



# Report on monitoring Eurasian lynx using camera trapping in the Romanian Carpathians

**Authors:** Ruben Iosif<sup>1</sup>, Viorel Popescu<sup>2,3</sup>, Liviu Ungureanu<sup>1</sup>, Călin Șerban<sup>1</sup>, Bogdan Sulică<sup>1</sup>, Lavinia Terciu<sup>1</sup>, Daniel Bîrloiu<sup>1</sup>, Claudiu Țoanță<sup>1</sup>, Barbara Promberger–Fürpass<sup>1</sup>

**Authors affiliation:**

1 – Foundation Conservation Carpathia, 27 Calea Feldioarei, 500471 Brașov, Romania;

2 – Department of Biological Sciences, Ohio University, 107 Irvine Hall, Athens, OH 45701, USA;

3 – Centre for Environmental Research (CCMESI), University of Bucharest, 1 N. Bălcescu, 010041 Bucharest, Romania;

**Contacts:**

Ruben Iosif: [r.iosif@carpathia.org](mailto:r.iosif@carpathia.org)

Barbara Promberger: [b.promberger@carpathia.org](mailto:b.promberger@carpathia.org)

Viorel Popescu: [popescu@ohio.edu](mailto:popescu@ohio.edu)

## Contents

1. Introduction .....	4
2. Study area .....	7
3. Methods .....	9
3.1. Monitoring sessions .....	9
3.2. Sampling design.....	10
3.3. Camera trap installation .....	10
3.4. Cataloguing camera trap images .....	12
3.5. Modelling population estimates.....	14
3.5.1. Encounter histories and input files .....	14
3.5.2. Modelling detection .....	14
3.5.3. Habitat – non habitat mask.....	15
3.5.4. Modelling population size and density .....	16
3.5.5. Density maps.....	16
4. Results .....	18
4.1. Camera functionality .....	18
4.2. Image analysis and capture history .....	20
4.3. Population parameters estimates .....	25
4.3.1. Regional population size .....	25
4.3.2. Population density .....	25
4.4. Predictors for density maps .....	25
5. Discussion .....	28
References .....	32
Acknowledgements.....	37

# Chapter 1

## Introduction



Photo © FCC

The viability of large carnivore populations across the European continent varies, and is largely driven by the economic status of different EU member nations (Kojola *et al.*, 2018). In general, the European Habitats Directive legislation enabled the recolonization of former carnivore ranges in human dominated landscapes (Chapron *et al.*, 2014). For example, protection from hunting and poaching and a strict conservation status allowed individual wolves to disperse thousands of kilometres over administrative boundaries in search of new territories (Mulej *et al.*, 2013). However, in the case of felids, there is limited evidence of the population trends at large geographic scales due to their secretive ecology and behaviour. Existing evidence points to the fact that natural recovery of felid populations is reduced in parts of their range where they have been extirpated from, low dispersal and demographic parameters limiting expansion and population growth (Goana, Ferreras & Delibes, 1998; Molinari-Jobin *et al.*, 2010). For example, population trend of the European wildcat (*Felix sylvestris*) in southern Spain is uncertain as habitat fragmentation continues to disrupt connectivity between the main populations (Gil-Sánchez *et al.*, 2020). The Iberian lynx too (*Lynx pardinus*) has severe difficulties in maintaining a viable population, revealing an urgent need to protect the species and its habitats outside of reserves (Garrote *et al.*, 2020).

The Eurasian lynx population was estimated at approx. 7200 individuals across Europe, but with little consistency on how national numbers were reached, many countries relying on uncertain, non-scientific, census methods (Breitenmoser *et al.*, 2000). The Eurasian lynx is facing increasing pressure from habitat degradation, reduced prey availability or high human-related mortality (Breitenmoser-Würsten *et al.*, 2008). Under these circumstances, lynx faced severe declines during the 20th century in ecosystems such as the Alps, Jura and Dinaric Mountains, where it became locally extinct. Wildlife managers have been developing reintroduction programs there using the central and eastern European populations as a source (Linnell *et al.*, 2009), but having limited success so far (Mueller *et al.*, 2020; Vandel *et al.*, 2006). On the other hand, countries missing solid population data faced impediments for sustainable conservation actions putting populations at risk through exaggerated hunting quotas of lynx prey base, or habitat loss through extensive deforestation. (Rozyłowicz *et al.*, 2011; Popescu *et al.*, 2016).

The Romanian Carpathians represent one of the last places in Europe where the entire large mammal community is present in good numbers, lynx co-occurring with bears and wolves (Breitenmoser *et al.*, 2000). They are characterized by a compact forest ecosystem interspersed with traditional agricultural landscapes, which together provide the habitat, food resource and space requirements of Eurasian lynx, as well as its large carnivore competitors and their ungulate prey (Promberger-Fürpass & Sürth, 2003; Rozyłowicz *et al.*, 2010; Popescu *et al.*, 2017; Pop *et al.*, 2018; Fedorca *et al.*, 2019; Sin *et al.*, 2019; Iosif *et al.*, 2020). Although the Romanian Carpathians provide a good example of a long-term populations' viability, co-occurrence between species and coexistence with human populations, the Romanian Carpathians are surprisingly one of the least studied mountain ecosystems in Europe. No science-based population monitoring is available so far for the Eurasian lynx here. Popescu *et al.* (2016) showed the officially reported data on Eurasian lynx in Romania likely underestimated population trend in comparison with simulated trends from other European populations, while the official data showed a perfect, yet questionable, linear increase in the lynx population (Cazacu *et al.*, 2014). Beside these uncertainties around population size, there are gaps in our knowledge of movement and habitat selection (Promberger-Fürpass, B., Sürth & Predoiu, 2002; Rozyłowicz *et al.*, 2010). The rapidly changing environment of the Romanian Carpathians due to logging and infrastructure development provides the impetus for understanding the responses of carnivores and their prey base to habitat fragmentation. As such, developing a baseline for population size and habitat selection is pre-requisite for evaluating long-term population viability of Eurasian lynx in Romania, and establishing a robust population monitoring program (Breitenmoser *et al.*, 2000). The information is also critical to safeguard this seemingly viable population, as it currently acts as one of the sources for natural recolonization of other areas, as well as for ongoing reintroduction programs (see [www.lifelynx.eu](http://www.lifelynx.eu)). In the context of reintroduction programs it is important to provide long term monitoring and population estimates for the source populations as well (Linnell *et al.*, 2009), particularly in countries where the officially reported numbers are biologically unrealistic (Kubala *et al.*, 2019; Popescu, Artelle, Pop, Manolache, & Rozyłowicz, 2016).

To understand species response to a changing environment, scientists and conservation practitioners need robust population estimates (Artelle *et al.*, 2014). A prerequisite for obtaining robust estimates is to capture, mark and recapture individuals in a population. Because physical capture and recapture of animals is time and cost intensive, and induces stress in captured animals, camera trapping has emerged as an effective and non-invasive method for capture-recapture techniques wherever individuals can be identified via unique patterns (Royle *et al.*, 2014). Most members of the cat (*Felidae*) family are known for their unique coat, and the variation of coat patterns of different felid species, including the definition of discrete categories of pelage patterns, has been described (Werdelin & Olsson, 1997). When applied to European lynx, fur patterns have proven useful for identifying individuals and constructing robust encounter histories for capture-recapture studies (Zimmermann *et al.*, 2013). Camera trapping was widely used for lynx in relatively short, up to 3 months

sessions (e.g., Kubala et al., 2019), but also on the much more elusive wildcat in longer monitoring sessions (up to three years monitoring in Sicily; Anile, Amico, & Ragni, 2012). Di Bitetti, Paviolo, & De Angelo, (2006) applied a similar long monitoring of 1400 trap days for the ocelot (*Leopardus pardalis*) living in the dense Atlantic forest in Argentina. To model population estimates from camera trapping encounters, scientists first used non-spatial capture-recapture models, which successfully predicted population parameters for a wide range of organisms, and influenced management decisions worldwide for more than three decades (Williams, Nichols & Conroy, 2002; Pollock, 2000). However, robustness of these methods for determining animal density may be low due to the artificial delineation of study areas and an increasing recognition of biases induced by edge effects (i.e., quantified as the exchange of individuals with the highly suitable habitat patches from outside of the established monitoring area). In continuous forest habitat of the Romanian Carpathians, this edge effect poses challenges to non-spatial capture recapture (Hupman *et al.*, 2018), especially for species with high movement capabilities (Keiter *et al.*, 2017). In such cases, scientists used independent telemetry data to calculate the effective sampling area, by controlling for potential movement distances (O'Connell, Nichols, & Karanth, 2010). When movement data was not available, scientists have derived ways of bootstrapping the moved distance from DNA sample-revealed movement, and corrected the local population estimates (Jerina, Jonozovič, Krofel, & Skrbinšek, 2013; Skrbinšek *et al.*, 2012). Recent research however, criticizes these approaches to indirectly estimate the effective sampling area, as the density estimates are very sensitive on varying distance choices (Royle *et al.*, 2014). The alternative is to use Spatially Explicit Capture Recapture models (hereafter SECR), a recent extension of classical capture-recapture models applicable to a wide range of study designs, including camera trapping (Jimenez *et al.*, 2019). Royle *et al.* (2014), explain that spatial variation in density, resource selection, space usage, and movement, can be assessed starting from encounter histories that are associated with information on the location of capture (i.e., spatial encounter histories). SECR predicts density as posterior probability distributions around the activity centres of each separate individual in the studied population. These predictions are of course dependent on the number of recaptures of each individual (Royle *et al.*, 2014).

In this study, we aimed to demonstrate the feasibility of camera trapping as a monitoring tool for the elusive Eurasian lynx in the Romanian Carpathians, by using SECR models to estimate density and population size. We also mapped seasonal hotspots of population density and evaluated predictors for seasonal variation of density hotspots. Specifically, we tested the following hypotheses:

- i. autumn-early winter monitoring will return better population estimates when compared to winter monitoring (which includes the full mating season), as the autumn-early winter home ranges are more stable and edge effect is less prevalent. For example, similar studies conducted in the Bohemian Forest showed that autumn monitoring was preferable, as stable home ranges increased recapture rate of the local population (Weingarth *et al.*, 2015).
- ii. density hotspots shift from autumn-early winter to winter according to topography and habitat structure (i.e., from the compact forest habitat to the mosaic of traditional agricultural landscape and vice versa). Our field observation and other habitat selection studies (e.g., Filla *et al.*, 2017) suggest that lynx occupancy is higher in the traditional agricultural landscape during winter, likely because these landscape attract roe deer due to lower snow cover.

## Chapter 2

# Study area



Photo © Călin Șerban

The study area is situated in the Southern Carpathians, Romania, covering 1200 km<sup>2</sup> in the eastern corner of the Făgăraș Mountains, Piatra Craiului, and parts of Leaota Mountains. Ranging in altitude between 600 and 2400 m (Figure 1), it includes a national park (i.e., Piatra Craiului National Park), and overlaps with four Natura 2000 sites of community importance. Forests cover most of the area (62%), followed by a mosaic of urban-rural landscape and agriculture with significant areas of natural vegetation (22%), and alpine grasslands and subalpine shrubs (16%).

Deciduous, coniferous and mixed forest have now equal proportions (22, 21 and 19%). Spruce (*Picea abies*) and fir (*Abies alba*) dominate higher elevations. Mixed forests are dominated by beech-fir or beech-fir-spruce and cover mid slopes. Lower slopes are mostly covered by beech (*Fagus sylvatica*). Transitional woods and shrubs are dominated by *Pinus mugo* and *Vaccinium subsp.* Forest management historically replaced significant areas with spruce monocultures. In the last three decades, the area was affected by chaotic deforestation, with frequent clear-cuts exceeding the three ha maximum-allowed patch (Kuemmerle *et al.*, 2009). These clear-cuts are now regenerating into a young forest with abundant understorey vegetation, potentially providing food and shelter for wildlife. The mosaic of traditional agricultural habitats with significant

areas of natural vegetation consists of patches of traditionally managed hayfields, orchards and crops separated by dense forest edges and shrubs providing good connectivity with the compact forest patches. Although bisected by a high traffic national road (DN73) along which localities are distributed, the area is recognized as a corridor for large carnivores' dispersal, with no major barriers outside the mountain ranges. The road network is dominated by unpaved forest roads and temporary logging roads.

The large mammal community is still intact throughout the study area, and composed of the three European large carnivores, wolf (*Canis lupus*), brown bear (*Ursus arctos*), and Eurasian lynx (*Lynx lynx*), as well as their main prey wild boar (*Sus scrofa*), roe deer (*Capreolus capreolus*), red deer (*Cervus elaphus*), and chamois (*Rupicapra rupicapra*) in the alpine areas. Hunting of large carnivores and chamois is banned while ungulates are hunted regularly only in the northern part of the study area and limited to extraction of conflict animals (wild boar) in the southern part. Logging still remains to be an important economic activity and is executed year-round. Grazing is another source of human impact, especially in the alpine areas, whereas lowlands are characterized by small scale traditional farming and tourism development.

Wildlife management is organised into eight different game management units (GMU), four of which are administrated by CARPATHIA (a private conservation initiative for the Făgăraș Mountains, consisting of several legal entities), and the remaining four being under the control of three different hunting clubs. Collaboration, especially in the northern part, was positive, and local hunters shared valuable information to identify lynx trap locations.

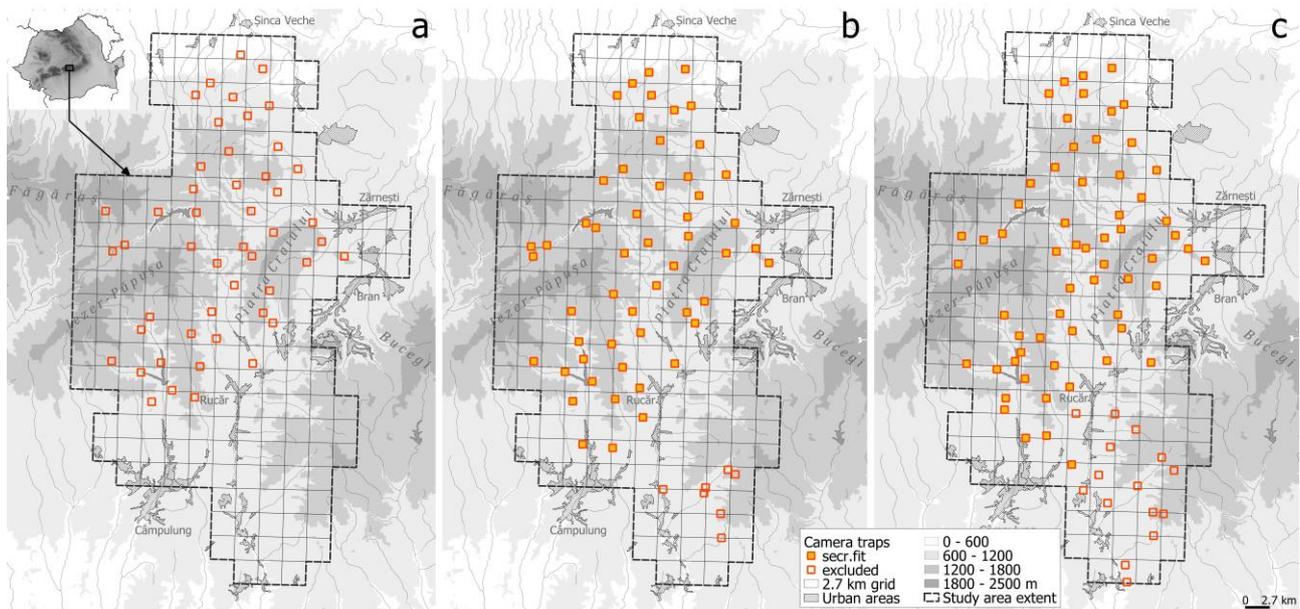


Figure 1. Study area for camera trapping of Eurasian lynx in the Southern Carpathians, Romania. Panel (a) represents a pilot monitoring session with 48 camera traps functioning between 24.02 – 04.04.2018. Panel (b) represents the winter session with 59 traps functioning between 17.12.2018 – 31.03.2019. Panel (c) represents the autumn-early winter session with 76 traps functioning between 09.10.2019 – 15.01.2020. See Methods for full description and rationale behind these sessions. Note that the majority of traps overlap from one session to another and symbology varies between traps, one for those used for fitting spatially explicit capture recapture models, and one for those excluded from modelling due to insufficient data.

## Chapter 3

# Methods



Photo © Liviu Ungureanu

### 3.1. Monitoring sessions

To achieve robust density estimates, scientists and conservation practitioners have to evaluate the optimal monitoring session. Establishing the monitoring session length is a trade-off between population closure and the number of recaptures needed for density estimates, this rule applying in particular for the highly elusive lynx. Furthermore, we had to evaluate the optimal time window within the year in respect to species movement, reproduction, etc. After monitoring a lynx population in the Bohemian Forest, at the border between Germany and Czech Republic, Weingarth et al. (2015) suggested the following session parameters: *i.* sufficient recaptures session length = 80 days; and *ii.* maximum recaptures session length = 120 days, and found the optimal time window for monitoring between September and November.

We first performed a short, pilot study, and then ran two full monitoring sessions:

- i. 24 February – 04 April 2018 → 40 days → 8 sampling occasions (one sampling occasion was set at 5 days), hereafter pilot session;
- ii. 17 December 2018 – 31 March 2019 → 105 days → 21 sampling occasions, hereafter winter session;

- iii. 09 October 2019 – 16 January 2020 → 100 days → 20 sampling occasions; hereafter autumn-early winter session.

The pilot session was set for the purpose of identifying lynx movement routes and for testing camera settings in different conditions at the installation sites. The winter session was set for the purpose of maximizing lynx detection given the scarce data we had on species movement in the study area (and in the Romanian Carpathians in general). Thus, we waited for a compact snow cover and used animal tracks on snow to improve the set-up of the camera traps from the previous session. We fully included the reproduction season (end of February till early April) in the monitoring given the species' higher mobility during that time (Jędrzejewski *et al.*, 2002), with the same rationale of maximizing detectability detrimental to the population closure assumption. The rationale behind setting the autumn-early winter session in the pre-mating season was to address demographic population closure for more robust density estimates, attempting to be in accordance with the findings of Weingarth *et al.* (2015) on optimal monitoring time window and length. Demographic population closure is expected during the autumn-early winter given the species movement ecology, with males being less active and having more stable home ranges as during the mating season, and with females with juveniles being fully mobile compared with summer when they seek shelter around the den (Signer, 2017).

### 3.2. Sampling design

We replicated a sampling design previously used on the same species in similar habitat conditions and with similar population density (1-2 individuals per 100 km<sup>2</sup>) (Zimmermann *et al.*, 2013). We divided our study area into a grid of 2.7 × 2.7 km cells and removed from sampling the cells with more than 2/3 of their area exceeding 1800 m altitude (Zimmermann *et al.*, 2013). We never set cameras above the tree line as we expect the species uses alpine or subalpine habitats only occasionally and detection probability would be low there. We also removed cells that only partially overlap our study area (with the remaining territory having low probability of including suitable lynx habitat). From the remaining territory, we sampled every second cell, resulting in a final predefined trap array of 78 cells. We used this predefined trap array as a reference for installing the cameras in all monitoring sessions. When it was not possible to reach a selected cell, we used an adjacent cell (Zimmermann *et al.*, 2013). The pilot session has 48 traps, the winter session had 59, while the autumn-early winter session had 76 traps installed in the predefined trap array, their majority overlapping between sessions (Figure 1).

Note that the trap array increased spatially from north to south between sessions with the last session having the highest density of traps (Figure 1). Not all traps were used for fitting spatially explicit capture recapture models, a subset being excluded from modelling due to insufficient reliable detections (see chapter 4.2 for clarifications; Figure 1).

### 3.3. Camera trap installation

Each trap station had two opposite cameras installed on trees at a height of 40 to 60 cm and pointing towards the expected animal paths in an almost rectangular angle to obtain best images from the side of the animal. We used two camera models, a CuddeBack C1 Model 1279 with white flash for high quality colour pictures in night conditions, and a Bushnell Trophy Cam infrared camera. The setup at the trap location was either one white flash camera facing one infrared camera (approx. 25% of the traps), or two white flash cameras facing each other (approx. 75% of the traps). Their fields of view were not perfectly aligned to avoid overexposing the picture of the opposite camera when triggered simultaneously. The distance between cameras and the expected animal path was set at about 5 m, varying depending on topography.

Altitude of the traps varied between 663 and 1600 m during the winter session, and between 788 and 1617 m during the autumn-early winter session, thus relatively constant across monitoring sessions. We installed the camera traps on animal trails along mountain ridges as the first choice, i.e., 53% of the traps during the winter session and 64% during the autumn-early winter session, as the ridges trails are frequently used not only by lynx but also by wolf, bears and ungulates. In rough terrain or where snow cover limited accessibility, we installed the cameras on mid-slopes, mostly on temporary logging roads, i.e., 28% during the winter session and 18% of the camera traps during the autumn-early winter session. We also installed traps along upper valleys, mostly on unpaved forestry roads adjacent to the rivers, i.e., 19% during the winter session and 18% during the autumn-early winter session.

The dominant habitat around trap locations was mostly mixed forests (54% - winter, 57% - autumn-early winter session), followed by deciduous (24 and 21% respectively), and coniferous forest (22 and 22% respectively). We expected a higher lynx density in mixed forests, which provides abundant understorey vegetation, thus food and shelter for both the species and its prey (mostly roe deer) (Filla *et al.*, 2017).

Camera installation and synchronization last for 1-2 weeks prior to the start of the monitoring session. We selected open spots with no grasses and branches that could trigger the cameras in wind or snow conditions. Wherever needed, we also cleared the grasses and branches in front of the camera while attempting to keep the impact on animal behaviour to a minimum. We selected spots with relatively flat topography to capture the animal in normal walking. Each trap station was checked on average every two weeks in respect to weather condition (more checks in periods of heavy snow or strong wind). Checking the traps involved replacing the SD cards, batteries when needed, double checking the settings, removing ice, snow and branches that blocked the field of view. We also adjusted camera height on the trees depending on snow cover. When logging sites were opened close to our active trap stations, we changed the location within the same grid cell to avoid reducing detection probability through human disturbances. In some occasions, we also re-located trap stations if a more promising location was found within the same grid cell. Such, 28.8% of the camera stations were moved in the winter session and 14.5% in the autumn-early winter session. We used lithium batteries, which perform better in freezing temperatures (0 to -10° C happen at a regular basis during the winter nights in the study area). Used batteries were collected by a local school and recycled within an educational collection program.

### 3.4. Cataloguing camera trap images

For the Eurasian Lynx, Thüler, (2002) described five distinct categories of coat patterns (large spots, small spots, clear rosettes, small spots and rudimentary rosettes, no spots), which we also used as a basis in this study. Individuals with large spots, which are the easiest to distinguish, made up for 37.5 % of the adult lynx captured on camera traps and the low representation of lynx with no distinctive coat pattern (only 2 individuals), makes the population in our study area suitable for capture-recapture surveys via camera trapping (Figure 2a, b).



Figure 2a. Different coat patterns used for lynx identification in a study area in Southern Carpathians, Romania.

To identify individual lynx from images, we scanned at least two to three different areas on the body for the occurrence of distinct spots or rosettes. Lynx were compared for the category of their coat pattern, the size and shape of spots or rosettes and their relative position to each other. As fur markings differ between the two sides, full identification of an individual requires good photographs from both flanks, which we were aiming for by setting two opposite cameras at each trap station. Each newly identified lynx is included into the catalogue and receives an identification number consisting of a letter and a 3-digit sequential number, where letter B stands for identified on both sides, while letters L and R indicate that only the left or the right side of the individual are known. In the process of a monitoring session, each lynx photograph was compared to already existing reference images in the catalogue, starting with individuals from the same region and, if no matches are found, gradually enlarging the search. If due to bad image quality (e.g. blurred and under- or overexposed pictures or only small parts of the lynx were captured) the individual could not be surely identified, we entered the image into the camera trap database as unknown lynx.

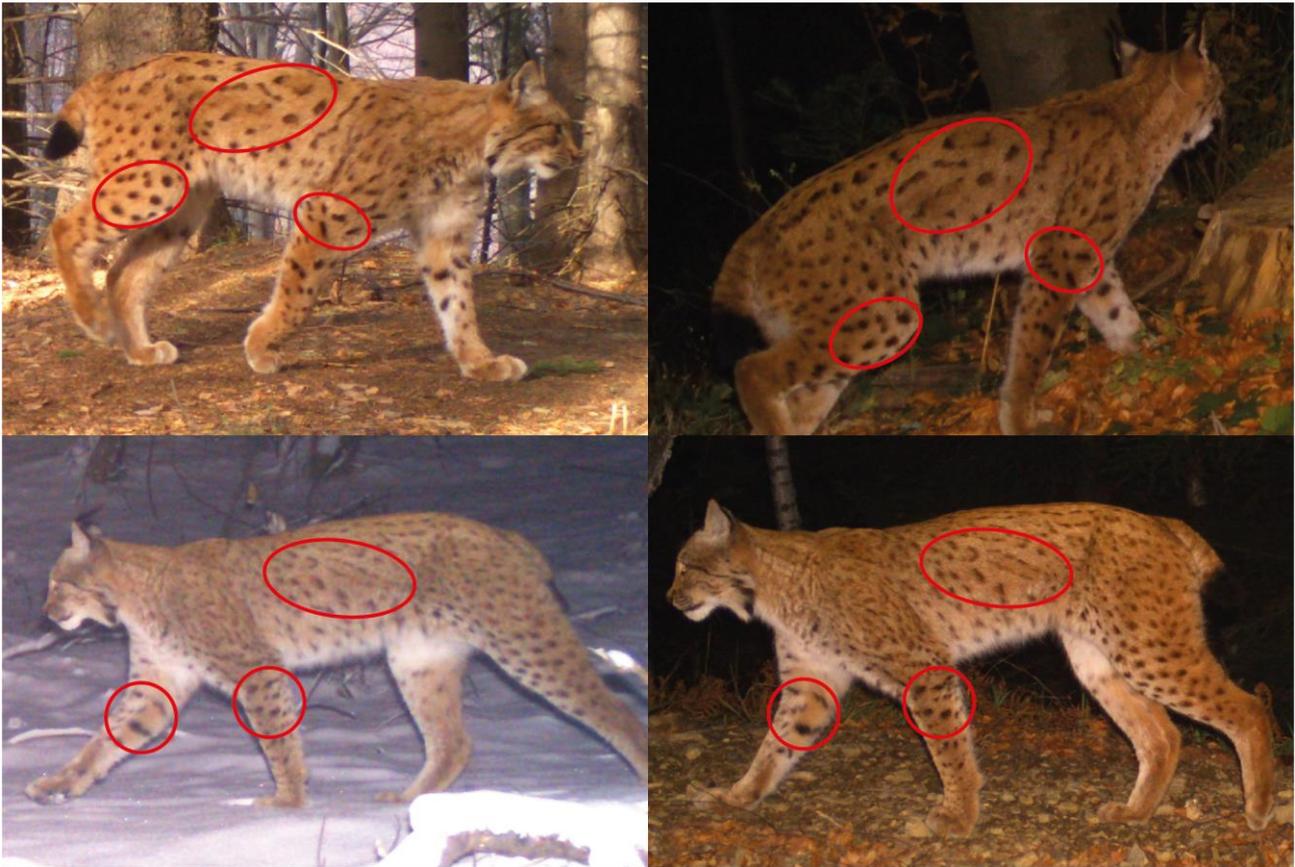


Figure 2b. Distinct spots and rosettes and their relative position to each other used for identification of individual lynx from images (up female B015, down male B010).

Sexing of lynx from lateral images is limited to captures of females with their offspring. Occasionally, however, we obtained photographs of the dorsal area which allowed the identification of males if the scrotum was seen, or of females if the genital area was clearly visible.

Based on their age and status, lynx can be categorised into adult/resident lynx, subadults (yearlings), and juveniles (lynx in their first year) that still follow their mother. Since independent dispersers cannot be distinguished from adult resident lynx based on photographs (Zimmermann et al., 2013), the following abundance and density estimates refer to all independent lynx. Also, following Zimmermann et al. (2013), family groups (females with juveniles) were registered as an encounter of the respective female, whenever any one of the family group members was detected. This is reasonable, since due to the time needed for the white flash to recharge (which can take up to 1 minute under certain conditions), not all the members of a family group might get detected when passing a camera trap. The minimum age of some lynx could be determined based on images from previous years.

Depending on the travel speed and the set-up, an animal can trigger the cameras at a trap station several times in a row. We therefore merged all lynx images resulting from a subsequent period of 5 minutes into one encounter event.

## 3.5. Modelling population estimates

### 3.5.1. Encounter histories and input files

To fit SECR models, we built the encounter data that tracks at which traps and at what time each individual encounter occurred. As a trap night we defined the 24h period between noon of the previous day and noon of the day with the given date. To ensure independent encounters and avoid zero-inflated datasets, we pooled time of capture to sample occasions of 5 days each, a common approach for elusive species with low detection probabilities (Rovero & Zimmermann, 2016). The encounter data includes all identified, sexed and unsexed, independent individuals. We removed juveniles still dependent on their mothers (i.e., from family groups) from further analysis but referred them as a new encounter of their specific identified female (as described in Chapter 3.4). All predictions are thus independent of animal's sex. For obtaining better estimates, we removed encounter histories from 7 biased traps for the winter session and 15 for the autumn – early winter, out of the initial array of 59 and 76 respectively (see Figure 1). Compared to the rest, the biased traps had lower detection rates although the species was captured but the quality of the pictures did not allow identification. The removed traps were all clustered in the southern part of the study area, in a game management unit where we had limited time to understand lynx movement routes and obviously had difficulties in finding the right spots for camera installation within the predefined trap array. The final population estimates still cover this region due to the large effective sampling area we predicted based on the remaining traps (see next chapters).

We also built the trap deployment data that summarizes the spatial information of the traps (coordinates), along with the effort for each trap, expressed in days per 5-days sample occasion (Royle et al. 2014). As lynx is a strictly protected species we can only share the encounter and trap deployment data upon request and for scientific, management and educational purposes only. Next chapters and Appendix 1 present in detail our modelling decisions from detection models to predicting density surface. We analysed the data in R using functions available in package 'secr', a tool developed for fitting SECR models (Efford, 2020). All GIS processing of spatial data happened in QGIS (QGIS Development Team, 2018).

### 3.5.2. Modelling detection

We first initialised  $\sigma$  model parameter using a root pooled spatial variance function (Calhoun & Casby, 1958; Slade & Swihart, 1983). According to Efford (2020) this function is a measure of the 2-D dispersion of the locations where individuals were detected, pooled over individuals. Second, we calculated the effective sampling area by varying a buffer width around our traps, a buffer which is a function of the  $\sigma$  parameter, in the sense of Borchers & Efford (2008). We tested different buffer widths from 3 to  $6 \times \sigma$  but selected  $5 \times \sigma$  as a final buffer as the probability of capturing a lynx from outside of this distance converged to zero during both monitoring sessions (Appendix 1) (see also Rovero & Zimmermann, 2016). Thus, this distance threshold removed our concerns about edge effect in density estimates and was used to calculate the effective sampling area. Third, we explored three different detection functions, with detection probability declining with distance away from our traps according to a half-normal, exponential, or hazard-rate curve (Efford, 2019). Half normal curve converged first for our datasets, thus we used this function for further model fitting (Appendix 1). Finally, we used 4 automatic predictors to model detection parameters and a null model, then compared model performance according to AIC. The four predictors we used were  $b$  - permanent global learned response,  $bk$  - permanent trap-specific learned response,  $t$  - time factor (one level for each occasion), and  $T$  - time trend (Efford, 2019). We showed that  $bk$  parameter worked best for both monitoring sessions. This model imply that an individual become trap-happy or trap-shy in relation to a particular trap. The learned response is positive (e.g.,  $g0.bk = 1.99 \pm 0.27$  SE for the winter session; Appendix 1), suggesting the animals became trap-happy after camera installation. This modelling behaviour is in consistency with the modelling case study of Rovero & Zimmermann, (2016) on the same species and in similar habitat conditions. However, studies that evaluated

detection probability of fishers (*Pekania pennanti*) by comparing camera trapping and telemetry data in the same population, concluded that camera trap data do reflect space use in a manner consistent with telemetry (Popescu, de Valpine & Sweitzer, 2014; Stewart *et al.*, 2018). Such findings may suggest that large carnivores like lynx didn't become trap-happy after camera installation, but that this is rather an artefact of the lynx using the same movement paths and become more and more detectable at the same traps. We used argument `model = g0 ~ bk` hereafter to fit SECR models.

### 3.5.3. Habitat – non habitat mask

The rationale behind using a habitat – non habitat mask to fit our SECR models is to remove cells with mostly urban landscape, especially in proximity of Zărnești and Câmpulung cities, industrial clusters with up to 30 000 inhabitants. We built the mask as a rectangle area around our traps that includes at least the  $5 \times \sigma$  buffer we discussed above. Because the mask regions are at least as large as the region of integration used to fit the models (i.e., area extent around our cameras), we can be sure all detected animals have activity centres within the mask (Efford, 2020).

The mask comprised of  $1 \times 1$  km resolution cells in which we calculated proportion cover of different land uses classes from Corine Land Cover 2018 dataset (European Environment Agency, 2020). We excluded as non-habitat the cells with proportion cover of human dominated landscape higher than 0.7 and lower than 0.1 forest habitat. Everything else was considered potential habitat, the thresholds being arbitrarily selected. Specifically, we defined the human dominated landscape by merging urban classes, extensively cultivated plots and artificial reservoirs, identified by Corine Land Cover level 3 classes 111, 112, 121, 124, 131, 132, 133, 141, 142, 211, 212, 221, 222, 231, 242, 511, and 512. Forest habitat included classes 311, 312 and 313 (see the full description [here](#)). The argument behind the 0.7-0.1 rule stays in both, direct observations and literature. We observed lynx occurring in narrow forest patches in traditional agricultural landscape if well connected with the extensive forest habitats nearby. Moreover, Filla *et al.*, (2017) describes the daily and seasonal variation in lynx habitat selection in a similar environment, their findings supporting our arbitrary decisions about what to consider habitat and what not. The masks used to estimate population parameters are plotted in Figure 3 for both winter and autumn-early winter monitoring sessions.

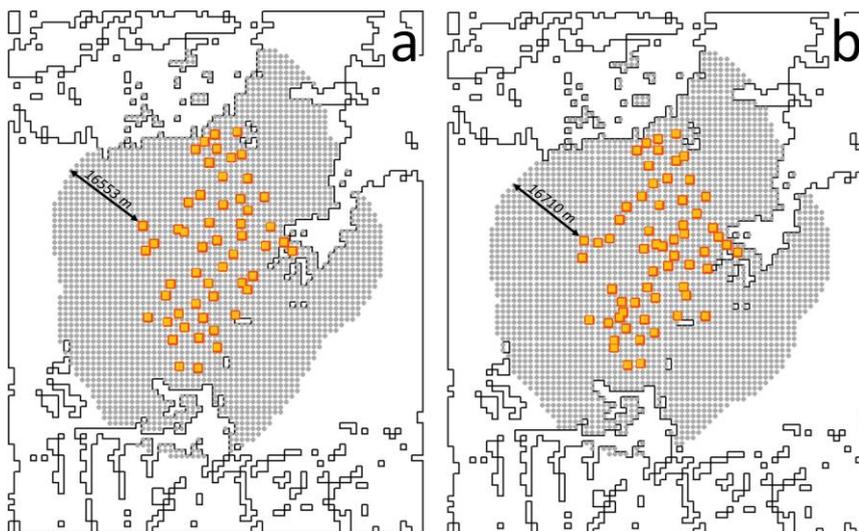


Figure 3. Habitat - non habitat masks used for SECR modelling and the effective sampling area for the winter (panel a) and for the autumn-early winter session (panel b). The excluded areas in white represent the background considered as non-habitat for the species (with  $> 0.7$  proportion cover of human-dominated landscape and  $< 0.1$  cover of forest habitats). Within the remaining space, we calculated the effective sampling area (grey dots array) using the  $5 \times \sigma$  buffer around our traps, separately for each session, and at a spacing of 1 km (Zimmermann *et al.*, 2013). We plotted the traps used to fit SECR models as filled squares in orange shades.

### 3.5.4. Modelling population size and density

Our next modelling step was to adjust for varying effort over the sample occasions as each trap ability to function depended on events such as high snow cover, battery drainage, etc. (Appendix 1). At this step we made combinations of observation models, by including or excluding habitat mask and the adjusted effort. In this modelling exercise, SECR fits observation models on lynx capture histories from incomplete spatial detections, the observation model being a distance-dependent detection function (Borchers & Efford, 2008; Efford, 2019). We evaluated model performance according to AIC and decided the observation model to interpret for the final population size ( $N$ ) and density ( $D$ ) estimates (Appendix 1).

From the model combinations we tested, the observation model accounting for variable effort over occasions and within habitat – non habitat mask stands out with highest model fit (Table 1). The model `fit.usage.mask` has the lowest AIC during the autumn-early winter session and second lowest (but comparable with the first `fit.usage.nomask`) during the winter session (Table 1). By comparing the estimates, but also by considering the importance of using a habitat – non habitat mask in our case study, we interpret `fit.usage.mask` as final population estimates for both monitoring sessions.

*Table 1. Model likelihood for different combinations of spatial detection modes: `fit.usage.mask` – the model accounting for variable effort over occasions and within habitat – non habitat mask; `fit.usage.nomask` – the model accounting for variable effort over occasions but without habitat – non habitat mask; and `fit.nousage.mask` – the model has no adjustment for effort but computes parameters within the mask.*

Monitoring Session	Model	logLik	AIC	AICc	dAICc	AICcwt
Winter	<code>fit.usage.mask</code>	-433.75	875.50	878.00	0.26	0.45
	<code>fit.usage.nomask</code>	-433.62	875.24	877.74	0.00	0.52
	<code>fit.nousage.mask</code>	-437.05	882.10	884.60	6.85	0.01
Autumn-early winter	<code>fit.usage.mask</code>	-547.97	1103.94	1106.16	0.00	0.76
	<code>fit.usage.nomask</code>	-549.16	1106.32	1108.54	2.38	0.23
	<code>fit.nousage.mask</code>	-556.47	1120.96	1123.18	17.01	0.00

### 3.5.5. Density maps

Our last modelling step was to predict density variation in space ( $D$  surface) by fitting SECR state models against environmental predictors. In this modelling exercise, SECR fits state models with a spatial Poisson process for animal activity centres (Borchers & Efford, 2008; Efford, 2019). The expected value of the process, measured as activity centres per  $1 \times 1$  km cell, is varying over space (Efford, 2019; López-Bao et al., 2018).

We used spatial predictors describing topography and habitat classes, measured as dominant category or average proportion cover per  $1 \times 1$  km cell within our habitat – non habitat mask. The predictors we used were: [Alt] – mean altitude per cell in meters, [Slo] – mean slope measured as degrees; [TRI] – a terrain roughness index averaged for nine neighbouring cells, [OpenHab] - average percent cover of open habitat, [TraditionalAgriculture] – average percent cover of traditionally managed landscape at the interface between compact forest and villages at the lower altitudes, [CLC\_311], [CLC\_312], and [CLC\_313] – average percent cover of deciduous, coniferous, and mixed forest respectively as extracted from Corine Land Cover 2018 dataset. We defined [Forest] by merging [CLC\_311], [CLC\_312], and [CLC\_313] percent cover. We defined [OpenHab] by merging Corine Land Cover classes, 321, 322, 332, 333, 411 and 512, classes that include habitats like the alpine grasslands, barerocks, etc. We defined [TraditionalAgriculture] by merging Corine Land

Cover classes 111, 112, 121, 142, 211, 222, 231, 242, and 243, including landscape features like crops, large stands of orchards, pastures around villages and other landscape features principally occupied by agriculture but with significant areas of natural vegetation. We defined [Reclass] as the dominant habitat category per cell with 5 classes: deciduous, coniferous, mixed forest, and open habitat and the traditional agricultural landscape as defined above. We defined [PublicRoad] as average density of paved and unpaved roads opened to public. All predictors were scaled prior to fitting the state models, and correlation between variable was tested. We never included [Alt], [Slo], and [TRI] in the same model as they were highly correlated.

Research on radio or GPS tracked lynx found that this species is not necessarily selecting any type or age class of forest when hunting, but prefers habitats with high complexity and low visibility (Podgórski *et al.*, 2008). During the summer, lynx was found close to small forest glades that provided food resources for its main prey – roe deer (Podgórski *et al.*, 2008). Filla *et al.*, (2017), with a larger sample size, found that lynx selected open habitat at night and forest with dense understorey vegetation probably correlated with the high abundance of its prey. Rugged terrain was also preferred during the day while altitude varied over seasons, with lower altitudes preferable during the winter (Filla *et al.*, 2017).

## Chapter 4

# Results



Photo © Călin Șerban

### 4.1. Camera functionality

The number of effective trap nights (24h periods in which at least one camera trap at a site was functioning) increased gradually from 94.7% during the pilot session to 95.5% during the winter session, and to 97.3% during the autumn-early winter session (average across trap stations). Failures were mainly due to temporary malfunctions of cameras (e.g., snow covering the lenses or sensors, drained batteries) or cameras being stolen (5% of traps per session).

Table 2. Information on effective trap nights, success rates of trap stations and image events for lynx for each monitoring session. \*Images and encounters refer to the entire time span the trap stations were active, including the installation and dismantling periods (approximately two weeks before and after the period used for statistical evaluation).

Monitoring session	Period	Days	Success rate of trap stations*	Effective trap nights	Lynx images*	Lynx encounters*	Identification rate
Pilot session	24.02-04.04.2018	40	41.7%	1,818 (94.7%)	123	43	39.5%
Winter session	17.12.2018-31.03.2019	105	71.2%	5,916 (95.5%)	474	148	81.1%
Autumn-early winter session	09.10.2019-16.01.2020	100	64.5%	7,216 (97.3%)	385	153	85.0%

Not all the trap stations detected lynx and the success rate differed significantly between sessions, increasing from 41.7% during the pilot session, to 71.2% during the winter session, but dropping to 64.5% during the autumn-early winter (Table 2). Throughout the three sessions we obtained a total of 982 lynx images that could be merged into 344 (43, 148, and 153) lynx encounter events, as multiple images per encounter occurred (e.g. 83 images taken during a mating event right at a trap station). About two thirds of the capture events were registered during night-time (59.9% in winter and 63.4% during the autumn-early winter session).

A total of 16 species co-occurred with lynx at the camera traps, including bear, wolf, fox, ungulates, mustelids and other small mammals. Fox, red deer and badger were the most observed during the pilot session (n = 313, 122 and 112 encounters respectively). Fox, wild boar, and red deer were the most observed during the winter session (n = 760, 362 and 339 encounters respectively). Red deer, wild boar and bears were the most observed during the autumn-early winter session (n = 569, 464 and 446 encounters respectively).

## 4.2. Image analysis and capture history

Images obtained during the pilot session had a low identification rate with only 39.5% of the detection events delivering images good enough to be attributed to an individual. Such we identified a total of 11 individual lynxes plus an additional family group with one juvenile. Except for two lynx (R002, R004), all other individuals were captured again during one or both of the following monitoring sessions. Recaptures were unsurprisingly low, with only two individuals being recaptured twice after their first identification.



Figure 4. Female B011 with three juveniles captured during the autumn-early winter session.

The identification rate of image events increased significantly during the winter session to an acceptable 81.1% and revealed a minimum count of 23 independent lynx (Table 3). Amongst them 13 could be identified as males and 5 as females. Only three of the identified females were seen with 1 or 2 juveniles, an additional family group with two juveniles did not match any of the other females, but image quality did not allow clear attribution. Such, the minimum number of juveniles detected during this session accumulated to six (Table 3). Based on the coat pattern and the general area, animals listed as L030 and R031 are most probably one and the same individual, but none of the trap stations that captured them delivered images from both sides and were removed from further statistical modelling. The average recapture rate per lynx was 4.04 during the winter monitoring session, with a maximum of 16 recaptures for the territorial male B010 and 12 recaptures for male B017. A total of 13 out of the identified lynx triggered at least three different encounter events (see Figure 5).

During the autumn-early winter session, the identification rate increased once more to 85%, while the minimum count remained at 23 independent individuals, despite the fact that the detection rate of the trap stations in the south-eastern part of the study area was extremely low (20% compared to 73.8% for the rest of the stations). All of the 9 males and 6 females, as well as 4 unsexed individuals detected during this session were already known from the previous winter monitoring or the pilot study (B008), assuming that their status is territorial (see Figure 6). Four of the 23 independent lynx, all of them unsexed, were identified for the first time during this session. In addition, an unidentified family group with two juveniles was captured in the very western part of the study area, increasing the number of offspring in 2019 to at least nine (Figure 4). Six animals, 4 males and 2 unsexed, from the winter session did not appear anymore in autumn-early winter, which might be explained by a higher mobility, especially of males, during the mating season in late winter. The average

recapture rate for this 100-day period increased to 4.54 with a maximum of 14 captures for male B010 and 13 captures for female B011 (Figure 5).

Table 3. Data of all independent lynx identified during one or more of the subsequent monitoring sessions, with date of first captures and category of coat pattern. Animals identified from both sides received a B-identification number, while letters R and L indicate that only the left or the right side of the individual are known. Sex identification was not always possible. M – Male, F – Female, U – Unknown. S1, 2 and 3 refer to the pilot session, winter session and the autumn-early winter session respectively.

Lynx ID	Sex	Coat pattern	First capture	S1	S2	S3	Comments
B001	M	Large spots	2018-02-26	y	y	y	
R002	M	Small spots/rudim. rosettes	2018-02-26	y	n	n	
B003	M	Large spots	2018-02-20	y	y	y	
R004	U	Small spots	2018-03-11	y	n	n	
B005	M	Small spots/rudim. rosettes	2018-01-12	y	y	y	
B006	M	Clear rosettes	2018-02-28	y	y	y	
B007	M	Small spots	2018-02-26	y	y	n	
B008	F	Large spots	2018-03-22	y	n	y	with 2 offspring in S3
B009	M	Small spots/rudim. rosettes	2018-03-26	y	y	y	
B010	M	Small spots/rudim. rosettes	2018-03-03	y	y	y	
B011	F	Large spots	2018-03-02	y	y	y	with 2 offspring in S2, 3 in S3
B014	M	Small spots/rudim. rosettes	2018-12-12	n	y	n	
B015	F	Large spots	2018-10-15	n	y	y	with 2 offspring in S3
B016	M	Small spots	2019-03-07	n	y	y	
B017	M	Small spots/rudim. rosettes	2019-02-14	n	y	n	
B019	U	Large spots	2019-03-10	n	y	y	
B021	U	Clear rosettes	2019-01-02	n	y	y	
B024	F	Large spots	2019-01-09	n	y	y	female with 1 offspring in S2
B025	U	Large spots	2019-01-09	n	y	y	offspring of B24 (S2)
B026	F	Without spots	2019-01-02	n	y	y	
B027	M	Large spots	2019-02-06	n	y	y	
B028	F	Small spots/rudim. rosettes	2019-01-25	n	y	y	with 1 offspring in S2
L030	U	Clear rosettes	2019-03-02	n	y	n	probably same as R31
R031	U	Clear rosettes	2019-03-02	n	y	n	probably same as L30
B032	M	Large spots	2018-12-25	n	y	y	
B033	U	Small spots/rudim. rosettes	2019-01-28	n	y	y	
B035	M	Without spots	2019-01-15	n	y	n	
R036	U	Large spots	2019-03-08	n	y	n	
B040	U	Small spots/rudim. rosettes	2019-11-20	n	n	y	

<b>L041</b>	U	Clear rosettes	2019-12-01	n	n	y	
<b>R042</b>	U	Large spots	2019-12-03	n	n	y	
<b>B043</b>	U	Clear rosettes	2020-01-04	n	n	y	
<b>UI</b>	F		2018-12-30		y		with 2 offspring in S2
<b>UI</b>	F		2020-09-10			y	with 2 offspring in S3

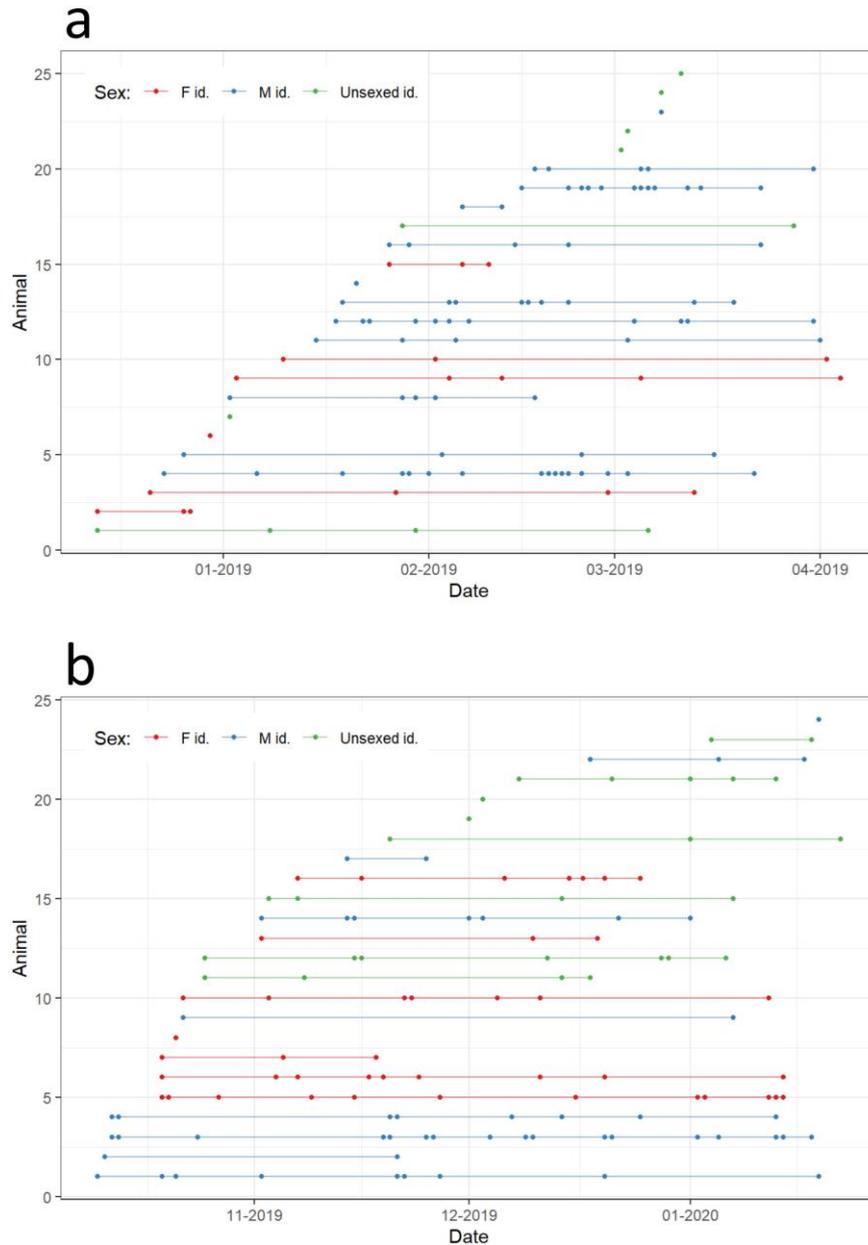


Figure 5. Mark recapture saturation graph with recaptures of each identified individual used for spatial explicit capture recapture modelling of population parameters for the winter (panel a) and for the autumn-early winter session respectively (panel b). F and M id. stands for identified female and male individuals while unsexed id. stands for identified individuals that we were not able to determine their sex. The R function behind this figure with generosity of Tomaž Skrbinšek.

We captured a high proportion of unsexed yet identified individuals in both session (Table 3, Figure 5), therefore decided to estimate population size and density independent from sex.

The spatial patterns of lynx recaptures at camera stations were complex. Some individuals had only scattered recaptures on the map, others appear to have stable home ranges (e.g. B009, B010, B011), and others shifted home ranges between sessions (e.g. B006, Figure 6 and 7). Males had a higher number of recaptures per trap than females during the winter (average pooled across identified males =  $1.8 \pm 1.1$  and females =  $1.3$  recaptures  $\pm 0.5$  SD). During the autumn-early winter session, however, males and females had an equal number of recaptures per trap ( $1.54$  recaptures  $\pm 0.5$  and  $0.4$  SD respectively). We found no significant difference in the trap revealed movement between males and females during the winter session ( $U = 358.0$ ,  $p$ -value =  $0.204$ ), but males showed significantly higher trap-revealed movement than females during the autumn-early winter session ( $U = 552.5$ ,  $p$ -value =  $0.021$ ) (Appendix 2). Consecutive recaptures of the same individual occurred at camera trap stations that were in a Euclidean distance of up to  $21.8$  km from each other during the winter, and  $15.0$  km during the autumn early winter (Appendix 2). The number of trap stations at which individuals were captured ranged between 1 and 8, no significant differences were found between sexes ( $U = 25.5$ ,  $p$ -Value =  $0.674$  in winter and  $U = 19.0$ ,  $p$ -value =  $0.379$  in autumn-early winter).

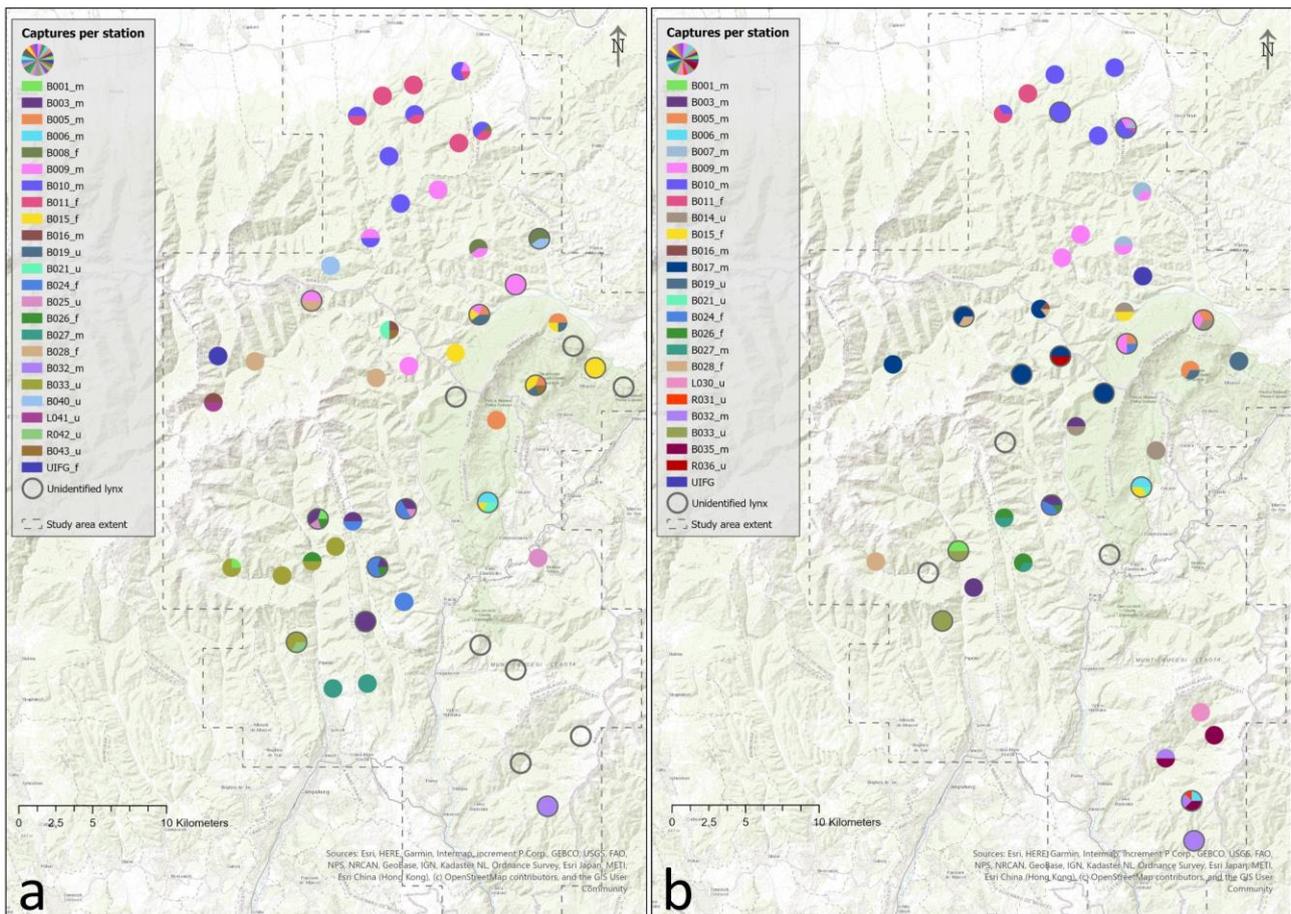


Figure 6. Maps showing how each trap station was frequented by the identified lynx in the winter (panel a) and in the autumn-early winter session respectively (panel b).

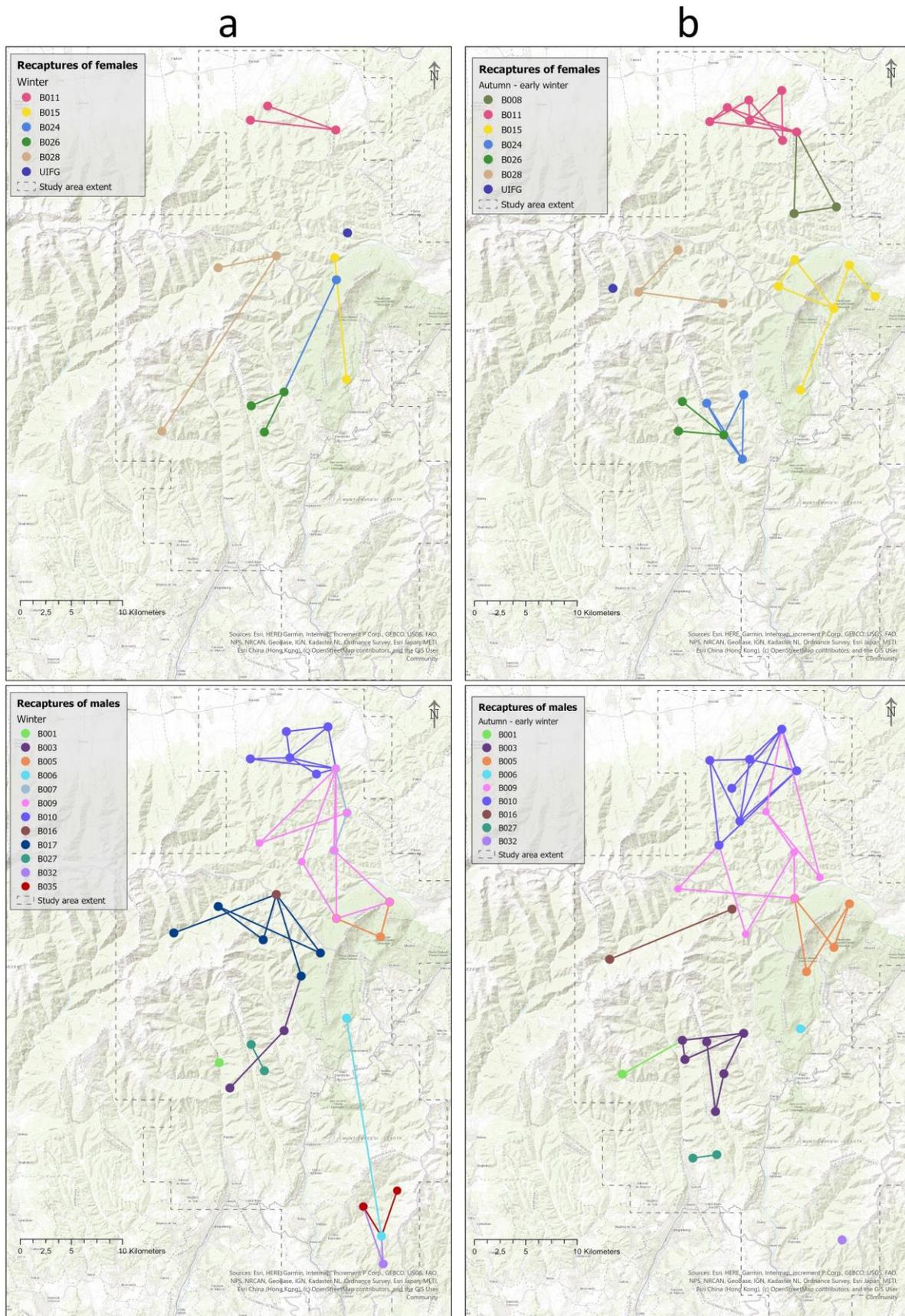


Figure 7. Examples of capture history of identified female and male lynx in Southern Carpathians, Romania. Panel (a) represents the winter session and panel (b) represents the autumn-early winter session. Recaptures of the same identified individual maintain line colour between the two monitoring sessions.

## 4.3. Population parameters estimates

### 4.3.1. Regional population size

Regional population size of adult, independent lynx individuals, is shown in Table 4. The adults' estimates are similar between the two sessions, slightly higher for autumn-early winter, with slightly better predictive power ( $se = 8.84$  for the winter and  $8.11$  for the autumn-early winter monitoring session when estimation method is 'realised  $N$ '; Table 4). Keep in mind that values from Table 4 are not calculated for the area extent of our traps but for the much larger effective sampling areas (see Figure 3 and next chapter). One might want to add the observed number of juveniles in family groups to the adults' estimates for obtaining a final picture of the regional population size.

Table 4. Regional population size for Eurasian lynx in a study area in the Southern Carpathians, Romania.  $N$  independent is the observed number independent lynx of whose capture history we used to fit SECR models;  $N$  juveniles is the observed number of juveniles in family groups, thus still dependent on females, and removed from further statistical modelling; Realised  $N$  is the number of independent individuals within the region for the current realisation of the process, while 'expected  $N$ ' is defined as the number of animals with activity centres fitted under a density surface (see M. G. Efford & Fewster, 2013 and Johnson, Laake, & Ver Hoef, 2010 for further details).  $lcl$  and  $ucl$  stands for lower and upper 95% confidence limit of  $N$ .

Monitoring Session	$N$ independent lynx	$N$ juveniles *in family groups	Method	Adults' Estimates	se	lcl	ucl
Winter	21	6	Expected $N$	44.12	10.77	27.52	70.72
			Realised $N$	44.12	8.48	32.51	67.41
Autumn-early winter	23	9	Expected $N$	48.06	10.67	31.26	73.87
			Realised $N$	48.06	8.11	36.50	69.52

### 4.3.2. Population density

The root pooled spatial variance function estimated  $\sigma$  at 3310 m for the winter session and 3343 m for the autumn-early winter session. After clipping the  $5 \times \sigma$  buffer regions by the habitat mask (see chapter 3.5.4 and Figure 3), we obtained an effective sampling area of 2729.5 km<sup>2</sup> for the winter session and 2767.4 km<sup>2</sup> for the autumn-early winter. Within these areas, we estimated the average adult population density at 1.60 and 1.73 lynx / 100 km<sup>2</sup>, with similar robustness of the estimates between monitoring sessions. See Table 5 for complete statistics around these estimates.

Table 5. Average adult population density for Eurasian lynx in a study area in Southern Carpathians, Romania.  $lcl$  and  $ucl$  stands for the lower and upper 95% confidence limit of  $N$ .

Monitoring Session	$D$ estimates (lynx / 100 km <sup>2</sup> )	se	lcl	ucl
Winter	1.60	0.39	1.00	2.57
Autumn-early winter	1.73	0.38	1.12	2.66

## 4.4. Predictors for density maps

For the winter session, with one exception, none of the variables stands out as really important for  $D$ . All models have similar AIC, with a difference lower than 1 unit from a model to another and close to the null

model (Table 6). The exception is [PublicRoad], but not necessarily suggesting this predictor explains  $D$ , being rather a bias we induced by placing the cameras close to unpaved forest roads (i.e., accessible areas in winter conditions, even though we respected the sampling design with a predefined trap array). Thus, we decided to leave out this predictor from further modelling.

Table 6. Model likelihood for different predictors we used to fit density variation in space ( $D$  surface). Predictors marked with an '\*' were selected to build the final models for fitting  $D$  surface in both monitoring sessions. We tried to select the predictors based on their predictive power, but, for the winter session this was not possible. In this case we selected predictors based on comparability with the autumn-early winter session. To quantify the predictive power of each predictor, we used AICwt (Akaike weights) that sum to 1 for the entire set of models, and can be interpreted as the weight of evidence in favour of a given predictor. [Alt] – mean altitude per cell in meters, [Slo] – mean slope measured as degrees; [TRI9] – a terrain roughness index, [OpenHab] - average percent cover of open habitats that include habitats like pasture, grasslands, [TraditionalAgriculture] – average percent cover of traditionally managed landscape at the interface between compact forest and villages at the lower altitudes including landscape features like crops and large stands of hayfields and orchards but with significant areas of natural vegetation, [CLC\_311], [CLC\_312], and [CLC\_313] – average percent cover of deciduous, coniferous, and mixed forest respectively. We defined [Forest] by merging [CLC\_311], [CLC\_312], and [CLC\_313] percent cover. We defined [Reclass] as the dominant habitat category per cell with 5 classes: deciduous, coniferous, mixed forest, and open habitat and human dominated landscape as defined above. We defined [PublicRoad] as average density of paved and unpaved roads opened to public.

Monitoring session	Model	logLik	AIC	AICc	dAIC	AICwt
Winter	$D \sim$ PublicRoad	-234.6	477.3	479.8	0.00	0.35
	$D \sim$ CLC_313	-235.7	479.4	481.9	2.16	0.11
	$D \sim$ 1	-237.0	480.0	481.4	2.73	0.08
	$D \sim$ <b>TraditionalAgriculture*</b>	-236.1	480.3	482.8	3.04	0.07
	$D \sim$ <b>Forest*</b>	-236.3	480.6	483.1	3.34	0.06
	$D \sim$ CLC_312	-236.5	481.0	483.5	3.72	0.05
	$D \sim$ OpenHab	-236.5	481.0	483.5	3.76	0.05
	$D \sim$ TRI	-236.6	481.3	483.8	4.05	0.04
	$D \sim$ <b>Slo*</b>	-236.7	481.4	483.9	4.10	0.04
	$D \sim$ CLC_311	-236.8	481.7	484.2	4.44	0.03
	$D \sim$ Alt	-236.9	481.9	484.4	4.67	0.03
	$D \sim$ Reclass	-234.2	482.4	491.0	5.12	0.02
Autumn-early winter	$D \sim$ <b>Slo*</b>	-263.3	534.7	536.9	0.00	0.15
	$D \sim$ TRI	-263.4	534.8	537.0	0.12	0.14
	$D \sim$ 1	-264.4	534.9	536.2	0.27	0.13
	$D \sim$ <b>TraditionalAgriculture*</b>	-263.7	535.4	537.6	0.74	0.10
	$D \sim$ Alt	-263.8	535.6	537.8	0.89	0.09
	$D \sim$ CLC_312	-264.0	536.0	538.2	1.34	0.07
	$D \sim$ PublicRoad	-264.1	536.3	538.6	1.66	0.06
	$D \sim$ OpenHab	-264.2	536.5	538.7	1.82	0.06
	$D \sim$ <b>Forest*</b>	-264.3	536.6	538.9	1.97	0.05
	$D \sim$ CLC_311	-264.4	536.9	539.1	2.20	0.05
	$D \sim$ CLC_313	-264.4	536.9	539.1	2.24	0.04
	$D \sim$ Reclass	-263.0	540.0	547.5	5.35	0.01

For the autumn-early winter session, it is shown that some of the predictors explained  $D$  surface better, i.e., [TraditionalAgriculture], [Slo]. In the autumn, the [PublicRoad] no longer induced the bias it induced during the winter, suggesting lynx are now detected at traps further from roads too. Thus, the predictors we used for

modelling  $D$  surface were  $\text{model} = D \sim \text{Forest} + \text{TraditionalAgriculture} + \text{Slo} + \text{I}(\text{Slo}^2)$  for both monitoring sessions, and which account for an AICwt of 0.17 for the winter and of 0.30 during the autumn-early winter. We used the syntax  $\text{I}(\text{Slo}^2)$  for accounting for a nonlinear relation with slope.

The resulting spatial models showed a shift in  $D$  from winter to autumn in the mosaic landscape region (at the interface between forest, agriculture, and rural developments) (Figure 8). The models highlight large patches with predicted  $D$  of 1.5 – 2 lynx /100 km<sup>2</sup> and above, in the lower mountains for the winter session.  $D$  was unrealistically high in the alpine area at the western limit of the study area, suggesting poor performance during both monitoring sessions there (see confidence limits in Appendix 1).

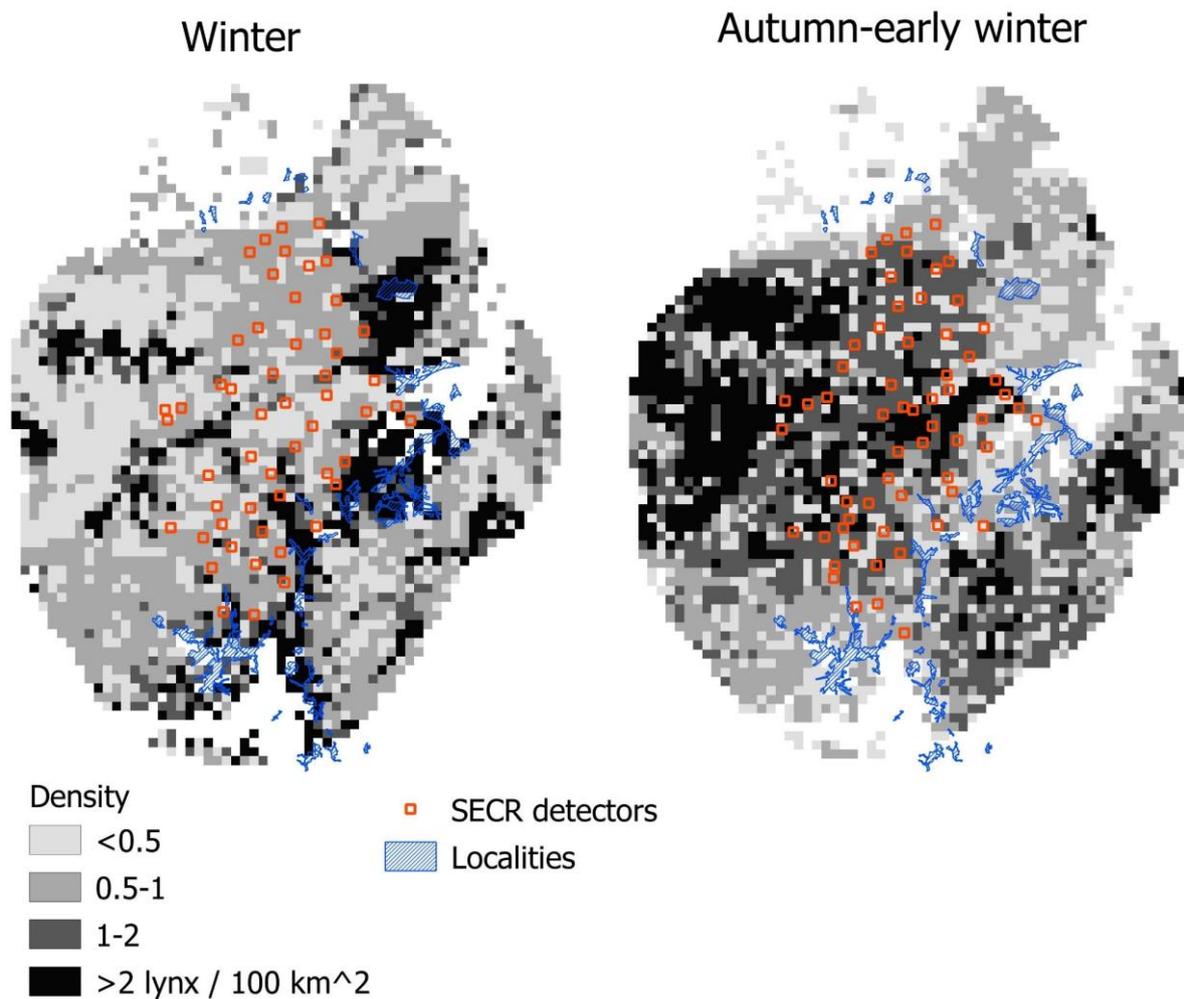


Figure 8. Density surface models derived through spatially explicit capture recapture for the Eurasian lynx in a study area in the Southern Carpathians, Romania. We modelled forest cover, traditional agricultural landscape and terrain slope against the encounter history of 21 identified lynx during the winter, respectively 23 during the autumn-early winter, within an effective sampling area of approx. 2700 km<sup>2</sup>.

## Chapter 5

# Discussion



Photo © Călin Șerban

The main purpose for our study was to obtain the first population size and density estimates of the Eurasian lynx in the Romanian Carpathians, a stronghold for this species in Europe. We conducted camera trap capture-recapture surveys across a pilot and two full monitoring sessions, used a Spatially Explicit Capture-Recapture framework, and tested the following hypotheses: i. autumn-early winter monitoring will return better population estimates when compared to the winter as the home ranges are more stable before the mating season; and ii. density shifts in space from autumn-early winter to the winter session according to topography and habitat structure (e.g., from compact forest habitat to the mosaic of habitats around localities and vice versa). Although we found the predictive power of population size  $N$  and density  $D$  models was similar between the winter and the autumn-early winter sessions, our cumulative number of detected lynx versus sampling occasions reached the asymptote faster during the autumn-early winter monitoring. This suggests that starting the monitoring earlier during the autumn and finishing it in January, prior to the mating season, is preferable for the Romanian lynx population, corroborating findings of Weingarth et al., (2015) in a Central-European population. The Eurasian lynx density ( $D$ ) in our study areas was  $1.6 \pm 0.39$  and  $1.7 \pm 0.38$  SE adult lynx / 100 km<sup>2</sup> for winter and autumn-early winter session respectively. These densities are higher than the 1.04 - 1.42 range reported

in the north-western Swiss Alps based on a similar class of spatial models (Zimmermann *et al.*, 2013), and also higher than the 0.7-0.8 resident adults / 100 km<sup>2</sup> in the Jura Mountains (Breitenmoser-Würsten *et al.*, 2007). When compared with other populations from the Slovak Carpathians, which have environmental conditions and landscape settings more similar to our study area, our density estimates were higher than in the Muránska Planina National Park (i.e., 1.4 lynx / 100 km<sup>2</sup>; Smolko *et al.*, 2018), and 2-3 times higher than in the Štiavnica Mountains and Veľká Fatra National Park (i.e., 0.5 lynx / 100 km<sup>2</sup>; Kubala *et al.*, 2019). It is likely that Eurasian lynx density in our study area is higher than in Central and Western Europe due to lower human-induced mortality rates, but also because of the relatively lower levels of human impact and human density. For example, we expect road mortality to be lower in the Romanian Carpathians due to a lower density of high-traffic roads. There are no highways in our study areas, and only one high traffic national road and four medium-traffic county roads pass through potentially suitable lynx habitat. In contrast, road mortality contributed to the decline of a reintroduced Eurasian lynx population in the Jura Mountains (Breitenmoser-Würsten *et al.*, 2008), and is acknowledged as the main cause of mortality for the Iberian lynx in the much more fragmented landscape of southern Spain (Garrote *et al.*, 2020). These findings corroborate felid ecology studies in North America, which also suggest that felids are susceptible to roadkill, strongly affecting their population dynamics (e.g., cougars, Dickson & Beier, 2002; and bobcats, Nielsen & Woolf, 2002; Bencin *et al.*, 2019). In addition to mortality from vehicle strikes, human access to remote areas via snowmobiles during winter determined high levels of poaching in Scandinavian lynx populations (Andrén *et al.*, 2006). This is not the case in the Romanian Carpathians, as most logging roads are inaccessible during the winter, fewer individuals have access to such equipment, and an effective anti-poaching control was in place throughout the entire study area.

Another possible explanation for potentially lower human-related mortalities relative to other European populations is the higher level of acceptance of lynx by the rural communities and livestock breeders in particular (Lescureux *et al.*, 2011). Unlike bears and wolves, Eurasian lynx generates almost no human-wildlife conflicts in the Romanian Carpathians, thus illegal killing is likely low and accidental, with local media reporting occasional killing of juvenile lynx by shepherd dogs. In contrast, Breitenmoser-Würsten *et al.*, (2008) estimated illegal killing reached 32% in a radio collared Eurasian lynx population in the Jura Mountains due to a much lower level of acceptance from local communities and the hunting community, which perceives lynx as a competitor for valuable game species. Iberian lynx populations were also impacted by leg-holds, box traps, snares and poisoning within non-selective predator control programs (Gil-Sánchez & McCain, 2011). Surprisingly, although the ungulate abundance is not as high as in Western Europe (Promberger-Fürpass, Predoiu & Ionescu, 2001), higher lynx densities are supported in Romania. However, data on ungulate abundance is scarce in Romania. It can be considered a certainty though that with the establishment of the Piatra Craiului National Park in 1999, chamois populations in that area increased significantly. In addition, both ungulate and predator populations benefited from the no-hunting policy of the CARPATHIA initiative, which eventually resulted in a hunting-free zone of over 80,000 ha, including an effective anti-poaching programme that kept illegal killings to a minimum.

We estimated the population size at 44 and 48 adult lynx for the winter and the autumn-early winter session respectively. The abundance is estimated at the level of an effective sampling area of ~2700 km<sup>2</sup>, which includes a buffer around the traps extent (Efford & Fewster, 2013). Research based on high resolution GPS data suggest significant differences in movement patterns and home ranges between males, non-breeding females and females with juveniles (e.g., Signer, 2017). In this respect, running separate models for each group would have returned even more robust estimates. Thus, a limitation to our density and population size predictions is posed by the difficulty in sexing all lynx camera trap encounters. With 35% unsexed individuals in our encounter history, we were unable to run separate estimates per sex. This limitation can be mitigated by increasing the long term monitoring to consecutive years that eventually allows sex identification for a higher proportion of a lynx population (Rovero & Zimmermann, 2016).

We found density to vary across space and sessions in response to topography (slope), forest cover and cover of traditional agricultural landscape. Density hotspots shifted between the traditional agricultural landscape at lower altitudes correlated with less steeper slopes during the winter session, and the compact forest stands at mid-altitudes with moderate slopes during the autumn-early winter. In general, our seasonal hotspots of densities, and their shifts, are in consistency with findings of Filla *et al.*, (2017) who investigated habitat selection based on GPS telemetry. Filla and colleagues found that Eurasian lynx significantly selected lower altitudes during the winter and preferred rugged terrain in summer and during the day, thus concluding that open habitats play an important role in lynx habitat preferences. Rozyłowicz *et al.*, (2010) too found that increased spatial heterogeneity of habitats, significantly increased the occurrence probability of a female lynx from the Eastern Carpathians, Romania. At the edge of the Romanian Carpathians, traditional farming created hayfields and orchards in a mosaic with significant areas of natural vegetation including deciduous forest fragments with dense understory vegetation and shrubs. This landscape mosaic is known to attract high densities of ungulates that form the main diet of lynx (mainly roe deer; Molinari-Jobin *et al.*, 2007; Basille *et al.*, 2009) and could be a plausible explanation for high densities estimated in traditional agricultural landscapes during the winter. Herfindal *et al.*, (2005) showed that variation in home-range size of both male and female lynx was explained by variation in roe deer density in Norway. Schmidt *et al.*, (1997) showed that male lynx home ranges are related to the distribution of females, whereas in the case of females, home ranges are determined by food-related factors. However, the question about how roe deer influences lynx density variation in space still remains to be answered until spatial data at fine resolution about roe deer abundances will be available for the Romanian Carpathians. Another important prey species for Eurasian lynx is the chamois (*Rupicapra rupicapra*) (Molinari-Jobin *et al.*, 2007). In this respect, one limitation of our study is the lack of camera trap effort at high altitudes in chamois habitat (i.e., alpine zone). Our trap array might not cover sufficiently the alpine area to document any possible lynx space use there due to chamois predation. Because density is defined as a spatial Poisson process for animal range centres in SECR models (Sun, Fuller & Andrew Royle, 2014), lacking traps in the alpine area was reflected in the high uncertainty of our spatial predictions there, compared to areas below the tree line, where traps were systematically distributed (Appendix 1). However, field observations elsewhere suggest that chamois prefer lower altitudes in late autumn and winter when foraging in mid-elevation forests (Kati *et al.*, 2020). These observations add to the usefulness of our density spatial predictions in areas below the tree line. Similar to roe deer, Romania lacks robust data on chamois abundance, thus limiting our inference on the importance of this prey item for influencing space use by lynx. Given the limitations in data availability on the prey base in our areas, we suggest that the models predicting density hotspots should be interpreted with care in terms of explaining general distribution patterns and shifts between seasons.

From the 23 individual lynx detected in the last monitoring session (autumn-early winter), 19 were recaptures from the previous session, and four, all of them unsexed, were detected for the first time during that session. Six animals, four males and two unsexed, identified in winter were not recaptured in autumn-early winter, suggesting home ranges of males increases. This is supported by telemetry data, which showed that (1) males had larger territories than females (Breitenmoser-Wursten *et al.*, 2007), and (2) there was overlap between home ranges of neighbouring males: 16.2% in Jura Mountains (Breitenmoser-Wursten *et al.*, 2007) and up to 30% in Białowieża Primeval Forest (Schmidt, Jędrzejewski & Okarma, 1997), significantly higher than the overlap between female ranges. Moreover, high number of lynx disappearing from the cameras at the last session may be explained by high turnover (high rates of emigration and immigration), which was found for lynx in the Dinaric Alps in Slovenia (Fležar *et al.*, 2019), as well as for Geoffroy's cat (*Leopardus geoffroyi*) in scrublands of central Argentina (Pereira *et al.*, 2011). Nevertheless, the differences we revealed for the trap revealed movement between males and females have to be interpreted with care, as the limited number of relocations per individual returned by camera trapping data, can't support strong inferences on movement and

spatial ecology, especially for an elusive species like lynx. We found the observed number of offspring is increasing between the two monitoring sessions, from six to nine. This is likely an outcome of increased camera trap detection success, and the fact that some kittens may have dispersed prior to the onset of the first (winter) season. However, our observations on reproductive females suggest that litter size is 2-3 kittens, which is consistent with the average litter size recorded in the Jura Mountains (Breitenmoser-Würsten *et al.*, 2008).

In comparison with other methods, camera trapping pictures led to higher recapture rates than a non-invasive DNA monitoring on the same lynx population in 2017-2018 (Skrbinšek, *et al.*, 2019). In that study we showed the challenges of finding sufficiently fresh DNA samples for this highly elusive species and in the rough terrain of the Romanian Carpathians. Although we collected 17 hair samples, their reliability was very low. Capturing non-target species from scat and urine was a problem too (i.e., fox and wildcat), eventually reducing the study performance to a genotyping success of 37.5% and detection of only 3 genotypes (Skrbinšek *et al.*, 2019). In contrast, we here identified 23 individuals through camera trapping. Anile *et al.*, (2014) too obtained better data with camera trapping than with DNA sampling on a wildcat population at Etna Volcano, Italy. They identified 14 wildcats on cameras and 10 through genetics, and had a similar low genetic sampling size of 39 samples. They showed better population estimates obtained on camera trapping data, through a similar SECR approach (Anile *et al.*, 2014). We suggest camera trapping can be used for a broader implementation in the Romanian Carpathians if based on a priori knowledge on lynx movement routes. This knowledge has to be obtained through pilot surveys targeted towards identifying suitable locations for trap installation. Knowledge from local hunters and game wardens is important here. For broad scale monitoring in the Romanian Carpathians we suggest replicating the autumn-early winter monitoring scheme we implemented here across several study sites representative for the entire Romanian Carpathians: e.g., sites representing the compact forest landscape of the Eastern Romanian Carpathians, sites covering the rugged terrain of the Southern Carpathians, and sites located at lower altitudes in the Western Romanian Carpathians. These surveys could be done at a 5 years interval, supplemented by annual reporting of track counts. Such a monitoring scheme will contribute to better national-level estimates and provide knowledge about density variation for an important source lynx population for both natural recolonization and reintroduction programmes.

## References

- Andrén, H., Linnell, J.D.C., Liberg, O., Andersen, R., Danell, A., Karlsson, J., Odden, J., Moa, P.F., Ahlqvist, P., Kvam, T., Franzén, R. & Segerström, P. (2006). Survival rates and causes of mortality in Eurasian lynx (*Lynx lynx*) in multi-use landscapes. *Biol. Conserv.* **131**, 23–32.
- Anile, S., Amico, C. & Ragni, B. (2012). Population density estimation of the European wildcat (*Felis silvestris silvestris*) in Sicily using camera trapping. *Wildl. Biol. Pract.* **8**, 1–12.
- Anile, S., Ragni, B., Randi, E., Mattucci, F. & Rovero, F. (2014). Wildcat population density on the Etna volcano, Italy: A comparison of density estimation methods. *J. Zool.* **293**, 252–261.
- Artelle, K.A., Reynolds, J.D., Paquet, P.C. & Darimont, C.T. (2014). When science-based management isn't. *Science*. **343**, 1311.
- Basille, M., Herfindal, I., Santin-Janin, H., Linnell, J.D.C., Odden, J., Andersen, R., Arild Høgda, K. & Gaillard, J.M. (2009). What shapes Eurasian lynx distribution in human dominated landscapes: Selecting prey or avoiding people? *Ecography (Cop.)*. **32**, 683–691.
- Bencin, H.L., Prange, S., Rose, C. & Popescu, V.D. (2019). Roadkill and space use data predict vehicle-strike hotspots and mortality rates in a recovering bobcat (*Lynx rufus*) population. *Sci. Rep.* **9**, 1–13.
- Di Bitetti, M.S., Paviolo, A. & De Angelo, C. (2006). Density, habitat use and activity patterns of ocelots (*Leopardus pardalis*) in the Atlantic Forest of Misiones, Argentina. *J. Zool.* **270**, 153–163.
- Borchers, D.L. & Efford, M.G. (2008). Spatially explicit maximum likelihood methods for capture–recapture studies. *Biometrics* **64**, 377–385.
- Breitenmoser-Würsten, C., Vandel, J.-M., Zimmermann, F. & Breitenmoser, U. (2008). Demography of lynx *Lynx lynx* in the Jura Mountains. *Wildlife Biol.* **13**, 381–392.
- Breitenmoser-Würsten, C., Zimmermann, F., Stahl, P., Vandel, J.-M., Molinari-Jobin, A., Molinari, P., Capt, S. & Breitenmoser, U. (2007). Spatial and social stability of a Eurasian lynx *Lynx lynx* population: an assessment of 10 years of observation in the Jura Mountains. *Wildlife Biol.* **13**, 365–380.
- Breitenmoser, U., Breitenmoser-würsten, C., Okarma, H., Kaphegyi, T., Kaphygyi, U., Müller, U.M., Bern, C. & Kaphegyi-wallmann, U. (2000). Action plan for the conservation of the Eurasian lynx in Europe. *Europe* **112**, 1–83.
- Calhoun, J.B. & Casby, J.U. (1958). *Calculation of home range and density of small mammals. Public Health Monograph*.
- Cazacu, C., Adamescu, M.C., Ionescu, O., Ionescu, G., Jurj, R., Popa, M., Cazacu, R. & Cotovelea, A. (2014). Mapping trends of large and medium size carnivores of conservation interest in Romania. *Ann. For. Res.* **57**, 97–107.
- Chapron, G., Kaczensky, P., Linnell, J.D.C., von Arx, M., Huber, D., Andrén, H., López-Bao, J.V., Adamec, M., Álvares, F., Anders, O., Balčiauskas, L., Balys, V., Bedő, P., Bego, F., Blanco, J.C., Breitenmoser, U., Brøseth, H., Bufka, L., Bunikyte, R., Ciucci, P., Dutsov, A., Engleder, T., Fuxjäger, C., Groff, C., Holmala, K., Hoxha, B., Iliopoulos, Y., Ionescu, O., Jeremić, J., Jerina, K., Kluth, G., Knauer, F., Kojola, I., Kos, I., Krofel, M., Kubala, J., Kunovac, S., Kusak, J., Kutal, M., Liberg, O., Majić, A., Männil, P., Manz, R., Marboutin, E., Marucco, F., Melovski, D., Mersini, K., Mertzanis, Y., Mysłajek, R.W., Nowak, S., Odden, J., Ozolins, J., Palomero, G., Paunović, M., Persson, J., Potočník, H., Quenette, P.-Y., Rauer, G., Reinhardt, I., Rigg, R., Ryser, A., Salvatori, V., Skrbinšek, T., Stojanov, A., Swenson, J.E., Szemethy, L., Trajçe, A., Tsingarska-Sedefcheva, E., Váňa, M., Veeroja, R., Wabakken, P., Wölfel, M., Wölfel, S., Zimmermann, F., Zlatanova, D. & Boitani, L. (2014). Recovery of large carnivores in Europe's modern human-dominated landscapes. *Science*. **346**, 1517–1519.

- Dickson, B.G. & Beier, P. (2002). Home-range and habitat selection by adult cougars in southern California. *J. Wildl. Manage.* **66**, 1235–1245.
- Efford, M. (2019). A tutorial on fitting spatially explicit capture–recapture models in “secr.” <https://www.otago.ac.nz/density/pdfs/secr-tutorial.pdf>
- Efford, M. (2020). secr: Spatially explicit capture-recapture models. R package version 4.2.0. <https://CRAN.R-project.org/package=secr>.
- Efford, M.G. & Fewster, R.M. (2013). Estimating population size by spatially explicit capture-recapture. *Oikos* **122**, 918–928.
- European Environment Agency. Corine Land Cover (CLC) 2018, Version 2020\_20u1. (2020). <https://land.copernicus.eu/pan-european/corine-land-cover/clc2018>
- Fedorca, A., Russo, I.R.M., Ionescu, O., Ionescu, G., Popa, M., Fedorca, M., Curtu, A.L., Sofletea, N., Tabor, G.M. & Bruford, M.W. (2019). Inferring fine-scale spatial structure of the brown bear (*Ursus arctos*) population in the Carpathians prior to infrastructure development. *Sci. Rep.* **9**, 1–12.
- Filla, M., Premier, J., Magg, N., Dupke, C., Khorozyan, I., Waltert, M., Bufka, L. & Heurich, M. (2017). Habitat selection by Eurasian lynx (*Lynx lynx*) is primarily driven by avoidance of human activity during day and prey availability during night. *Ecol. Evol.* **7**, 6367–6381.
- Fležar, U., Pičulin, A., Bartol, M., Černe, R., Stergar, M. & Krofel, M. (2019). *Eurasian lynx (Lynx lynx) monitoring with camera traps in Slovenia in 2018-2019*. <https://www.lifelynx.eu/wp-content/uploads/2019/09/Lynx-monitoring-SLO-2018-19.pdf>
- Garrote, G., Bueno, J.F., Ruiz, M., de Lillo, S., Martin, J.M., Moral, M. & Simón, M.A. (2020). Breaking barriers: Iberian lynx *Lynx pardinus* Temminck, 1827 (Mammalia: Carnivora: Felidae) colonizing Olive groves. *J. Threat. Taxa* **12**, 15221–15228.
- Gil-Sánchez, J.M., Barea-Azcón, J.M., Jaramillo, J., Herrera-Sánchez, F.J., Jiménez, J. & Virgós, E. (2020). Fragmentation and low density as major conservation challenges for the southernmost populations of the European wildcat. *PLoS One* **15**, 1–21.
- Gil-Sánchez, J.M. & McCain, E.B. (2011). Former range and decline of the Iberian lynx (*Lynx pardinus*) reconstructed using verified records. *J. Mammal.* **92**, 1081–1090.
- Goana, P., Ferreras, P. & Delibes, M. (1998). Dynamics and viability of a metapopulation of the endangered Iberian Lynx (*Lynx pardinus*). *Ecol. Monogr.* **68**, 349–370.
- Herfindal, I., Linnell, J.D.C., Odden, J., Nilsen, E.B. & Andersen, R. (2005). Prey density, environmental productivity and home-range size in the Eurasian lynx (*Lynx lynx*). *J. Zool.* **265**, 63–71.
- Hupman, K., Stockin, K.A., Pollock, K., Pawley, M.D.M., Dwyer, S.L., Lea, C. & Tezanos-Pinto, G. (2018). Challenges of implementing Mark-recapture studies on poorly marked gregarious delphinids. *PLoS One* **13**, 1–27.
- Iosif, R., Pop, M.I., Chiriac, S., Sandu, R.M., Berde, L., Szabó, S., Rozyłowicz, L. & Popescu, V.D. (2020). Den structure and selection of denning habitat by brown bears in the Romanian Carpathians. *Ursus* **2020**, 1.
- Jędrzejewski, W., Schmidt, K., Okarma, H. & Kowalczyk, R. (2002). Movement pattern and home range use by the Eurasian lynx in Białowieża Primeval Forest (Poland). *Ann. Zool. Fennici* **39**, 29–41.
- Jerina, K., Jonozovič, M., Krofel, M. & Skrbinšek, T. (2013). Range and local population densities of brown bear *Ursus arctos* in Slovenia. *Eur. J. Wildl. Res.* **59**, 459–467.
- Jimenez, J., Chandler, R., Tobajas, J., Descalzo, E., Mateo, R. & Ferreras, P. (2019). Generalized spatial mark–resight models with incomplete identification: An application to red fox density estimates. *Ecol. Evol.* **ece3.5077**.

- Kati, V., Kassara, C., Vassilakis, D. & Papaioannou, H. (2020). Balkan chamois (*Rupicapra rupicapra balcanica*) avoids roads, settlements, and hunting grounds: An ecological overview from Timfi Mountain, Greece. *Diversity* **12**.
- Keiter, D.A., Davis, A.J., Rhodes, O.E., Cunningham, F.L., Kilgo, J.C., Pepin, K.M. & Beasley, J.C. (2017). Effects of scale of movement, detection probability, and true population density on common methods of estimating population density. *Sci. Rep.* **7**, 1–12.
- Kojola, I., Hallikainen, V., Helle, T. & Swenson, J.E. (2018). Can only poorer European countries afford large carnivores? *PLoS One* **13**, 1–9.
- Kubala, J., Smolko, P., Zimmermann, F., Rigg, R., Tám, B., Ilko, T., Foresti, D., Breitenmoser-Würsten, C., Kropil, R. & Breitenmoser, U. (2019). Robust monitoring of the Eurasian lynx *Lynx lynx* in the Slovak Carpathians reveals lower numbers than officially reported. *Oryx* **53**, 548–556.
- Kuemmerle, T., Müller, D., Griffiths, P. & Rusu, M. (2009). Land use change in Southern Romania after the collapse of socialism. *Reg. Environ. Chang.* **9**, 1–12.
- Lescureux, N., Linnell, J.D.C., Mustafa, S., Melovski, D., Stojanov, A., Ivanov, G., Avukatov, V., Von Arx, M. & Breitenmoser, U. (2011). Fear of the unknown: Local knowledge and perceptions of the Eurasian lynx *Lynx lynx* in western Macedonia. *Oryx* **45**, 600–607.
- Linnell, J.D.C., Breitenmoser, U., Breitenmoser-Würsten, C., Odden, J. & Von Arx, M. (2009). Recovery of Eurasian Lynx in Europe: what part has reintroduction played? *Reintroduction of Top-Order Predators* 72–91.
- López-Bao, J. V., Godinho, R., Pacheco, C., Lema, F.J., García, E., Llaneza, L., Palacios, V. & Jiménez, J. (2018). Toward reliable population estimates of wolves by combining spatial capture-recapture models and non-invasive DNA monitoring. *Sci. Rep.* **8**, 1–8.
- Molinari-Jobin, A., Marboutin, E., Wölfel, S., Wölfel, M., Molinari, P., Fasel, M., Kos, I., Blažič, M., Breitenmoser, C., Fuxjäger, C., Huber, T., Koren, I. & Breitenmoser, U. (2010). Recovery of the Alpine lynx *Lynx lynx* metapopulation. *Oryx* **44**, 267–275.
- Molinari-Jobin, A., Zimmermann, F., Ryser, A., Molinari, P., Haller, H., Breitenmoser-Würsten, C., Capt, S., Eyholzer, R. & Breitenmoser, U. (2007). Variation in diet, prey selectivity and home-range size of Eurasian lynx *Lynx lynx* in Switzerland. *Wildlife Biol.* **13**, 393–405.
- Mueller, S.A., Reiners, T.E., Middelhoff, T.L., Anders, O., Kasperkiewicz, A. & Nowak, C. (2020). The rise of a large carnivore population in Central Europe: genetic evaluation of lynx reintroduction in the Harz Mountains. *Conserv. Genet.* **21**, 577–587.
- Mulej, J., Bertonec, I., Černe, R., Kavčič, I., Jerina, K., Marinko, U., Majič-Skrbinšek, A., Jelenčič, M., Potočnik, H., Vidrih, M. & Skrbinšek, T. (2013). Overall evaluation and monitoring of the project conservation achievements.
- Nielsen, C.K. & Woolf, A. (2002). Survival of unexploited Bobcats in Southern Illinois. *J. Wildl. Manag.* **66**, 833–838.
- O'Connell, A.F., Nichols, J.D. & Karanth, K.U. (2010). *Camera traps in animal ecology: methods and analyses*. Springer.
- Pereira, J.A., Di Bitetti, M.S., Fracassi, N.G., Paviolo, A., De Angelo, C.D., Di Blanco, Y.E. & Novaro, A.J. (2011). Population density of Geoffroy's cat in scrublands of central Argentina. *J. Zool.* **283**, 37–44.
- Podgórski, T., Schmidt, K., Kowalczyk, R. & Gulczyńska, A. (2008). Microhabitat selection by Eurasian lynx and its implications for species conservation. *Acta Theriol. (Warsz)*. **53**, 97–110.
- Pollock, K.H. (2000). Capture-Recapture Models. *J American Statistical Association.* **95**, 293–296.
- Pop, M.I., Iosif, R., Miu, I. V., Rozyłowicz, L. & Popescu, V.D. (2018). Combining resource selection functions

- and home-range data to identify habitat conservation priorities for brown bears. *Anim. Conserv.* **21**, 352–362.
- Popescu, V.D., Artelle, K.A., Pop, M.I., Manolache, S. & Rozyłowicz, L. (2016). Assessing biological realism of wildlife population estimates in data-poor systems. *J. Appl. Ecol.* **53**, 1248–1259.
- Popescu, V.D., Iosif, R., Pop, M.I., Chiriac, S., Bouroș, G. & Furnas, B.J. (2017). Integrating sign surveys and telemetry data for estimating brown bear (*Ursus arctos*) density in the Romanian Carpathians. *Ecol. Evol.* **7**.
- Popescu, V.D., de Valpine, P. & Sweitzer, R.A. (2014). Testing the consistency of wildlife data types before combining them: The case of camera traps and telemetry. *Ecol. Evol.* **4**, 933–943.
- Promberger-Fürpass, B., Predoiu, G. & Ionescu, O. (2001). *Density estimation of Roe deer, Red deer, and Wild boar. pp. 20-23. In: Carpathian Large Carnivore Project Annual Report 2000.*
- Promberger-Fürpass, B., Sürth, P. & Predoiu, G. (2002). *Lynx. pp. 27–29. In: Carpathian Large Carnivore Project Annual Report 2001.*
- Promberger-Fürpass, B. & Sürth, P. (2003). *Lynx. pp. 12–13. In: Carpathian Large Carnivore Project Annual Report 2002.*
- Rovero, F. & Zimmermann, F. (2016). *Camera Trapping for Wildlife Research (Data in the Wild)*. 320 pg. Pelagic Publishing.
- Royle, J.A., Chandler, R.B., Sollmann, R. & Gardner, B. (2014). *Spatial Capture-Recapture*. 612 pg. Academic Press.
- Rozyłowicz, L., Chiriac, S., Sandu, R.M. & Manolache, S. (2010). The habitat selection of a female lynx (*Lynx lynx*) in the northwestern part of the Vrancea Mountains, Romania. *North. West. J. Zool.* **6**, 122–127.
- Rozyłowicz, L., Popescu, V.D., Pătroescu, M. & Chișamera, G. (2011). The potential of large carnivores as conservation surrogates in the Romanian Carpathians. *Biodivers. Conserv.* **20**, 561–579.
- Schmidt, K., Jędrzejewski, W. & Okarma, H. (1997). Spatial organization and social relations in the Eurasian lynx population in Białowieża Primeval Forest, Poland. *Acta Theriol. (Warsz)*. **42**, 289–312.
- Signer, S. (2017). Movement patterns of Eurasian Lynx (*Lynx lynx*) in the Northwestern Swiss Alps. [https://www.kora.ch/fileadmin/file\\_sharing/5\\_Bibliothek/52\\_KORA\\_Publikationen/524\\_Thesis/Signer\\_2017\\_Movement\\_patterns\\_of\\_Eurasian\\_lynx\\_in\\_Swiss\\_Alps.pdf](https://www.kora.ch/fileadmin/file_sharing/5_Bibliothek/52_KORA_Publikationen/524_Thesis/Signer_2017_Movement_patterns_of_Eurasian_lynx_in_Swiss_Alps.pdf)
- Sin, T., Gazzola, A., Chiriac, S. & Rîșnoveanu, G. (2019). Wolf diet and prey selection in the South Eastern Carpathian Mountains, Romania. *PLoS One* **14**, 1–15.
- Skrbinšek, T., Jelenčič, M., Boljte, B., Konec, M., Erich, M., Iosif, R., Moza, I. & Promberger, B. (2019). *Report on analysis of genetic samples collected in 2017 – 2018 on brown bears (Ursus arctos), Eurasian lynx (Lynx lynx) and grey wolf (Canis lupus) in a pilot area in Southern Carpathians, Romania.*
- Skrbinšek, T., Jelenčič, M., Waits, L., Kos, I., Jerina, K. & Trontelj, P. (2012). Monitoring the effective population size of a brown bear (*Ursus arctos*) population using new single-sample approaches. *Mol. Ecol.* **21**, 862–875.
- Slade, N.A. & Swihart, R.K. (1983). Home range indices for the hispid cotton rat (*Sigmodon hispidus*) in Northeastern Kansas. *J. Mammal.* 580–590.
- Smolko, P., Kubala, J., Klinga, P., Tám, B., Il'ko, T., Tesák, J. & Guimaraes, N.F. (2018). *Lynx monitoring in the Muránska Planina NP, Slovakia and its importance for the national and European management and conservation of the species.* DIANA –Carpathian Wildlife Research, Banská Bystrica, Slovakia.
- Stewart, F.E.C., Fisher, J.T., Burton, A.C. & Volpe, J.P. (2018). Species occurrence data reflect the magnitude of animal movements better than the proximity of animal space use: *Ecosphere* **9**, e02112.

- Sun, C.C., Fuller, A.K. & Andrew Royle, J. (2014). Trap configuration and spacing influences parameter estimates in spatial capture-recapture models. *PLoS One* **10**, e0141634.
- Team, Q.D. (2018). QGIS Geographic Information System. Open Source Geospatial Foundation Project. <http://qgis.osgeo.org>.
- Thüler, K. (2002). Spatial and temporal distribution of coat patterns of Eurasian Lynx (*Lynx lynx*) in two re-introduced populations in Switzerland. KORA Report No. 13e.
- Vandel, J.M., Stahl, P., Herrenschmidt, V. & Marboutin, E. (2006). Reintroduction of the lynx into the Vosges mountain massif: From animal survival and movements to population development. *Biol. Conserv.* **131**, 370–385.
- Weingarth, K., Zeppenfeld, T., Heibl, C., Heurich, M., Bufka, L., Daniszová, K. & Müller, J. (2015). Hide and seek: extended camera-trap session lengths and autumn provide best parameters for estimating lynx densities in mountainous areas. *Biodivers. Conserv.* **24**, 2935–2952.
- Werdelin, L. & Olsson, L. (1997). How the leopard got its spots: a phylogenetic view of the evolution of felid coat patterns. *Biol. J. Linn. Soc.* **62**, 383–400.
- Williams, B., Nichols, J. & Conroy, M. (2002). *Analysis and management of animal populations*. 817 pg. Academic Press.
- Zimmermann, F., Breitenmoser-Würsten, C., Molinari-Jobin, A. & Breitenmoser, U. (2013). Optimizing the size of the area surveyed for monitoring a Eurasian lynx (*Lynx lynx*) population in the Swiss Alps by means of photographic capture-recapture. *Integr. Zool.* **8**, 232–243.

# Acknowledgements

This research was funded by the [OAK Foundation](#) grant number OCAY-11-136 and partly funded by the European Commission through the Operational Programme 'Environment', grant number SMIS 102086. We thank Bârsa-Braşov, Jderul-Argeş and GTS Muntenia-Argeş Hunting Associations for facilitating access on the game management units and for valuable information on lynx movement. We thank Piatra Craiului National Park Administration for access permits and logistic support. We thank Ion Bughianu, Liviu Bulgaru, Adi Ciocan, Adi Fulga, Viorel Ganci, Radu Geantă, Dan Nicolae, Cora and Enya Promberger, Răzvan Rohan, Maria Spătaru, Bogdan Vasilescu and David Zărnovan for help with camera installation and check. We thank Mihai Banu, Remus Cucu and Radu Vulcu for independently installing and checking four traps within Piatra Craiului National Park during the last field session. We thank Fridolin Zimmerman and Mihai Zotta for guidance during early stages of designing the field surveys, and Tomaž Skrbinšek, Mihai Pop and Marissa Dyck for help with the modelling workflow. We thank Tomaž Skrbinšek for kindly sharing the R script behind Figure 5. We thank Dragoş Lazarin for designing the report template.





**FOUNDATION  
CONSERVATION CARPATHIA**

27 Calea Feldioarei  
Brasov – Romania 500471  
Tel./Fax: +40 368 45 24 11  
info@carpathia.org

[www.carpathia.org](http://www.carpathia.org)