	0.09	-2.81	0.00
AIC:	7149.7			

Based on MacKenzie and Bailey goodness of fit for Royle & Nichols model, observed data were consistent with the model based on Pearson's Chi-squared test (p = 0.539) and a c-hat value of 0.82. All possible combinations of the covariates in the global model were used to fit 512 models. Best performing 10 models among these 512 models are presented in Appendix B, Table B.13. A set of high performing models (delta AICc < 2.5) among these combinations were selected for model averaging and beta coefficients were calculated (Figure 4.2).

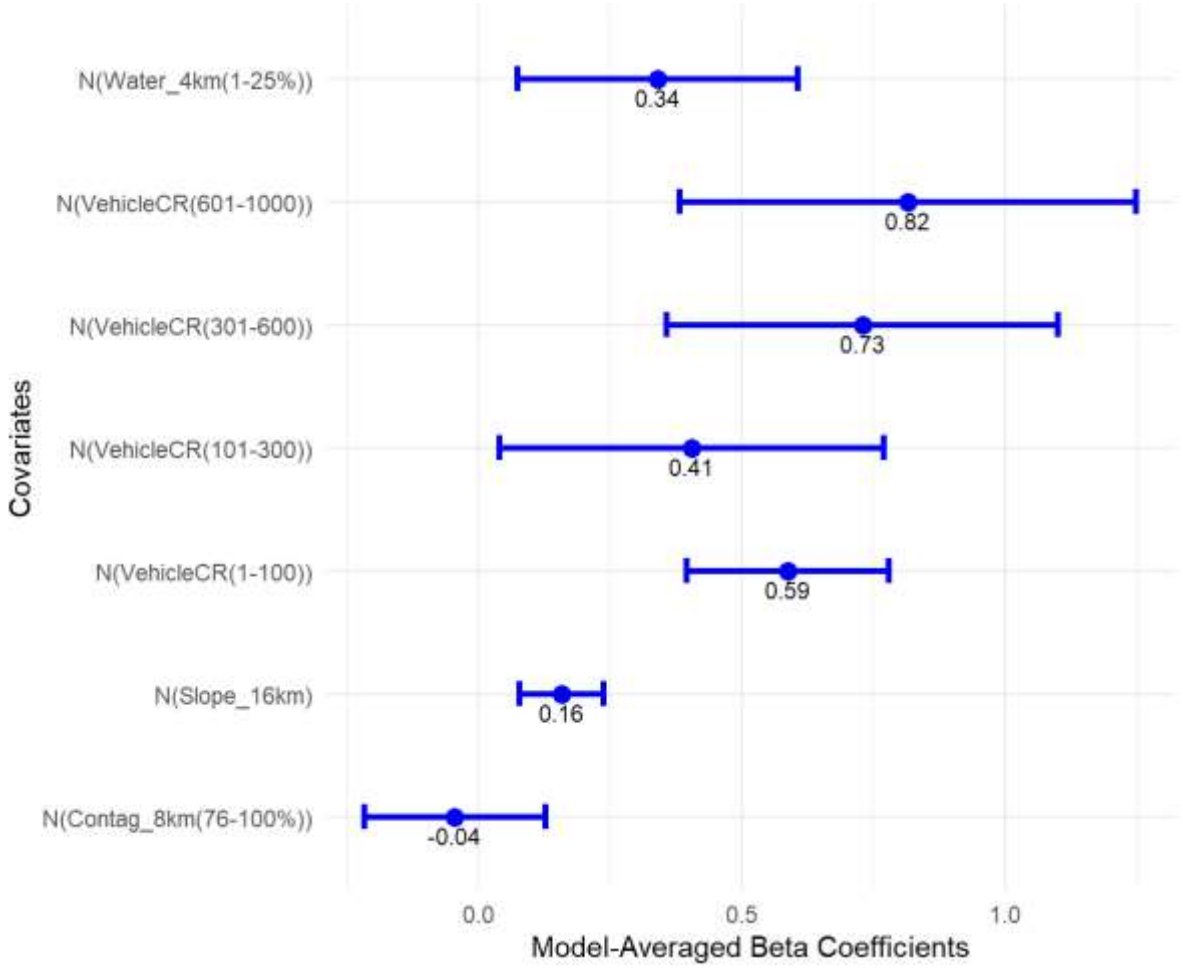


Figure 4.2. Model averaged coefficients for abundance and detection probability.

4.3.3. Species Richness and Brown Bear Relationship based on Linear Regression

Relative bear abundance and species richness relationship model was built based on linear regression. The linear regression model indicates a statistically significant positive relationship between the presence of brown bears and species richness. About 14% of the variance in the species richness is accounted for by the variation in the brown bear occupancy (Table 4.5).

Table 4.5. Linear regression model of relative brown bear abundance and species richness.

	Estimate	SE	t value	P(>|t|)
(Intercept)	-2.42	0.46	-5.17	<0.01
Brown bear	1.33	0.25	5.16	<0.01

Multiple R-squared: 0.14, Adjusted R-squared: 0.139, F-statistic: 26.66 on 1 and 158 DF, p-value < 0.01

4.3.4. Species Interactions Occupancy Modeling

Royle-Nichols abundance estimation models were run using data collected from 160 sites over 4935 trap nights. We included 12 species and the top five significant covariates in our global model successfully. However, the challenge of achieving model convergence prevented us from incorporating species interactions into the overall Royle-Nichols abundance estimate model. Therefore, we ran simpler models to gain insights into species interactions in the studied area.

4.3.4.1. Brown bear-wolf co-occurrence. Brown bear and wolf co-occurrences were estimated assuming independent and dependent patterns. The dependent pattern assumes one species more likely occurs at an area where the other species occurs, and the independent pattern assumes no dependence between occurrences of the two or more species. According to the model results, wolf and brown bear occurred together more frequently than expected by chance ($P = 0.05$). Incorporating dependence between species lowered AIC values 2.5 points (Tables 4.6 and 4.7).

Table 4.6. Multi-species occupancy model of wolf and bear (assuming independence).

Occupancy:		**Estimate**	**SE**	**z**	**P(>|z|)**
Bear	(intercept)	0.0422	0.191	0.221	0.82
Wolf	(intercept)	-1.4515	0.242	-5.992	0.00
Detection		**Estimate**	**SE**	**z**	**P(>|z|)**
Bear	(intercept)	-0.97	0.111	-8.74	<0.01
Wolf	(intercept)	-1.16	0.181	-6.40	<0.01
AIC: 1194.27					

Table 4.7. Multi-species occupancy model of wolf and bear (assuming dependence).

Occupancy:		**Estimate**	**SE**	**z**	**P(>|z|)**
Bear	(intercept)	-0.18	0.21	-0.83	0.40
Wolf	(intercept)	-2.10	0.44	-4.71	<0.01
Bear:Wolf	(intercept)	1.10	0.56	1.97	0.05
Detection		**Estimate**	**SE**	**z**	**P(>|z|)**
Bear	(intercept)	-0.96	0.11	-8.70	<0.01
Wolf	(intercept)	-1.16	0.18	-6.41	<0.01
AIC: 1191.85					

4.3.4.2. Brown bear-ungulate co-occurrence. Brown bear and ungulate co-occurrences were estimated assuming independent and dependent patterns. Multi-species occupancy model including brown bear and herbivorous ungulate species red deer, roe deer, and wild boar showed brown bear and wild boar occur together more frequently than expected by chance ($P < 0.01$). Similar dependence between species was observed between roe deer and red deer ($P < 0.05$). Brown bear and roe deer and brown bear and red deer did not co-occur together more frequently than expected by chance (Tables 4.8 and 4.9).

Table 4.8. Multi-species occupancy model of bear and herbivorous species (assuming independence).

Occupancy:		**Estimate**	**SE**	**z**	**P(>|z|)**
Bear	(intercept)	0.04	0.19	0.22	0.82
Red deer	(intercept)	-0.59	0.19	-2.97	<0.01
Roe deer	(intercept)	0.27	0.19	1.37	0.17
Wild boar	(intercept)	0.60	0.23	2.52	0.01
Detection		**Estimate**	**SE**	**z**	**P(>|z|)**
Bear	(intercept)	-0.97	0.11	-8.74	<0.01
Red deer	(intercept)	-1.06	0.13	-7.65	<0.01
Roe deer	(intercept)	-0.91	0.10	-8.97	<0.01
Wild boar	(intercept)	-1.27	0.11	-10.99	<0.01
AIC: 3205.99					

Table 4.9. Multi-species occupancy model of bear and herbivorous species (assuming dependence).

Occupancy:		**Estimate**	**SE**	**z**	**P(>|z|)**
Bear	(intercept)	-1.50	0.66	-2.27	0.02
Red deer	(intercept)	-1.14	0.53	-2.18	0.03
Roe deer	(intercept)	0.15	0.43	0.36	0.72
Wild boar	(intercept)	0.14	0.47	0.29	0.77
Bear:Red deer	(intercept)	0.55	0.59	0.93	0.35
Bear:Roe deer	(intercept)	0.00	0.55	0.01	0.99
Bear: Wildboar	(intercept)	2.07	0.65	3.18	<0.01
Red deer: Roe deer	(intercept)	1.10	0.51	2.15	0.03
Red deer: Wild boar	(intercept)	-0.66	0.62	-1.07	0.29
Roe deer: Wild boar	(intercept)	-0.39	0.62	-0.62	0.53
Detection		**Estimate**	**SE**	**z**	**P(>|z|)**
Bear	(intercept)	-0.98	0.11	-8.76	<0.01
Red deer	(intercept)	-1.06	0.14	-7.65	<0.01
Roe deer	(intercept)	-0.91	0.10	-8.97	<0.01
Wild boar	(intercept)	-1.26	0.11	-11.07	<0.01
AIC: 3194.92					

4.3.4.3. Co-occurrences of canid species. Multiple species occupancy model of canid species in the study area showed wolf and jackal co-occurrence is observed more frequently than expected by chance ($P < 0.01$). A similar pattern was observed between fox and wolf ($P < 0.05$). AIC values favored model incorporating dependence between species. AIC value of the model assuming dependence between two species was lower compared to the model that assumes and independent pattern (Table 4.10 and 4.11)

Table 4.10. Multi-species occupancy model of canid species (assuming independence).

Occupancy:		**Estimate**	**SE**	**z**	**P(>|z|)**
Fox	(intercept)	-0.57	0.17	-3.35	<0.01
Jackal	(intercept)	-0.41	0.17	-2.35	0.02
Wolf	(intercept)	-1.45	0.24	-5.99	<0.01
Detection		**Estimate**	**SE**	**z**	**P(>|z|)**
Fox	(intercept)	-0.20	0.11	-1.88	0.06
Jackal	(intercept)	-0.49	0.10	-4.71	<0.01
Wolf	(intercept)	-1.16	0.18	-6.40	<0.01
AIC: 911.99					

Table 4.11. Multi-species occupancy model of canid species (assuming dependence).

Occupancy:		**Estimate**	**SE**	**z**	**P(>|z|)**
Fox	(intercept)	-1.10	0.25	-4.34	<0.01
Jackal	(intercept)	-0.97	0.25	-3.92	<0.01
Wolf	(intercept)	-2.82	0.51	-5.57	<0.01
Fox: Jackal	(intercept)	0.68	0.41	1.66	0.10
Fox : Wolf	(intercept)	1.15	0.53	2.19	0.03
Jackal: Wolf	(intercept)	1.57	0.56	2.78	0.01
Detection		**Estimate**	**SE**	**z**	**P(>|z|)**
Fox	(intercept)	-0.20	0.11	-1.85	0.07
Jackal	(intercept)	-0.48	0.10	-4.68	<0.01
Wolf	(intercept)	-1.16	0.18	-6.41	<0.01
AIC: 1893.73					

4.3.4.4. Wolf–ungulate co-occurrence. Multiple species occupancy model of wolf and its potential prey species, red deer, roe deer and wild boar showed no dependence of wolf with its potential prey species in our study area. AIC values favored the model assuming independence between species. AIC value of the model assuming independence between species (Table 4.12) was lower when compared to the model that assumes an independent pattern (Table 4.13). Roe deer was more likely occur in the areas where red deer occurs ($P < 0.05$).

Table 4.12. Multi-species occupancy model of wolf and its potential prey species (assuming independence).

Occupancy:		**Estimate**	**SE**	**z**	**P(>|z|)**
[wolf]	(Intercept)	-1.45	0.24	-5.99	<0.01
[reddeer]	(Intercept)	-0.56	0.20	-2.82	<0.01
[roedeer]	(Intercept)	0.24	0.19	1.24	0.22
[wildboar]	(Intercept)	0.60	0.24	2.53	0.01
Detection:		**Estimate**	**SE**	**z**	**P(>|z|)**
[wolf]	(Intercept)	-1.16	0.18	-6.40	<0.01
[reddeer]	(Intercept)	-1.07	0.14	-7.71	<0.01
[roedeer]	(Intercept)	-0.90	0.10	-8.92	<0.01
[wildboar]	(Intercept)	-1.27	0.12	-10.99	<0.01
AIC: 2759.15					

Table 4.13. Multi-species occupancy model of wolf and potential prey species (assuming dependence).

Occupancy:		**Estimate**	**SE**	**z**	**P(>|z|)**
[wolf]	(Intercept)	-1.44	0.61	-2.37	0.02
[reddeer]	(Intercept)	-0.87	0.52	-1.66	0.10
[roedeer]	(Intercept)	0.12	0.45	0.27	0.79
[wildboar]	(Intercept)	0.79	0.44	1.79	0.07
[wolf:reddeer]	(Intercept)	-0.28	0.58	-0.48	0.63
[wolf:roedeer]	(Intercept)	-0.51	0.53	-0.95	0.34
[wolf:wildboar]	(Intercept)	0.51	0.64	0.80	0.42
[reddeer:roedeer]	(Intercept)	1.08	0.51	2.13	0.03
[reddeer:wildboar]	(Intercept)	-0.47	0.52	-0.89	0.37
[roedeer:wildboar]	(Intercept)	-0.21	0.54	-0.39	0.70
Detection:		**Estimate**	**SE**	**z**	**P(>|z|)**
[wolf]	(Intercept)	-1.17	0.18	-6.42	<0.01
[reddeer]	(Intercept)	-1.07	0.14	-7.69	<0.01
[roedeer]	(Intercept)	-0.91	0.10	-8.95	<0.01
[wildboar]	(Intercept)	-1.26	0.12	-10.95	<0.01
AIC: 2761.04					

4.4. Discussion

In this study we used camera trap detection-non detection data to model community occupancy, species interactions and estimate species richness to better understand large and medium mammal communities in Yenice Forest and provide baseline information for conservation decisions. The species accumulation curve showed we were able to cover the targeted 12 species. 12 out of 21 covariates tested significantly contributed to the model. Vehicle capture rate, mean slope within 16 km radius, percent land of water resources within 4 km radius, and landscape metrics such as contrast weighed edge density within 8 km radius, contagion within 8 km radius were found to influence community level abundance of 12 mammal species in the area the most. Protected area (PA) status and distance to PA border did not influence the occurrence of these species. Medium and large mammal abundance was not significantly different between Yenice Wildlife Enchancement Area (YWLDA) and the surrounding state hunting grounds. There was a significant positive relationship between brown bear occupancy and medium and large mammal species richness. The areas with a higher presence of brown bears tended to have higher species richness. Multiple species occupancy model including species interactions showed that brown bear and wild boar; red deer and roe deer tended to co-occur in the study area, and brown bear and the

other apex predator grey wolf showed a similar pattern. Grey wolf also co-occurred with golden jackal and red fox.

Vehicle capture rate (CR), was one of the factors influential in detecting targeted mammal species. Vehicle capture rate may cause mammals to avoid the roads especially during active periods of log transport and cause lower detections on the roads although the species is present in the locality. However, in our global model, low and moderate levels of vehicle capture rate seem to affect animal detection probability negatively and detection probability increased with high-level vehicle capture rates, although *P* values were not significant at this high level. High-level vehicle capture rates were recorded at only three (CR: 601-1000) and five (CR: 301-600) out of 160 camera traps while low and moderate-level vehicle capture rates were recorded at 45 (CR: 1-100) and six (CR: 101-300) out of 160 camera traps. Finally, 101 camera traps recorded no vehicles. Considering that very low number of camera traps recorded high-level vehicle activity, it is possible that other factors are involved when high-level vehicle capture rates appear to have a positive effect on detection probability. The detection probability of the target species by the camera traps located on the paths was lower. There were only 24 vehicles captured on the forest paths compared to 2537 vehicles on the roads and most of the forest paths were inaccessible by vehicles. Hence, the observed pattern of decreased detection probability in areas with low traffic levels could be influenced by the reduced detectability on the forest paths. The lower detection probability of the target species by the camera traps located on the paths might be due to the more frequent use of roads than paths for easier navigation. Forest paths in the study area are steep paths where animals navigate among dense understories. There are numerous paths that intersect the main roads. A camera trap at a path that is rarely used by mammals can detect fewer species/individuals than a camera trap placed on a road, where the paths end up and mammals are channeled to. Several other studies reported detection probabilities influenced by camera trap location (Soyumert, 2010; Burton et al., 2012; Srbek-Araujo and Chiarello, 2013; Boron et al., 2019). Another covariate that was influential in detection probability of target species was year. Detection probability was lower in the second year of the study. These covariates collectively contributed to explaining the variation in detection probabilities. Consequently, they were included in the analysis together.

Species distributions are often shaped by human disturbance, and environmental factors such as water, food resources and vegetation cover (Rich et al., 2016). Our results, in line with the literature, highlight the importance of water and vegetation cover in addition to the metrics such as Shannon

diversity index of vegetation cover, patch density, contrast weighed edge density and contagion in maintaining medium and large mammal species diversity in Yenice Forest.

We evaluated human activity in three different categories: percent of urban land at five scales (1, 2, 4, 8, 16 km), distance to settlement and human activity based on camera trap captures (i.e. Vehicle CR). Vehicle CR was the top covariate that had a relationship with mammal abundance in the study area. Increasing vehicle capture rates were positively related with the abundance of target species. The higher abundance of target species at camera trap locations with vehicle captures compared to zero captures (intercept), which were mostly on the forest paths, might be also related to other environmental factors or higher activity of the species on the forest roads, which provides easier movement across the sites. Animals may become habituated to human activity and find it predictable and do not avoid areas with high human activity (McLellan, 1990). Habituation of the animals to human presence in the area or temporal partitioning of the space use can also comprise factors that explain higher abundance of target species where vehicle capture rates were higher. The vehicle activity was high during the day whereas most of the target species exhibited low activity levels during the day (Chapter 3). The lower day time activity by medium and large mammals was also reported by other studies in northwestern Anatolia (Töke, 2023; Ugış, 2023; Horasan, 2018; Ertürk; 2017; Çoğal, 2016; Soyumert 2010).

Another anthropogenic variable, distance to settlement, did not contribute to the model and it was excluded from further analysis. As a univariate model, percent land cover by urban land (1-25%) within 8 km was related with higher abundance of target mammals. However, its contribution was weak (AIC 7193.14 vs 7195.91 for m1) and the covariate was not included in our global model. The study area consists of forests and the nearest settlements are forest villages which are not urbanized and have low populations. Therefore, the impact of urban effects in this context can be expected to differ from that observed in large cities. On contrary, crops that villagers cultivate in their gardens, livestock or garbage dumps might be attractive to the animals. It is important to note that this covariate influenced the abundance positively when the percent of urban land was low (%1-25). We did not have data for higher percent of urban land in our study area. Increase of agricultural land percent within an 8 km radius reduced the abundance at community level. However, the contribution of this covariate to the model was very weak and the covariate was not included in our global model (AIC 7193.95 vs 7195.91 for m1). Mammals respond to agricultural land differently depending on other factors such as availability and distribution of natural forest patches and types of crops cultivated (Ehlers Smith et al., 2018). Different

mammal species can show different responses to croplands (Ramesh and Downs, 2015). In a study including different variety of mammals, community level habitat use was highest at croplands and forests in Cerrado, Brazil (Bellon et al., 2022). Use of croplands by some mammal species may be related with the sources they offer (Bellon et al., 2022). In Zonguldak, badger, red fox, roe deer and brown bear preferred forested areas far from human settlements while wild boar and golden jackal were frequently recorded at the camera traps stations near human settlements (Çoğal 2016). Wolves avoid areas with high road densities and high rural human populations (Ertürk, 2017). Similary, in our study area, the species composition might have caused distinct responses. The avoidance of agricultural areas within an 8 km radius, coupled with a higher abundance in regions with settlements within the same distance might reflect the differences in the availability of food resources, and the perceived risks associated with these areas. Given the inherent need for flat and open terrain in agriculture, and medium and large mammals typically finding shelter in forested areas, if a particular crop is not suitable to their needs, probabilities of occurrence can be lower near agricultural lands. Besides, an increase in agricultural land could result in larger, more contiguous patches, potentially leading to a decrease in patch density. High patch density indicates numerous and closely spaced habitat patches, while low patch density refers to fewer and more dispersed patches. According to our model animal abundance was higher in the areas with higher patch density. Therefore, less abundance near agricultural land can also be related to this tendency of the community.

Landscape metrics such as contrast-weighted edge density (cwed) and contagion (contag), Shannon's diversity index (shdi) were also influential in the abundanc of medium and large mammals in our study area. CWED considers the amount of contrast between different land cover types along the edges of patches. Our global model showed that animal abundance increased with increasing CWED. Higher contrast between the habitat patch edges provided better habitat for the community. Contagion measures the degree to which same land cover types are clustered together. Our global model showed that animal abundance decreased with increasing contagion. This means larger areas with the same land cover were not preferred by medium and large mammal community in our study area.

Shannon's Diversity Index (shdi) takes into account the number of land cover types and their relative abundance, providing a measure of landscape diversity. Higher shdi values indicate greater diversity in land cover types within the landscape. shdi was not included in the global model as it was not among the top five covariates. Univariate model showed that increase of shdi was positively influenced animal

abundance at community level. When landscape metrics are interpreted together, our model shows that medium and large mammals in our study area prefer the areas with high diversity of land cover types, higher contrast between the patches of this land cover types, and smaller patches of the same land cover types. To the best of our knowledge these landscape metrics were used for the first time in our study for evaluating medium and large mammal community. The contribution of various landscape metric into the models were reported for several species (white-tailed deer, Dechen Quinn, Williams, & Porter, 2013; roe deer, Saïd & Servanty, 2005; mammal predators, (Gehring & Swihart, 2003).

Water is fundamental for all species. Percent land of water bodies within 4 km radius, was an important factor influencing mammal occurrence. Distance to river, however, was not influential. Several studies report declining occupancy of mammals with increasing distance to water resources (Pettorelli et al., 2010; Schuette et al., 2013; Cavada et al., 2019; Boron et al., 2019; Ramesh and Downs, 2015; Ugış, 2023).

Forest at 8 km radius and coniferous forest at 1 km radius were influential on the abundance of mammal species. All species included in the analysis were forest dwelling species and according to the model results, medium and large mammal abundance decreased with increasing coniferous forest percent. Less diverse food resources in coniferous forests compared to deciduous forest may have caused this result. Medium and large mammal occurrence reduced when percent forest land cover was above 75 %. This means dense forest did not provide the most suitable habitat at community level. 1-25% open space land within a 2 km radius also reduced the occurrence probabilities. Based on corine classification open spaces include the areas with sparse vegetation, burnt and bare rock areas. According to combined interpretation of these two variables, medium and large mammals in the study area seem to favor areas with moderate forest cover, at least 2 km far from open spaces. Ugış (2023) reported that although most of the study area consists of coniferous forest, medium and large mammals in Kastamonu preferred the areas with mixed forests. Erb et al., (2012) reported large mammals' positive relationship with amount of deciduous forest cover within a 5 km radius, and medium mammals' negative relationship with amount of deciduous forest cover within 10 km radius Similarly, the species included in our analysis can show different preferences and species-based analysis can reveal more detailed patterns in habitat use of each species.

Abundance of medium and large mammals in the Yenice Wildlife Development Area (YWLDA) and the surrounding hunting grounds was not significantly different from each other, tested using the variables i) distance to protected area and ii) protected area/not protected, as covariates. This might be a result of the recorded illegal hunting and intensive forestry activities within YWLDA, which might affect the community at a similar level to the surrounding state hunting grounds. Forestry activities can have a different impact compared to transient vehicle movements on the roads. In forestry operations, workers often remain in the forest for extended periods. The noise generated by forestry work, coupled with prolonged human presence in core areas, may have influenced the populations. Studies showed that forestry activities force mammals to leave their habitats (in Düzce, Kahraman, 2019; in Bolu, Naiboğlu, 2016). In addition, several studies were conducted to evaluate the populations in protected and unprotected areas. Soyumert (2010) compared the medium and large mammal capture rates in protected and unprotected areas in northwestern Anatolia. In Kartdağ, WLDA red deer as a protected species had lower capture rates compared to unprotected area Kurtgirmez. Another protected area Sökü WLDA did not show higher capture rates for roe deer which is the target protected species of this WLDA. Besides, red deer and wolf capture rates were lower compared to that in the unprotected area Kurtgirmez. In general, when protected and unprotected areas were compared, there was no significant difference between the capture rates of medium and large mammals. Cazalis et al. (2019) and, Drouilly, Clark and O-Riain (2018) found no relationship between protection status and species richness or abundance. Drouilly et al., (2018), however, reported that large mammal occupancy was higher in PAs. Burton et al. (2012) reported carnivore habitat use was not influenced by hunting. Proximity to tourism infrastructure and proximity to the edge of the park and hunting together reduced the occupancy and the detectability of mammal species and occupancy rates of rare mammal species were higher in a PA in Brazil (Xavier da Silva, Paviolo, Tambosi, Pardini, 2018). Another study from Brazil reported negative effect of PAs on medium sized mammal occupancy in Pampa biome, indicating that planning or the existing conservation actions in the PA are not sufficient (Lima et al., 2020). Human disturbance can have a significant effect on the distribution and occupancy of wild animals and this disturbance often occurs in protected areas (PAs) as well (Petracca et al., 2020). Hence designation of an area as a PA by itself is not sufficient to conserve species, especially when lack of funding prevents effective management (Lindsey et al., 2018) as is the case in Yenice WLDA. Limiting forestry activities can increase abundance of target mammals, in case the reason for similar abundance in and around the protected areas is the similar level of forestry activities in both areas. Variations in other environmental factors between protected and unprotected areas might have obscured the influence of the protected area on species abundance.

Incorporating species interactions into our global Royle-Nichols abundance estimate model with five covariates and 12 species was not possible as model convergence was not achieved. Multispecies interaction model provided insights regarding species co-occurrences in our study area. Based on species interaction models, brown bear and grey wolf, and brown bear and wild boar occur together more frequently in the study area than expected by chance. This pattern was observed also between roe deer and red deer; wolf and jackal; and wolf and fox.

Interestingly, wolf occurrence was not found to be dependent on the roe deer, red deer, and wild boar occurrence in our study area. They were not captured by our camera traps in the same areas more than expected by chance. Wolf captures by our camera traps were more confined to specific areas, possibly indicating the territories of wolf packs, while roe deer, red deer, and wild boar captures were more dispersed in the study areas as can be seen in Figures 1.13 – 1.16 (Chapter 1). Ertürk (2017) reported that grey wolf showed a preference to the sites with higher prey species richness in Kastamonu. However, when evaluated together with the other variables, the contribution of this variable to the model was low. As we could not include other parameters in our interaction model, it is not possible to make solid conclusions regarding the reasons of the pattern observed in our study.

Another interesting finding is that golden jackal and red fox did not show avoidance from grey wolf, and grey wolf and brown bear did not show avoidance from each other. Within the canid guild, dominant species control populations of subordinate species and may result in exclusion of the subordinate species from the sites. Interspecific killing in the Carnivora order, including brown bear, is common and for some species account for 68% of all mortalities. These mortalities include adults and cubs of the other species. Body size and forming packs play important roles in either being killed or kill. As a result, one species may alter its space use or activity patterns (Palomares & Caro, 1999). For instance, grey wolf is known to kill golden jackal (Mohammadi, Kaboli, & López-Bao, 2017), and studies show that re-colonization of wolves in jackal territories resulted in the displacement of jackals from newly established wolf home ranges (Krofel et al., 2017). Wolves are known to kill bear cubs and two species show aggressive behavior on prey kill sites. Of the 108 encounters between brown bear and grey wolf, the most common behavior was of bear and wolf having fought and chased each other (Boitani & Mech, 2003). However, it is also important to consider that the avoidance behavior and encounters that result in a kill mostly occur when food resources are scarce (Palomares & Caro, 1999). In our study area,

the species seem not to have avoided each other spatially and temporally (Chapter 3). Considering the high capture rates of ungulates in our study area, the availability of food resources could have facilitated the coexistence of these species.

The study has also some limitations. Although we made comparisons using the basic model, due to the high number of mammal species in the community and the complexity of models, we did not include species interactions such as predation and competition that could influence species distributions (Caro and Stoner 2003) in our global model. These interactions need to be further studied by reducing the number of species in the model and including the most relevant environmental variables. We placed camera traps on the forest paths and roads due to thick understory vegetation in many parts of the study area and to increase detection probabilities. In our preliminary studies, capture rates of medium and large mammals were very low when camera traps were located outside the roads and paths. However, placing camera traps only on the roads and paths can lead to lower detection of some species when making community level analysis. For example, in a study in a Brazilian National Park, the species composition of mammal communities detected in interior forest was significantly different from that on the roads (Srbek-Araujo and Chiarello, 2013). On the other hand, placing camera traps both on the forest roads and forest paths made interpretation of the effect of vehicle capture rates difficult. Camera traps placed solely on roads can provide a higher resolution regarding the effect of high and low vehicle activity on animal populations, disregarding the effects of other factors related to camera trap location. Similarly, evaluation of carnivore and herbivorous communities separately can also provide more detailed information regarding these groups. It is important to note that this model was based on combined analysis of all species together. Different species may respond the environmental conditions differently. Therefore, it is informative to make species level analysis to better understand the needs of these species.

Although having some limitations, community occupancy analysis can help evaluate species' occupancy and richness with limited funds, time and personnel (Zipkin et al 2009). Conservation efforts can be focused on a species or a region. When the focus is a region, evaluating community level abundance and distribution patterns is more efficient for designing effective management plans (Devaran et al. 2020). Given the extensive human influence within PAs, rapid and effective ways to evaluate the presence and distribution of multiple species to inform conservation efforts is essential (Petracca et al. 2020).

5. CONCLUSIONS AND RECOMMENDATIONS

In Yenice Forest, despite its appearance as a relatively undisturbed natural habitat with limited human activity, the most frequently detected species by camera traps was human with a total of 2737 captures, which included 2561 capture of vehicles (e.g., trucks, tractors, private cars, official cars, and heavy equipment) and 176 pedestrians throughout 4935 trap nights at 59 out of 160 camera traps sites. The total number of captured wild animals was 2709. Among the medium and large mammals, the most frequently captured species was golden jackal and the least frequently captured species was European lynx. Capture rates of the species varied between year, area status, and camera trap location, either road or path. Capture rates for some species such as golden jackal and red fox were inflated at a few camera traps which may be close to their dens or frequently used paths. Capture rates can also be influenced by environmental factors. Therefore, these rates do not indicate the abundance or site preferences of the species without controlling the other factors that might have influenced their detections. However, a pattern was observed from the capture rates of predator and prey species being opposite on forest paths and roads. Carnivores used roads more frequently than paths in the forest while ungulates used forest paths more frequently compared to roads. Wild boar was an exception to this pattern with similar capture rates for both locations.

Hunting dogs and people with rifles captured by camera traps, gun shots heard during field visits, and interviews with locals indicate poaching in Yenice WLDA. The size of YWLDA and its distance to Karabük city where Nature Protection and National Parks Directorate is located, make it difficult to monitor the area for poaching. Patrolling by forest guards in the area on a daily basis and establishing security systems such as automated barriers at the entrances of the protected area can help reduce illegal hunting activity in Yenice WLDA. Another option could be the placement of real time acoustic sensors in the protected forest and immediate action by the forest guards upon detection of gun shots. Utilizing drones also can be another option. Although the forest provides cover, roads and trails can be controlled for parked vehicles that could belong to people who entered the area without permits. These measures would require investment, both equipment and personnel, by the Ministry of Agriculture and Forestry.

Detection probabilities were observed to be influenced by camera trap location for brown bears when using Royle-Nichols abundance estimate models, and by the combination of camera trap location, vehicle capture rate, and year in medium and large mammals when using community level Royle-Nichols

abundance estimate models. In both models detection probability was lower on the forest paths. At the community analysis, low and medium-level vehicle capture rates reduced detection probability and high-level traffic had a positive relationship with detection probability, but it was not at a significant level. High level traffic was recorded only at eight camera traps and 101 camera traps which were mostly on the forest paths did not record any vehicles.

Among 21 covariates evaluated, percent of land cover with open spaces within 16km radius, mean slope within 16km radius, percent of land cover with water bodies within 4km radius, mean elevation within 16km radius, percent land cover with forest within 8km radius were the covariates that influenced brown bear abundance in our study. Community-level analysis with the same set of covariates showed that vehicle capture rates, mean slope within 16km radius, percent land cover with water bodies within 4km radius, contrast weighted edge density within 8km radius and contagion within 8km radius were the most important covariates that contributed to the model.

Human-related covariates were distance to settlement, and percent land cover with urban and vehicle capture rate. Distance to settlement did not contribute to the brown bear and community models. As a univariate model, the percent land cover with urban land within 8 km (1-25%) was related to higher abundance of target mammals. However, the contribution of this covariate to the model was very weak. The study area consists of forests and the nearest settlements are forest villages that are not urbanized and have low populations. Therefore, the impact of urban effects in this context can be expected to differ from that observed in large cities. The crops that villagers cultivate in their gardens, livestock, or garbage dumps might be attractive to the animals. Vehicle capture rate showed a positive relationship with abundance however the higher abundance of target species at camera trap locations with vehicle captures compared to zero captures, might be also related to other environmental factors considering most of the zero vehicle camera traps were located in the forest paths, and roads can host higher activity of the species providing easier movement across the sites. Desensitization of the animals to vehicles passing or temporal partitioning of the space use can also be factors that explain this result. The vehicle activity was high during the day whereas the target species exhibited low activity levels during the day. We were not able to measure the effects of forestry activities and possible poaching.

Forestry activities can have a different impact on mammals compared to transient vehicle movements on the roads. In forestry operations, workers often remain in the forest for extended periods.

The noise generated by forestry work, coupled with prolonged human presence in core areas, may influence the populations. The extensive human use of the area, such as through intensive logging activities, can have other effects on the populations, such as limiting foraging time, exerting stress on populations, and disturbing hunting, mating, and denning behaviors. Studies showed that forestry activities force mammals to leave their habitats and species in a community show different responses, some of them being more sensitive to this type of disturbance (in Düzce, Kahraman, 2019; in Bolu, Naiboğlu, 2016). We can recommend that for healthy management of these populations, there is a need for designating core areas with no forestry activities, where these species can meet their daily energetic requirements, reproduce, and raise cubs. Especially denning sites of the species should not be disturbed and forestry activities should be planned accordingly.

Royle-Nichols abundance estimate models did not show a negative effect of human-related covariates on the abundance of brown bears and other medium and large mammals in our study area. Activity pattern analysis showed that target species reduce their activities to low levels during the day hours when human activities take place. Although we cannot suggest a causal relationship between these two observations, temporal avoidance can be their strategy to avoid humans in the area. Other factors such as genetic adaptation as a result of hundreds of years of persecution, hunting success, and prey behavior could also have influenced the observed patterns, though more research is necessary to confirm their influence.

Large and medium mammals of Yenice forest did not show a preference for YWLDA based on our analysis. We cannot conclude that YWLDA does not provide a safe zone for these species compared to surrounding state hunting grounds, solely based on our models. Variations in other environmental factors between protected and unprotected areas might have obscured the influence of the protected area on species abundance.

The landscape metrics used in the analysis showed that medium and large mammals in our study area prefer the areas with a high diversity of land cover types, higher contrast between the patches of these land cover types, and smaller patches of the same land cover types. To the best of our knowledge landscape metrics were evaluated for the first time in our study for medium and large mammal communities and this, along with the other findings, can be valuable for any forest intervention, whether

it involves delineating areas for human activities or formulating management plans, allowing for informed and strategic planning.

Currently, there are 29 mines in Karabük province, including dolomite, kaolinite, limestone, marble, quartzite, and manganese. There are four hydroelectric power plants and another two are under planning. Mining causes habitat destruction and fragmentation, soil degradation, vegetation suppression, alteration of migration routes, water and air pollution, and noise. Habitat loss extends beyond the boundaries of the mines, as new roads and facilities are constructed for mining operations. Road kills are also an introduced risk for wildlife in these areas. Medium and large mammal populations were shown to decline due to mining (Blanco et al., 2010). Moreover, trace metals that were emitted into the environment from mining centuries ago were found to remain bioavailable to wildlife today (Camizuli et al., 2018). Water resources, as evident also by our models, are vital for all animal populations. Hydroelectric power plants combined with the effects of climate change and destruction caused by their construction can be catastrophic for wild populations, including those in the Yenice Forest, given their dependency on water resources and forest.

The medium and large mammals in our study area showed mostly nocturnal and crepuscular activity. Limiting the human use of forest, including activities such as forestry works, recreational activities, and livestock grazing between 8:00 am and 6:00 pm would reduce the probability of encounters between medium and large mammals and humans, and thus reduce the risk of being harmed or killed for both humans and animals. Human activity in and around protected areas was responsible for 12.32% of all incidents between brown bears and humans in Türkiye (Sıkdokur et al., 2023). Regulating human use of forest, especially within YWLDA is necessary and complies with the purpose of the WLDA for protecting and increasing the population of its targeted species, namely brown bear, roe deer, and red deer.

This study was the first to evaluate the habitat use of brown bears and the medium and large mammal community of Yenice Forest. An assessment of the viability of these populations and a better understanding of the human impact on them requires continued and detailed monitoring of the occurrence and demographic rates of these species. Long-term monitoring of the mammal community combining different non-invasive techniques such as camera trapping, DNA barcoding from scat and hair samples, and examining trends in populations would contribute to effective management of the Yenice WLDA.

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APPENDIX A. ROYLE NICHOLS ABUNDANCE ESTIMATION MODELS FOR BROWN BEAR

Table A.1. Royle-Nichols community abundance estimation model with mean slope within 16 km. radius as abundance parameter.

Abundance	**Estimate**	**SE**	**z**	**P(>|z|)**
(Intercept)	-0.07	0.18	-0.40	0.68
slope_16km	0.33	0.11	2.83	0.00
Detection:				
	Estimate	**SE**	**z**	**P(>|z|)**
(Intercept)	-1.28	0.21	-6.05	0.00
Cam_Loc(path)	-0.80	0.25	-3.12	0.00
AIC:	774.98			

Table A.2. Royle-Nichols community abundance estimation model with mean elevation within 16 km radius as abundance parameter.

Abundance:	**Estimate**	**SE**	**z**	**P(>|z|)**
(Intercept)	-0.01	0.18	-0.09	0.92
elevation_16km	0.28	0.12	2.41	0.01
Detection:				
	Estimate	**SE**	**z**	**P(>|z|)**
(Intercept)	-1.22	0.20	-6	0.00
Cam_Loc(path)	-0.97	0.26	-3.7	0.00
AIC:	777.45			

Table A.3. Royle-Nichols community abundance estimation model with percent landcover by water bodies within 4 km radius as abundance parameter.

Abundance:	**Estimate**	**SE**	**z**	**P(>|z|)**
(Intercept)	-0.14	0.18	-0.75	0.44
1-25%_water_4km	0.85	0.29	2.86	0.00
Detection:				
	Estimate	**SE**	**z**	**P(>|z|)**
(Intercept)	-1.27	0.21	-6.01	0.00
Cam_Loc(path)	-0.77	0.25	-3.01	0.00
AIC:	776.84			

Table A.4. Royle-Nichols community abundance model with and percent landcover by open spaces within 16 km radius as abundance parameter.

Abundance:	**Estimate**	**SE**	**z**	**P(>|z|)**
(Intercept)	-0.33	0.22	-1.51	0.13
1-25%_OpenSpace_16km	0.77	0.22	3.42	0.00
Detection:				
	Estimate	**SE**	**z**	**P(>|z|)**
(Intercept)	-1.33	0.22	-6.06	0.00
Cam_Loc(path)	-0.88	0.25	-3.51	0.00
AIC:	771.70			

Table A.5. Royle-Nichols community abundance model with percent landcover by forest within 8 km radius as abundance parameter.

Abundance:	**Estimate**	**SE**	**z**	**P(>|z|)**
(Intercept)	0.78	0.35	2.21	0.03
76-100%_forest_8km	-0.90	0.35	-2.6	0.00
Detection:				
	Estimate	**SE**	**z**	**P(>|z|)**
(Intercept)	-1.28	0.21	-6.1	0.00
Cam_Loc(path)	-0.72	0.26	-2.78	0.00
AIC:	778.21			

Table A.6. Top models among brown bear Royle-Nichols abundance estimation models with all possible combinations of the final covariates.

r(Int)	N(Int)	p(Cam_Loc)	N(elevation_16km)	N(forest_8km)	lam(OpenSpaces_16km)	lam(Water_4km)	lam(slope_16km)	df	logLikelihood	AIC	ΔAIC	AICweight
-1.42	0.52	-0.83	0.43	+		+	0.29	7.00	-375.89	765.78	0.00	0.19
-1.39	-0.08	-0.89	0.39			+	0.31	6.00	-376.93	765.86	0.08	0.18
-1.40	0.87	-0.85	0.41	+			0.30	6.00	-377.61	767.22	1.44	0.09
-1.42	0.53	-0.84	0.44	+	+	+	0.30	8.00	-375.89	767.77	1.99	0.07
-1.38	-0.06	-0.89	0.42		+	+	0.33	7.00	-376.91	767.81	2.03	0.07
-1.42	0.74	-0.83	0.31	+	+		0.25	7.00	-377.31	768.61	2.83	0.05
-1.41	0.47	-0.76		+	+			5.00	-379.45	768.89	3.11	0.04
-1.37	0.69	-0.85	0.39	+		+		6.00	-378.68	769.37	3.59	0.03
-1.42	0.38	-0.74		+	+		0.15	6.00	-378.76	769.51	3.73	0.03
-1.40	0.72	-0.82	0.18	+	+			6.00	-378.84	769.68	3.90	0.03

APPENDIX B. ROYLE NICHOLS ABUNDANCE ESTIMATION MODELS FOR MEDIUM AND LARGE MAMMALS

Table B.1. Royle-Nichols community abundance model with vehicle capture rate as abundance parameter.

Abundance:	**Estimate**	**SE**	**z**	**P(>|z|)**
(Intercept)	-1.58	0.25	-6.46	0.00
speciesbear	1.20	0.29	4.10	0.00
speciesfox	0.66	0.28	2.39	0.02
specieshare	3.17	2.30	1.38	0.17
speciesjackal	0.83	0.28	2.97	0.00
specieslynx	3.14	2.75	1.14	0.25
speciesmarten	1.05	0.28	3.74	0.00
speciesreddeer	0.50	0.29	1.72	0.08
speciesroedeer	1.14	0.28	4.14	0.00
specieswildboar	1.50	0.32	4.76	0.00
specieswildcat	1.58	0.30	5.25	0.00
specieswolf	-0.14	0.33	-0.42	0.68
VehicleCR(1-100)	0.63	0.12	5.33	0.00
VehicleCR(101-300)	0.68	0.25	2.75	0.01
VehicleCR(301-600)	0.83	0.23	3.62	0.00
VehicleCR(601-1000)	0.66	0.27	2.46	0.01
Detection:	**Estimate**	**SE**	**z**	**P(>|z|)**
(Intercept)	-1.37	0.30	-4.62	0.00
speciesbear	0.00	0.33	0.01	0.99
speciesfox	0.98	0.30	3.24	0.00
specieshare	-5.90	2.35	-2.51	0.01
speciesjackal	0.65	0.31	2.11	0.03
specieslynx	-5.88	2.80	-2.10	0.04
speciesmarten	0.18	0.31	0.58	0.57
speciesreddeer	0.30	0.32	0.95	0.34
speciesroedeer	0.29	0.31	0.93	0.35
specieswildboar	-0.37	0.35	-1.05	0.30
specieswildcat	-0.14	0.34	-0.40	0.69
specieswolf	0.22	0.35	0.62	0.54
VehicleCR(1-100)	-0.15	0.15	-0.99	0.32
VehicleCR(101-300)	-0.19	0.28	-0.70	0.49
VehicleCR(301-600)	0.05	0.28	0.19	0.85
VehicleCR(601-1000)	0.49	0.31	1.58	0.12
Cam_Loc(path)	-0.22	0.12	-1.84	0.07
year	-0.22	0.09	-2.48	0.01
AIC:	7166.19			

Table B.2. Royle-Nichols community abundance model with mean slope within 16km radius as abundance parameter.

Abundance:	**Estimate**	**SE**	**z**	**P(>|z|)**
(Intercept)	-1.21	0.25	-4.91	0.00
speciesbear	1.20	0.31	3.88	0.00
speciesfox	0.57	0.28	2.00	0.05
specieshare	3.25	2.87	1.13	0.26
speciesjackal	0.75	0.29	2.60	0.01
specieslynx	3.28	2.25	1.45	0.15
speciesmarten	1.06	0.29	3.67	0.00
speciesreddeer	0.50	0.30	1.66	0.10
speciesroedeer	1.09	0.28	3.87	0.00
specieswildboar	1.51	0.33	4.58	0.00
specieswildcat	1.65	0.33	4.99	0.00
specieswolf	-0.08	0.35	-0.23	0.82
slope_16km	0.18	0.04	4.64	0.00
Detection:	**Estimate**	**SE**	**z**	**P(>|z|)**
(Intercept)	-1.72	0.30	-5.82	0.00
speciesbear	0.03	0.33	0.08	0.94
speciesfox	1.01	0.30	3.34	0.00
specieshare	-5.91	2.92	-2.03	0.04
speciesjackal	0.69	0.30	2.27	0.02
specieslynx	-5.94	2.31	-2.57	0.01
speciesmarten	0.27	0.31	0.87	0.39
speciesreddeer	0.43	0.32	1.33	0.18
speciesroedeer	0.49	0.30	1.61	0.11
specieswildboar	-0.30	0.36	-0.82	0.42
specieswildcat	-0.19	0.36	-0.55	0.58
specieswolf	0.07	0.35	0.21	0.84
VehicleCR(1-100)	0.34	0.12	2.75	0.01
VehicleCR(101-300)	0.22	0.20	1.09	0.28
VehicleCR(301-600)	0.60	0.21	2.82	0.00
VehicleCR(601-1000)	0.90	0.26	3.45	0.00
Cam_Loc(path)	-0.30	0.12	-2.47	0.01
year	-0.19	0.09	-2.13	0.03
AIC:	7176.064			

Table B.3. Royle-Nichols community abundance model with percent landcover by water bodies within 4 km radius as abundance parameter.

Abundance:	**Estimate**	**SE**	**z**	**P(>|z|)**
(Intercept)	-1.26	0.25	-5.06	0.00
speciesbear	1.23	0.31	3.92	0.00
speciesfox	0.60	0.29	2.10	0.04
specieshare	3.09	2.62	1.18	0.24
speciesjackal	0.72	0.29	2.52	0.01
specieslynx	3.11	2.20	1.42	0.16
speciesmarten	1.01	0.29	3.51	0.00
speciesreddeer	0.47	0.30	1.57	0.12
speciesroedeer	1.08	0.28	3.83	0.00
specieswildboar	1.56	0.34	4.60	0.00
specieswildcat	1.61	0.33	4.95	0.00
specieswolf	-0.14	0.35	-0.40	0.69
water_4km(1-25%)	0.53	0.12	4.54	0.00
Detection:	**Estimate**	**SE**	**z**	**P(>|z|)**
(Intercept)	-1.68	0.30	-5.63	0.00
speciesbear	-0.02	0.34	-0.05	0.96
speciesfox	0.96	0.31	3.16	0.00
specieshare	-5.77	2.67	-2.16	0.03
speciesjackal	0.72	0.31	2.37	0.02
specieslynx	-5.79	2.25	-2.57	0.01
speciesmarten	0.32	0.31	1.02	0.31
speciesreddeer	0.45	0.32	1.39	0.16
speciesroedeer	0.49	0.31	1.59	0.11
specieswildboar	-0.37	0.38	-0.97	0.33
specieswildcat	-0.16	0.35	-0.46	0.64
specieswolf	0.13	0.35	0.37	0.71
VehicleCR(1-100)	0.34	0.13	2.74	0.01
VehicleCR(101-300)	0.25	0.20	1.21	0.23
VehicleCR(301-600)	0.55	0.22	2.49	0.01
VehicleCR(601-1000)	0.90	0.26	3.41	0.00
Cam_Loc(path)	-0.28	0.12	-2.31	0.02
year	-0.28	0.09	-3.10	0.00
AIC:	7179.95			

Table B.4. Royle-Nichols community abundance model with contrast weighed edge density within 8 km radius as abundance parameter.

Abundance:	**Estimate**	**SE**	**z**	**P(>|z|)**
(Intercept)	-1.27	0.25	-5.11	0.00
speciesbear	1.20	0.31	3.87	0.00
speciesfox	0.58	0.28	2.02	0.04
specieshare	3.20	2.73	1.17	0.24
speciesjackal	0.72	0.29	2.53	0.01
specieslynx	3.22	2.35	1.37	0.17
speciesmarten	1.01	0.29	3.52	0.00
speciesreddeer	0.47	0.30	1.58	0.12
speciesroedeer	1.06	0.28	3.79	0.00
specieswildboar	1.56	0.34	4.61	0.00
specieswildcat	1.60	0.32	4.95	0.00
specieswolf	-0.10	0.35	-0.28	0.78
cwed_8kmC	0.38	0.09	4.13	0.00
Detection:	**Estimate**	**SE**	**z**	**P(>|z|)**
(Intercept)	-1.69	0.30	-5.69	0.00
speciesbear	0.02	0.34	0.05	0.96
speciesfox	1.00	0.30	3.29	0.00
specieshare	-5.87	2.78	-2.12	0.03
speciesjackal	0.71	0.31	2.33	0.02
specieslynx	-5.89	2.40	-2.45	0.01
speciesmarten	0.32	0.31	1.02	0.31
speciesreddeer	0.44	0.32	1.39	0.17
speciesroedeer	0.50	0.30	1.65	0.10
specieswildboar	-0.35	0.37	-0.95	0.34
specieswildcat	-0.16	0.35	-0.44	0.66
specieswolf	0.09	0.35	0.27	0.79
VehicleCR(1-100)	0.32	0.12	2.57	0.01
VehicleCR(101-300)	0.21	0.21	1.00	0.32
VehicleCR(301-600)	0.56	0.22	2.60	0.01
VehicleCR(601-1000)	0.91	0.26	3.48	0.00
Cam_Loc(path)	-0.29	0.12	-2.42	0.02
year	-0.23	0.09	-2.54	0.01
AIC:	7182.21			

Table B.5. Royle-Nichols community abundance model with contagion within 8 km radius as abundance parameter.

Abundance:	**Estimate**	**SE**	**z**	**P(>|z|)**
(Intercept)	-0.89	0.26	-3.44	0.00
speciesbear	1.16	0.31	3.75	0.00
speciesfox	0.57	0.29	1.97	0.05
specieshare	3.22	2.40	1.34	0.18
speciesjackal	0.73	0.29	2.51	0.01
specieslynx	3.22	2.39	1.35	0.18
speciesmarten	1.02	0.29	3.50	0.00
speciesreddeer	0.46	0.30	1.52	0.13
speciesroedeer	1.05	0.28	3.72	0.00
specieswildboar	1.51	0.34	4.52	0.00
specieswildcat	1.59	0.33	4.87	0.00
specieswolf	-0.10	0.35	-0.30	0.77
contag_8km(76-100%)	-0.36	0.09	-3.84	0.00
Detection:	**Estimate**	**SE**	**z**	**P(>|z|)**
(Intercept)	-1.72	0.30	-5.75	0.00
speciesbear	0.06	0.34	0.18	0.86
speciesfox	1.00	0.31	3.28	0.00
specieshare	-5.89	2.46	-2.40	0.02
speciesjackal	0.71	0.31	2.29	0.02
specieslynx	-5.89	2.44	-2.42	0.02
speciesmarten	0.31	0.32	0.97	0.33
speciesreddeer	0.46	0.32	1.43	0.15
speciesroedeer	0.51	0.31	1.67	0.09
specieswildboar	-0.30	0.37	-0.82	0.41
specieswildcat	-0.14	0.35	-0.38	0.70
specieswolf	0.10	0.36	0.27	0.79
VehicleCR(1-100)	0.33	0.13	2.66	0.01
VehicleCR(101-300)	0.27	0.20	1.33	0.19
VehicleCR(301-600)	0.58	0.22	2.68	0.01
VehicleCR(601-1000)	0.93	0.26	3.55	0.00
Cam_Loc(path)	-0.28	0.12	-2.27	0.02
year	-0.22	0.09	-2.43	0.02
AIC:	7184.27			

Table B.6. Royle-Nichols community abundance model with Shannon diversity index of vegetation within 8 km radius as abundance parameter.

Abundance:	**Estimate**	**SE**	**z**	**P(>|z|)**
(Intercept)	-1.23	0.25	-4.86	0.00
speciesbear	1.16	0.31	3.79	0.00
speciesfox	0.57	0.29	2.01	0.05
specieshare	3.20	2.20	1.46	0.15
speciesjackal	0.74	0.29	2.57	0.01
specieslynx	3.22	1.90	1.70	0.09
speciesmarten	1.01	0.29	3.52	0.00
speciesreddeer	0.47	0.30	1.58	0.12
speciesroedeer	1.06	0.28	3.76	0.00
specieswildboar	1.53	0.34	4.56	0.00
specieswildcat	1.60	0.32	4.93	0.00
specieswolf	-0.09	0.35	-0.26	0.79
shdi_8km(26-50%)	-0.10	0.09	-1.07	0.29
shdi_8km(51-75%)	0.19	0.11	1.78	0.08
shdi_8km(76-100%)	0.43	0.14	3.08	0.00
Detection:	**Estimate**	**SE**	**z**	**P(>|z|)**
(Intercept)	-1.70	0.30	-5.71	0.00
speciesbear	0.06	0.33	0.17	0.87
speciesfox	1.00	0.30	3.31	0.00
specieshare	-5.88	2.25	-2.61	0.01
speciesjackal	0.70	0.31	2.28	0.02
specieslynx	-5.90	1.96	-3.01	0.00
speciesmarten	0.31	0.31	1.00	0.32
speciesreddeer	0.44	0.32	1.37	0.17
speciesroedeer	0.51	0.31	1.67	0.10
specieswildboar	-0.33	0.37	-0.88	0.38
specieswildcat	-0.15	0.35	-0.42	0.67
specieswolf	0.09	0.36	0.25	0.80
VehicleCR(1-100)	0.32	0.13	2.55	0.01
VehicleCR(101-300)	0.21	0.20	1.02	0.31
VehicleCR(301-600)	0.55	0.22	2.47	0.01
VehicleCR(601-1000)	0.89	0.27	3.35	0.00
Cam_Loc(path)	-0.28	0.12	-2.33	0.02
year	-0.21	0.09	-2.32	0.02
AIC:	7186.12			

Table B.7. Royle-Nichols community abundance model with percent landcover by forest within 8 km radius as abundance parameter.

Abundance:	**Estimate**	**SE**	**z**	**P(>|z|)**
(Intercept)	-0.80	0.28	-2.89	0.00
speciesbear	1.14	0.31	3.74	0.00
speciesfox	0.56	0.29	1.97	0.05
specieshare	3.18	2.72	1.17	0.24
speciesjackal	0.73	0.29	2.55	0.01
specieslynx	3.20	2.29	1.40	0.16
speciesmarten	1.02	0.29	3.54	0.00
speciesreddeer	0.48	0.30	1.60	0.11
speciesroedeer	1.07	0.28	3.78	0.00
specieswildboar	1.48	0.33	4.52	0.00
specieswildcat	1.57	0.32	4.90	0.00
specieswolf	-0.09	0.35	-0.27	0.79
Forest_8km(76-100%)	-0.43	0.14	-3.08	0.00
Detection:	**Estimate**	**SE**	**z**	**P(>|z|)**
(Intercept)	-1.70	0.30	-5.68	0.00
speciesbear	0.08	0.33	0.24	0.81
speciesfox	1.01	0.30	3.34	0.00
specieshare	-5.85	2.77	-2.12	0.03
speciesjackal	0.70	0.31	2.27	0.02
specieslynx	-5.87	2.34	-2.51	0.01
speciesmarten	0.29	0.31	0.94	0.35
speciesreddeer	0.43	0.32	1.32	0.19
speciesroedeer	0.49	0.31	1.61	0.11
specieswildboar	-0.28	0.36	-0.76	0.45
specieswildcat	-0.12	0.35	-0.36	0.72
specieswolf	0.08	0.36	0.24	0.81
VehicleCR(1-100)	0.31	0.13	2.48	0.01
VehicleCR(101-300)	0.24	0.20	1.18	0.24
VehicleCR(301-600)	0.51	0.22	2.31	0.02
VehicleCR(601-1000)	0.85	0.27	3.20	0.00
Cam_Loc(path)	-0.29	0.12	-2.41	0.02
year	-0.19	0.09	-2.08	0.04
AIC:	7189.45			

Table B.8. Royle-Nichols community abundance model with patch density in 8 km radius as abundance parameter.

Abundance:	**Estimate**	**SE**	**z**	**P(>|z|)**
(Intercept)	-1.22	0.25	-4.86	0.00
speciesbear	1.14	0.31	3.71	0.00
speciesfox	0.56	0.29	1.94	0.05
specieshare	3.28	1.70	1.93	0.05
speciesjackal	0.73	0.29	2.53	0.01
specieslynx	3.22	2.59	1.24	0.22
speciesmarten	1.02	0.29	3.50	0.00
speciesreddeer	0.47	0.30	1.56	0.12
speciesroedeer	1.06	0.28	3.73	0.00
specieswildboar	1.47	0.33	4.48	0.00
specieswildcat	1.57	0.32	4.86	0.00
specieswolf	-0.09	0.35	-0.25	0.80
patchdensity_4km(26-50%)	0.36	0.12	2.96	0.00
Detection:	**Estimate**	**SE**	**z**	**P(>|z|)**
(Intercept)	-1.71	0.30	-5.70	0.00
speciesbear	0.09	0.33	0.26	0.80
speciesfox	1.01	0.30	3.33	0.00
specieshare	-5.95	1.78	-3.35	0.00
speciesjackal	0.70	0.31	2.26	0.02
specieslynx	-5.89	2.64	-2.23	0.03
speciesmarten	0.31	0.31	0.98	0.33
speciesreddeer	0.44	0.32	1.37	0.17
speciesroedeer	0.51	0.31	1.66	0.10
specieswildboar	-0.26	0.36	-0.73	0.47
specieswildcat	-0.13	0.35	-0.36	0.72
specieswolf	0.08	0.36	0.21	0.83
VehicleCR(1-100)	0.32	0.12	2.54	0.01
VehicleCR(101-300)	0.25	0.20	1.26	0.21
VehicleCR(301-600)	0.54	0.22	2.45	0.01
VehicleCR(601-1000)	0.88	0.26	3.34	0.00
Cam_Loc(path)	-0.30	0.12	-2.47	0.01
year	-0.19	0.09	-2.12	0.03
AIC:	7189.97			

Table B.9. Royle-Nichols community abundance model with percent landcover by coniferous forest within 1 km radius as abundance parameter.

Abundance:	**Estimate**	**SE**	**z**	**P(>|z|)**
(Intercept)	-1.17	0.25	-4.66	0.00
speciesbear	1.14	0.31	3.68	0.00
speciesfox	0.52	0.29	1.81	0.07
specieshare	3.18	2.91	1.09	0.28
speciesjackal	0.71	0.29	2.45	0.01
specieslynx	3.19	2.41	1.33	0.18
speciesmarten	1.00	0.29	3.43	0.00
speciesreddeer	0.45	0.30	1.49	0.14
speciesroedeer	1.06	0.29	3.71	0.00
specieswildboar	1.48	0.33	4.45	0.00
specieswildcat	1.58	0.33	4.82	0.00
specieswolf	-0.11	0.36	-0.30	0.77
pland_conifer_1kmB	0.39	0.17	2.26	0.02
pland_conifer_1kmE	-0.89	0.50	-1.77	0.08
Detection:	**Estimate**	**SE**	**z**	**P(>|z|)**
(Intercept)	-1.75	0.30	-5.80	0.00
speciesbear	0.09	0.33	0.25	0.80
speciesfox	1.05	0.31	3.44	0.00
specieshare	-5.84	2.96	-1.98	0.05
speciesjackal	0.72	0.31	2.31	0.02
specieslynx	-5.85	2.46	-2.38	0.02
speciesmarten	0.33	0.32	1.05	0.30
speciesreddeer	0.47	0.32	1.46	0.14
speciesroedeer	0.51	0.31	1.65	0.10
specieswildboar	-0.26	0.37	-0.71	0.48
specieswildcat	-0.13	0.35	-0.36	0.72
specieswolf	0.10	0.36	0.27	0.79
VehicleCR(1-100)	0.33	0.13	2.66	0.01
VehicleCR(101-300)	0.22	0.21	1.09	0.27
VehicleCR(301-600)	0.61	0.22	2.86	0.00
VehicleCR(601-1000)	0.96	0.26	3.68	0.00
Cam_Loc(path)	-0.30	0.12	-2.48	0.01
year	-0.19	0.09	-2.11	0.03
AIC:	7190.94			

Table B.10. Royle-Nichols community abundance model with and percent landcover by urban within 8km radius as abundance parameter.

Abundance:	**Estimate**	**SE**	**z**	**P(>|z|)**
(Intercept)	-1.25	0.25	-4.96	0.00
speciesbear	1.14	0.31	3.73	0.00
speciesfox	0.55	0.29	1.92	0.05
specieshare	3.32	2.81	1.18	0.24
speciesjackal	0.73	0.29	2.53	0.01
specieslynx	3.34	2.46	1.36	0.18
speciesmarten	1.04	0.29	3.58	0.00
speciesreddeer	0.47	0.30	1.56	0.12
speciesroedeer	1.06	0.28	3.75	0.00
specieswildboar	1.49	0.33	4.51	0.00
specieswildcat	1.58	0.33	4.88	0.00
specieswolf	-0.10	0.35	-0.28	0.78
urban_8km(1-25%)	0.17	0.08	2.20	0.03
Detection:	**Estimate**	**SE**	**z**	**P(>|z|)**
(Intercept)	-1.73	0.30	-5.81	0.00
speciesbear	0.08	0.33	0.25	0.81
speciesfox	1.02	0.30	3.37	0.00
specieshare	-5.98	2.86	-2.10	0.04
speciesjackal	0.70	0.31	2.28	0.02
specieslynx	-6.00	2.51	-2.39	0.02
speciesmarten	0.29	0.31	0.91	0.36
speciesreddeer	0.45	0.32	1.40	0.16
speciesroedeer	0.51	0.31	1.68	0.09
specieswildboar	-0.28	0.36	-0.76	0.45
specieswildcat	-0.13	0.35	-0.37	0.71
specieswolf	0.09	0.35	0.25	0.81
VehicleCR(1-100)	0.33	0.12	2.67	0.01
VehicleCR(101-300)	0.27	0.20	1.32	0.19
VehicleCR(301-600)	0.59	0.22	2.73	0.01
VehicleCR(601-1000)	0.92	0.26	3.52	0.00
Cam_Loc(path)	-0.29	0.12	-2.39	0.02
year	-0.18	0.09	-2.06	0.04
AIC:	7193.14			

Table B.11. Royle-Nichols community abundance model with and percent landcover by open spaces within 2km radius as abundance parameter.

Abundance:	**Estimate**	**SE**	**z**	**P(>|z|)**
(Intercept)	-1.15	0.25	-4.59	0.00
speciesbear	1.12	0.31	3.65	0.00
speciesfox	0.51	0.29	1.78	0.07
specieshare	3.23	6.12	0.53	0.60
speciesjackal	0.74	0.29	2.53	0.01
specieslynx	3.35	2.50	1.34	0.18
speciesmarten	1.03	0.29	3.52	0.00
speciesreddeer	0.46	0.30	1.52	0.13
speciesroedeer	1.07	0.29	3.76	0.00
specieswildboar	1.43	0.33	4.41	0.00
specieswildcat	1.56	0.32	4.83	0.00
specieswolf	-0.09	0.35	-0.25	0.80
OpenSpaces_2km(1-255)	-1.22	0.71	-1.71	0.09
Detection:	**Estimate**	**SE**	**z**	**P(>|z|)**
(Intercept)	-1.81	0.30	-6.08	0.00
speciesbear	0.11	0.33	0.34	0.74
speciesfox	1.06	0.30	3.51	0.00
specieshare	-5.91	6.14	-0.96	0.34
speciesjackal	0.67	0.31	2.17	0.03
specieslynx	-6.03	2.55	-2.37	0.02
speciesmarten	0.31	0.31	0.97	0.33
speciesreddeer	0.46	0.32	1.43	0.15
speciesroedeer	0.50	0.31	1.63	0.10
specieswildboar	-0.21	0.36	-0.58	0.56
specieswildcat	-0.10	0.35	-0.30	0.77
specieswolf	0.07	0.35	0.19	0.85
VehicleCR(1-100)	0.33	0.13	2.63	0.01
VehicleCR(101-300)	0.18	0.20	0.89	0.37
VehicleCR(301-600)	0.56	0.21	2.64	0.01
VehicleCR(601-1000)	0.86	0.26	3.38	0.00
Cam_Loc(path)	-0.31	0.12	-2.54	0.01
AIC:	7193.84			

Table B.12. Royle-Nichols community abundance model with and percent landcover by by agriculture within 8km radius as abundance parameter.

Abundance:	**Estimate**	**SE**	**z**	**P(>|z|)**
(Intercept)	-0.97	0.27	-3.58	0.00
speciesbear	1.11	0.31	3.61	0.00
speciesfox	0.52	0.29	1.81	0.07
specieshare	3.28	2.83	1.16	0.25
speciesjackal	0.73	0.29	2.50	0.01
specieslynx	3.30	2.26	1.47	0.14
speciesmarten	1.01	0.29	3.45	0.00
speciesreddeer	0.45	0.30	1.48	0.14
speciesroedeer	1.05	0.29	3.68	0.00
specieswildboar	1.42	0.33	4.35	0.00
specieswildcat	1.60	0.33	4.82	0.00
specieswolf	-0.11	0.36	-0.30	0.76
Agriculture_8km(1-25%)	-0.22	0.11	-2.05	0.04
Detection:	**Estimate**	**SE**	**z**	**P(>|z|)**
(Intercept)	-1.76	0.30	-5.82	0.00
speciesbear	0.11	0.33	0.34	0.73
speciesfox	1.04	0.31	3.41	0.00
specieshare	-5.94	2.88	-2.06	0.04
speciesjackal	0.69	0.31	2.23	0.03
specieslynx	-5.97	2.31	-2.58	0.01
speciesmarten	0.33	0.32	1.03	0.30
speciesreddeer	0.48	0.32	1.46	0.14
speciesroedeer	0.53	0.31	1.70	0.09
specieswildboar	-0.20	0.36	-0.54	0.59
specieswildcat	-0.15	0.36	-0.42	0.67
specieswolf	0.09	0.36	0.25	0.80
VehicleCR(1-100)	0.35	0.13	2.77	0.01
VehicleCR(101-300)	0.31	0.20	1.52	0.13
VehicleCR(301-600)	0.64	0.21	2.98	0.00
VehicleCR(601-1000)	0.97	0.26	3.72	0.00
Cam_Loc(path)	-0.30	0.12	-2.48	0.01
year	-0.20	0.09	-2.29	0.02
AIC:	7193.95			

Table B.13. Top models among medium and large mammals Royle-Nichols abundance estimation models with all possible combinations of the final covariates

r(Int)	N(Int)	r(Cam_Loc)	N(species)	N(Traffic)	r(year)	N(contag_8km)	N(cwed_4km)	N(pland_6_4km)	N(slope_16km)	N(species)	N(Traffic)	df	logLikelihood	AIC	ΔAIC	AICweight
-1.38	-1.63	-0.18	+		-0.25			+	0.16	+	+	32.00	-3537.73	7139.46	0.00	0.23
-1.47	-1.67		+		-0.26			+	0.16	+	+	31.00	-3539.19	7140.39	0.93	0.15
-1.39	-1.53	-0.18	+		-0.25	+		+	0.15	+	+	33.00	-3537.19	7140.39	0.93	0.15
-1.47	-1.56		+		-0.25	+		+	0.15	+	+	32.00	-3538.63	7141.25	1.79	0.10
-1.41	-1.42	-0.17	+		-0.20	+			0.16	+	+	32.00	-3539.39	7142.78	3.32	0.04
-1.34	-1.64	-0.21	+	+	-0.27			+	0.16	+	+	36.00	-3535.50	7143.01	3.55	0.04
-1.49	-1.45		+		-0.20	+			0.16	+	+	31.00	-3540.78	7143.55	4.09	0.03
-1.35	-1.54	-0.21	+	+	-0.26	+		+	0.15	+	+	37.00	-3534.99	7143.99	4.53	0.02
-1.52	-1.64		+	+	-0.28			+	0.16	+	+	35.00	-3537.03	7144.06	4.60	0.02
-1.39	-1.63	-0.18	+		-0.25		+	+	0.15	+	+	35.00	-3537.27	7144.53	5.07	0.02