Abundance Estimation of Himalayan Lynx Through Multiple Modelling Approach

By

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2020

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A thesis submitted in partial fulfillment of the requirements for the

Degree of

Master of Philosophy

In

Wildlife Ecology

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بسم الله الرحمن الرحيم

CERTIFICATE

This dissertation "Abundance Estimation of Himalayan Lynx Through Multiple Modelling Approach" submitted by **Muhammad Ilyas,** is accepted in its present form by the Department of Zoology, Faculty of Biological Sciences, Quaid-i-Azam University, Islamabad as satisfying the thesis requirement for the degree of Master of Philosophy in Wildlife Ecology.

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This thesis is dedicated

to

My loving and caring parents and siblings,

to

My teachers for training and guiding me up to this stage

Declaration

The material and information contained in this thesis is my original work. Work of others if used has been acknowledged and referred. Neither the thesis nor any part of it has been previously presented for any other degree.

Muhammad Ilyas

Acknowledgements

All glories and praises to Allah Almighty, the fountain of all goodness, and the ultimate source of knowledge, power and wisdom. His countless blessings gave me strength to work, mind to think, words to write and enabled me to reach this stage in the journey of learning. My Best commendations to the source of light and guidance for bewildered, The Holy Prophet Muhammad (ﷺ).

Heartful thanks to my research supervisor **Dr Muhammad Ali Nawaz**, for guiding me from the first day of the initiation and planning of work to the last day of writing thesis. I am sincerely thankful to him for his valued suggestions, developing appropriate field work approach, infusing into me the ethics of scientific research and encouraging me to dig and explore new ideas.

I am especially thankful to my class fellow and friend **Muhammad Asif** who helped me throughout my research work from day first till the last day of this thesis. I am also thankful to my class fellow **Mahvish Rauf** who actively participated in the first phase of field work.

Bundle of thanks to **Rubina Noor** (Sub Divisional Wildlife Officer) from National Parks Khyber Pakhtunkhwa (KP) Wildlife Department, Peshawar, for her continuous support from the day of signing of MOU for this study till the completion of field work. Without her help this study would not have been completed with ease.

I humbly acknowledge **National Parks Project, Khyber Pakhtunkhwa Wildlife Department** and **Pakistan Snow Leopard and Ecosystem Protection Program (PSLEP)** for providing funds and field equipments without which this research was not possible. Sincere thanks to **Pakistan Metrological Department** for providing climatic data for this study.

I owe my sincere thanks to the **Fathul Bari** (Lecturer) and his students, **Sami Ullah** and **Muhammad Younas** from University of Chitral, for participating in the field work and making this research possible.

Thanks to **Chitral Gol National Park** office staff especially **Irshad Ahmad** (Divisional Forest.Officer) and **Ali** (Office Assistant) for providing logistics and field staff who worked day and night to make field work successful. I am thankful to all other staff members who actively participated in field activities especially **Zakir Ullah, Israr Ullah,** and **Jan Muhammad**.

Special thanks to the Snow Leopard Foundation (Chitral office) for providing accommodation and help during my field work especially **Ejaz Ur Rehman**, for helping in planning the field work and supported me throughout my stay at Chitral.

I owe my sincere gratitude and very special thanks to my family members for the faith they have in me, bearing me through the thick and thin of my study tenure. Without their support, co-operation and encouragement this endeavor would not have been possible.

I am also thankful to all members of CCL, especially **Shakeel Ahmad,** and **Hussain Ali.**

I highly acknowledge his company and moral support and say special thanks to my dearest and nearest friend **Muhammad Asim,** PhD candidate at Parasitology Lab.

Finally, I humbly acknowledge all those who directly or indirectly contributed in accomplishing this thesis.

Muhammad Ilyas

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Acronyms and Abbreviations

AIC	Akaike Information Criterion
AICwt	Akaike Information Criterion weight
AICΔ	Akaike Information Criterion Delta
CE	Critically Endangered
CGNP	Chitral Gol National Park
CMR	Capture-Mark-Recapture
CPW	Colorado Parks and Wildlife
DEM	Digital Elevation Model
EN	Endangered
GIS	Geographic Information System
GLMs	Generalized Linear Models
GPS	Global Positioning System
HDI	Highest Density Interval
ISE	Instantaneous sampling estimator
IUCN	International Union for Conservation of Nature
km^2	Square Kilometre
LCI	Lower Confidence Interval
MCMC	Markov Chain Monte Carlo
NB	Negative Binomial
NDVI	Normalized difference vegetation index
P	Poisson
PAs	Protected Areas
REM	Random Encounter Model
REST	Random Encounter and Staying Time
SC	Spatial Count
SE	Standard Error
STE	Space to Event
TTE	Time to Event
UCI	Upper Confidence Interval
VCC	Village Conservation Committee
ZIP	Zero Inflated Poisson

Abstract

Abundance is the key parameter of concern to wildlife management and conservation. Different filed and analytical methods are in practice to obtain and analyse data for wildlife abundance estimation. Owing to their low cost, easy management and production of large data sets, camera traps have been the tool of choice in field surveys. Data from camera traps have widely been used to estimate population parameters for marked animals using Capture-Recapture approach. Since this approach requires identification of individuals, which limits its usefulness as most of the photo-captured species either totally lack identifiable markings or have unclear markings. Several analytical approaches are emerging to address this limitation so that the population parameters of unmarked animals are estimated. This study tried to compare abundance estimates for the different unmarked approaches using Himalayan lynx as a case study, in the Chitral Gol National Park (CGNP), Khyber Pakhtunkhwa, Pakistan. The species has elusive and nocturnal behaviour and occurs at the low densities.

Out of the total 103 grids of 1 km^2 over the 77.5 km^2 study area, cameras were installed at 30 grids. Due to some technical errors 5 cameras were excluded from the analysis. A database for the pictures from the remaining 25 cameras were created, to simplify data retrieval and analysis. The modelling approaches used in this study included N-Mixture models, Random Encounter Modelling (REM), Space and Time models and Spatial Count (SC) models.

The operational 25 cameras resulted in 1125 camera trap days. Lynx was captured 16 times at 6 trap sites. The N-Mixture model with the Poisson approach estimated abundance at 7.88 individuals for the CGNP. REM estimated an abundance of 2.69 individuals with a density of $3.47/100km^2$. The abundance estimates for the three approaches of Space and Time Modelling were 5.74±1.78(SE), and 12.52±3.76(SE), 30.5±15.5(SE), for Time-To-Event (TTE), Space-To-Event (STE) and Instantaneous Sampling Estimator (ISE), respectively. The SC model reported an abundance of 6.19±4.68(SD) with density of $8/100km^2$.

The population estimates derived from different models ranged between 3-30 lynx in CGNP, which is a huge variation and limits applicability of such results for the management purpose. However, estimates of four models (N-Mixture, TTE, STE, and

SC) are closer 6-12 individuals, and seem more realistic for the study area. REM appears to underestimate the population, while ISE has overestimated.

We believe these analytical approaches has a promise for investigating populations of unmarked animals, however with small data sets it is difficult to get agreement between them. Increasing sample size and combining data from multiple sources can potentially solve this issue.

Chapter 1

INTRODUCTION

Abundance is the number of individuals of a species present over a sampling area. It is a parameter of relative representation of a specie in a community (Wright 1991) while the total number of species occurring in the area denote the species richness of the community (Lande 1996). The quality of the underlying habitat can be determined by species richness and abundance (Broadley et al. 2019). Ahumada et al. (2011) reported higher levels species richness and abundance in areas with undamaged or uninterrupted forests compared to areas with fragmented forests. A study over Sumatran tiger reported higher abundance of the tiger and its prey in areas with no or low anthropogenic activities compared to high disturbances (O'Brien et al. 2003).

Being a key estimate needed for species' status assessment, abundance is the variable of interest in monitoring and research programs for animal populations that aim at conservation and management (Williams et al. 2002; Nichols & MacKenzie 2004; Fuller et al. 2016; Clare et al. 2017; Burgar et al. 2018). Effective practices such as efforts for protection of threatened species (Bradley et al. 2017), addressing human-wildlife conflicts (McGregor et al. 2015) and sustainable harvesting (Kachel et al. 2017) require unbiased information on abundance (Tobler et al. 2008). Clear understanding of abundance is especially important for management of rare and vulnerable species so that any decline in population is realized, and addressed through well informed and timely strategies (Buckland et al. 2005).

Monitoring carnivores' populations in a community is often the target of conservation efforts because of their role in maintaining the integrity of the ecosystem (Gompper et al. 2006). Nevertheless this is a daunting task, most carnivores have sparse distribution over large landscapes, rugged and remote habitats, lower densities and the nocturnal and elusive behavior (Mills 1996; Long et al. 2007; Broekhuis & Gopalaswamy 2016; Young et al. 2019). Therefore, to derive reliable estimates and design effective monitoring strategies for carnivores, intensive field efforts are usually required (Kindberg et al. 2009).

Modern technological tools characterized by better development, low cost and ease of availability are becoming an essential part of conservation efforts (Berger-Tal & Lahoz-

Monfort 2018). They improved the process of collecting more and refined data about monitoring of wildlife, their habitat and threats (Snaddon et al. 2013; Pimm et al. 2015; Berger-Tal & Lahoz-Monfort 2018). The use of satellites and drones for landcover classification and wildlife monitoring (Turner et al. 2003; Kakaes et al. 2015), GPS, sensor tags and camera traps for monitoring are the examples of such technology (Hussey et al. 2015; Kays et al. 2015; Berger-TAL & Saltz 2015). They have replaced the old practice of invasive and labour-intensive field surveys (Burton et al. 2015). Also, the emerging trends in molecular biology, genetic engineering and bioinformatics are changing the conservation approaches e.g., eradication of invasive species through gene drive techniques (Owens 2017) and species de-extinction through cloning (O'Brien 2015).

Similarly, traditional and simple statistical methods have been replaced by complex modern statistical tools e.g., Spatially-Explicit-Capture-Recapture (SECR) models, an extension of classical capture-recapture models that incorporate the underlying spatial processes (Borchers & Efford 2008; Royle & Young 2008). SECR methods have been used for density and abundance estimation of different animal species including mammals (Royle et al. 2011), birds (Mollet et al. 2015), amphibians (Muñoz et al. 2016), sharks (Bradley et al. 2017) and even invertebrates (Torres-Vila et al. 2012). These methods can be applied to data obtained from camera trap surveys (Avgan et al. 2014), genetic sampling (Gardner et al. 2010b) and acoustics (Dawson & Efford 2009), to reliably estimate densities.

The availability of high-quality and low-cost camera traps has increased their use in field studies, to answer questions of demographics and behavior for many species. Camera traps are advantageous being more resource-efficient when surveying large areas, non-invasive and less risky to target and non-target animals and able to monitor many variables on many species simultaneously (Silveira et al. 2003; Gompper et al. 2006; Bowkett et al. 2008). Camera traps have been used to answer different ecological questions; presence-absence of species (Linkie et al. 2007), behavioral aspects (Harmsen et al. 2009), biodiversity estimates (McCarthy et al. 2010), and relative abundance(O'Brien et al. 2003). They have particularly enhanced the ability to study elusive carnivores (Augustine et al. 2018).

Camera trap data has been used for abundance estimation of many species using the traditional capture-recapture (Otis et al. 1978) and the modern spatial capture-recapture models (Royle & Young 2008). The SECR methods model the spatial dimensions of both ecological and observational processes. This is done by modelling a spatial point process for latent locations of individuals, then modelling capture probability as a function of location (Borchers & Fewster 2016). As both these methods require the unique identity of the captured individuals, they had been frequently used to estimate abundance of felids having a clear and unique pelage pattern (Karanth & Nichols 1998; Karanth et al. 2006; Royle et al. 2009; Sollmann et al. 2013a). Table 1.1 enlists studies where SECR approach has been used for estimating density and abundance of carnivores, using camera trap data.

The requirement of unique identification of individuals limits the use of CMR/SECR for camera trap data of those species that either totally lack natural markings or have abstruse markings (Evans & Rittenhouse 2018; Moeller et al. 2018; Dey et al. 2019). As most of the elusive species are more active at night hence the pictures captured may not be identified as unique individuals with surety. Although other methods, such as telemetry data and genetic sampling, can be combined with camera trap data to maximize the use of CMR, yet it has certain limitations (Sollmann et al. 2013a). Telemetry is invasive, requires more logistics, and will increase the cost (Long et al. 2012). Genetic data can be obtained from multiple sources e.g. fecal samples and hair follicles collected from rub-posts, hair snares, kill sites, or beds to identify individuals (Crowley & Hodder 2017). The quantity and quality of the genetic material limit its use to drive meaningful information (Long et al. 2012).

To tackle these limitations, alternative approaches are emerging that don't need individual identification. These include N-Mixture models (Royle & Nichols 2003; Royle 2004), Random Encounter Model (REM) (Rowcliffe et al. 2008; Cusack et al. 2015), Random Encounter and Staying Time (REST) model (Nakashima et al. 2018), Space and Time models (Moeller et al. 2018; Loonam 2019; Moeller 2019), and Spatial Count (SC) (Evans & Rittenhouse 2018) or Spatial Mark Resight models(Chandler & Royle 2013). The first 4 approaches estimate abundance and density of unmarked populations based on detection probability and detection rates (Royle 2004; Nakashima et al. 2018; Loonam 2019). The SC models are usually used for partially marked populations when counts of unmarked detections are combined with individual capture histories of marked animals (Efford & Hunter 2018). Table 1.2 enlists studies incorporating camera trap data for estimation of unmarked mammal populations.

Himalayan lynx or Turkestan lynx (*Lynx lynx isabellinus*), is a sub-species of Eurasian lynx (*Lynx lynx*). It is the largest member among four species of the genus *Lynx*. The other three species are bobcat (*Lynx rufus*), Canada lynx (*Lynx canadensis*), and Iberian lynx (*Lynx pardinus*) (Heaver & Waters 2019).

The Eurasian lynx is the most widely distributed felid among the extant felids (Breitenmoser et al. 2015) across the Eurasia (İbİş et al. 2019). Globally it is listed as "least concern" on The International Union for Conservation of Nature (IUCN) Red List of Threatened Species (Breitenmoser et al. 2015) with the exception of some of the isolated subpopulations in Europe that are listed as Endangered (EN) and Critically Endangered (CR) (Breitenmoser et al. 2015; İbİş et al. 2019).

Being a solitary predator (López-Bao et al. 2019), lynx usually inhabits areas having enough forest cover with ample prey amount (Niedziałkowska et al. 2006; İbİş et al. 2019; López-Bao et al. 2019). Lynx usually preys upon medium and large sized ungulates and other small mammals e.g. hare, squirrel, vole, and shrew etc. Occasionally it also preys upon carnivore species including golden jackal, red fox, stray dogs, stone marten and domestic cats (Mengüllüoğlu et al. 2018).

They are seasonal breeders (Painer et al. 2014) giving birth to 1-4 kittens with an optimal litter size of 2 (Gaillard et al. 2014; Antonevich et al. 2020) and survival rate of 50% for the first year (Schmidt 1998; Andrén et al. 2006; Breitenmoser-Würsten et al. 2007). The breeding occurs from mid-May to mid-June (Nilsen et al. 2012; Antonevich et al. 2020). Survival is mainly dependent on the availability of enough food (Andrén et al. 2006).

Many studies have focused on abundance estimation of lynx by using data from camera traps, telemetry and genetic sampling. The common methodological approach of these studies is SECR. Most of the studies are from across the Europe focusing on monitoring of the reintroduction programs (Hetherington & Gorman 2007; Weingarth et al. 2012; Pesenti & Zimmermann 2013; Blanc et al. 2013; Zimmermann et al. 2013). A few studies have assessed the Asian populations of lynx (Avgan et al. 2014). Two studies have reported lynx population dynamics from Pakistan (Din & Nawaz 2010; Din et al. 2015) but both lack sound statistical analysis.

The present study aims to use the unmarked modelling approaches, for abundance estimation of Himalayan lynx from the camera trap data. The soft and dense fur characterized by monochromatic greyish or white-brownish color without spots or only faintly visible spots (Kitchener et al. 2017), does not allow for individual recognition, thus make lynx a good candidate for evaluating unmarked approaches for abundance estimation.

Objectives

The current study was designed to assess lynx population in Pakistan, with following two secondary objectives:

- Assess effectiveness of unmarked modelling approaches for elusive carnivores
- Assess effects of environmental variables on abundance estimation

Table 1.1 Studies on camera trap data using SECR modelling for abundance estimation of carnivores

Family	Species	References
Felidae	Tiger (*Panthera tigris*)	(Harihar et al. 2011; Gopalaswamy et al. 2012; Carter et al. 2012; Singh et al. 2014; Xiao et al. 2016; Wang et al. 2018; Naing et al. 2019; Ngoprasert & Gale 2019; Dey et al. 2019)
	Pampas Cat (*Leopardus colocolo*)	(Gardner et al. 2010a; Caruso et al. 2012; Huaranca et al. 2019)
	Jaguar (*Panthera onca*)	(Sollmann et al. 2013c; Borchers et al. 2014; Figel et al. 2016; Petit et al. 2018)
	Andean Mountain Cat (*Leopardus jacobita*)	(Reppucci et al. 2011; Huaranca et al. 2019)
	Leopard (*Panthera pardus*)	(Harihar et al. 2011; Strampelli et al. 2018; Devens et al. 2018; Lukarevskiy & Lukarevskiy 2019; Havmøller et al. 2019)
	Snow Leopard (*Panthera uncia*)	(Alexander et al. 2015, 2016)
	Cheetah (*Acinonyx jubatus*)	(Brassine & Parker 2015; Broekhuis & Gopalaswamy 2016; Linden et al. 2019)
	Cougar (*Puma concolor*)	(Guarda et al. 2017; Murphy et al. 2019)
	Geoffroy's Cat (*Leopardus geoffroyi*)	(Caruso et al. 2012; Tirelli et al. 2019)

	Leopard Cat (*Prionailurus bengalensis*)	(Bashir et al. 2013; Mohamed et al. 2013; Srivathsa et al. 2015; Chua et al. 2016)
	Margay (*Leopardus wiedii*)	(Pérez-Irineo et al. 2017)
	Eurasian Lynx (*Lynx lynx*)	(Weingarth et al. 2012, 2015; Blanc et al. 2013; Zimmermann et al. 2013; Kubala et al. 2019; Gimenez et al. 2019)
	Iberian Lynx (*Lynx pardinus*)	(Sarmento & Carrapato 2019)
	Serval (*Leptailurus serval*)	(Ramesh & Downs 2013; Loock et al. 2018; Edwards et al. 2018)
	Bobcat (*Lynx rufus*)	(Thornton & Pekins 2015; Jacques et al. 2019)
	Clouded Leopard (*Neofelis nebulosa*)	(Singh & Macdonald 2017; Naing et al. 2019)
	Ocelot (*Leopardus pardalis*)	(Martínez-Hernández et al. 2015; Penido et al. 2016; Rocha et al. 2016; Pérez-Irineo et al. 2017)
	African Golden Cat (*Caracal aurata*)	(Bahaa-el-din et al. 2016)
	Marbled Cat (*Pardofelis marmorata*)	(Hearn et al. 2016; Singh & Macdonald 2017; Naing et al. 2019)
	European Wildcat (*Felis silvestris*)	(Kilshaw et al. 2015)

	Domestic Cat (*Felis catus*)	(McGregor et al. 2015; Cove et al. 2018; Rees et al. 2019)
Ursidae	Asian Black Bear (*Ursus thibetanus*)	(Ngoprasert et al. 2012)
	Sun Bear (*Helarctos malayanus*)	(Ngoprasert et al. 2012)
	Brown Bear (*Ursus arctos*)	(Ngoprasert et al. 2012)
	Andean Bear (*Tremarctos ornatus*)	(Burton et al. 2018)
	Ferret (*Mustela furo*)	(Molina et al. 2017)
Mustelidae	Wolverine (*Gulo gulo*)	(Royle et al. 2011)
	American Pine Marten (*Martes americana*)	(Siren et al. 2016)
	American Badgers (*Taxidea taxus*)	(Gould & Harrison 2017)
Hyaenidae	Aardwolf (*Proteles cristata*)	(O'Brien & Kinnaird 2011)
	Spotted Hyena (*Crocuta crocuta*)	(O'Brien & Kinnaird 2011)

	Brown Hyena (*Parahyaena brunnea*)	(O'Brien & Kinnaird 2011; Edwards et al. 2019)
	Striped Hyena (*Hyaena hyaena*)	(Tichon et al. 2017)
Canidae	Red Fox (*Vulpes vulpes*)	(Carter et al. 2019)
	Grey Wolf (*Canis lupus*)	(Mattioli et al. 2018)
Procyonidae	Northern Raccoon (*Procyon lotor*)	(Sollmann et al. 2013b)

Table 1.2 Studies on modelling approaches for abundance estimation of partially marked and unmarked mammal species based on camera trap data

Modelling Approach	**Species**	**Reference**
N-mixture models	White-Tailed Deer (*Odocoileus virginianus*)	(Keever et al. 2017; Haus et al. 2019; Kautz et al. 2019)
	Domestic Cat (*Felis catus*)	(Nichols et al. 2019; Taggart et al. 2019)
	Fisher (*Pekania pennanti*)	(Furnas et al. 2017)
	Chital (*Axis axis*)	(Kafley et al. 2019)

	Sambar (*Rusa unicolor*)	(Kafley et al. 2019)
	Caracal (*Caracal caracal*)	(Singh et al. 2015)
	Barking Deer (*Muntiacus muntjac*)	(Kafley et al. 2019)
	Wild Boar (*Sus scrofa*)	(Kafley et al. 2019)
	Striped Hayena (*Hyaena hyaena*)	(Shamoon & Shapira 2019)
	Eurasian Lynx (*Lynx lynx*)	(Blanc et al. 2014)
Random Encounter Model	European Pine Marten (*Martes martes*)	(Manzo et al. 2012)
	Red Deer (*Cervus elaphus*)	(Ghoddousi et al. 2019)
	Roe Deer (*Capreolus capreolus*)	(Marcon et al. 2019)
	Iberian Lynx (*Lynx pardinus*)	(Garrote et al. 2019)
	Wild Boar (*Sus scrofa*)	(Ghoddousi et al. 2019)
	Baird's Tapir (*Tapirus bairdii*)	(Carbajal-Borges et al. 2014)
	Crab-Eating Foxes (*Cerdocyon thous*)	(Monteiro-Alves et al. 2019)
	Lion (*Panthera leo*)	(Cusack et al. 2015)
	European Roe Deer (*Capreolus capreolus*)	(Romani et al. 2018)
	Red-Necked Wallabies (*Notamacropus rufogriseus*)	(Havlin et al. 2017)
	Bawean Warty Pig (*Sus blouchi*)	(Rademaker et al. 2016)
	Bawean Deer (*Axis kuhlii*)	(Rahman et al. 2017)
	Brown Bear (*Ursus arctos*)	(Popova et al. 2018)
	Grevy's Zebra (*Equus grevyi*)	(Zero et al. 2013)
	Cacomistle (*Bassariscus sumichrasti*)	(Hernández-Sánchez et al. 2017)
	Striped Hog-Nosed Skunk (*Conepatus semistriatus*)	(Hernández-Sánchez et al. 2017)
	Tayra (*Eira barbara*)	(Hernández-Sánchez et al. 2017)

	Greater Grison (*Galictis vittata*)	(Hernández-Sánchez et al. 2017)
	Jaguarundi (*Herpailurus yagouaroundi*)	(Hernández-Sánchez et al. 2017)
	Ocelot (*Leopardus pardalis*)	(Hernández-Sánchez et al. 2017)
	Margay (*Leopardus wiedii*)	(Hernández-Sánchez et al. 2017)
	Bridled Weasel (*Mustela frenata*)	(Hernández-Sánchez et al. 2017)
	Coatimundi (*Nasua narica*)	(Hernández-Sánchez et al. 2017)
	Kinkajou (*Potos flavus*)	(Hernández-Sánchez et al. 2017)
	Grey Fox (*Urocyon cinereoargenteus*)	(Hernández-Sánchez et al. 2017)
	Irish Hare (*Lepus timidus*)	(Caravaggi et al. 2016)
Random Encounter and Staying Time	Blue and Red Duikers (*Philantomba sps.*)	(Nakashima et al. 2018)
	Yellow-Backed, Ogilby's and Bay Duikers (*Cephalophus sps.*)	(Nakashima et al. 2020)
Spatial Count or Spatial Mark-Resight	American Black Bear (*Ursus americanus*)	(Evans & Rittenhouse 2018; Burgar et al. 2019)
	Black-Tailed Deer (*Odocoileus hemionus*)	(Macaulay et al. 2019)
	Cougar (*Puma concolor*)	(Sollmann et al. 2013a; Rich et al. 2014; Zanón-Martínez et al. 2016)
	Brown bear (*Ursus arctos*)	(Whittington et al. 2018)
	Lion (*Panthera leo*)	(Kane et al. 2015)
	Woodland Caribou (*Rangifer tarandus caribou*)	(Burgar et al. 2019)
	Grey Wolf (*Canis lupus*)	(Burgar et al. 2019)
	Coyote (*Canis latrans*)	(Burgar et al. 2019)
	Moose (*Alces alces*)	(Burgar et al. 2019)
	White-Tailed Deer (*Odocoileus virginianus*)	(Burgar et al. 2019)
	Red Fox (*Vulpes vulpes*)	(Ramsey et al. 2015; Jimenez et al. 2019)

	Fisher (*Pekania pennanti*)	(Sweitzer et al. 2015; Burgar et al. 2018)
Space and Time	Cougar (*Puma concolor*)	(Loonam 2019)
	Elk (*Cervus canadensis*)	(Moeller et al. 2018)

Chapter 2

MATERIALS AND METHODS

2.1 Study Area

Chitral Gol National Park (CGNP) lies in the Hindu Kush range, about three km in west of the Chitral town, in the Khyber-Pakhtunkhwa province of Pakistan. It has an estimated area of 77.5 km^2, and geographically located at 35.89508N Latitude and 71.69078E (Figure 2.1). It was declared as a royal hunting reserve in 1880's followed by ban on livestock grazing in 1907. In 1971 the then Commissioner of Malakand division designated it as a wildlife sanctuary for five years (Green 1990). It was declared as National Park on 18th October 1984 by the provincial government under the IUCN's Category II of Protected Areas (PAs) for the protection of populations and habitat of endangered Kashmir markhor (*Capra falconeri cashmiriensis*) and snow leopard (*Panthera uncia*). The Khyber Pakhtunkhwa Wildlife Department exercises the powers for administration and management of the park, under Khyber Pakhtunkhwa Wildlife and Biodiversity (Protection, Preservation, Conservation and Management) Act, 2015 (Pakhtunkhwa 2015). With a gradient varying from 45° to 120° (Hess 2002) this park is a narrow valley having steep slopes and sharply defined ridges. Elevation of the park area ranges from 1500 to 4950 m (Akbar 1974; Khan et al. 2011).

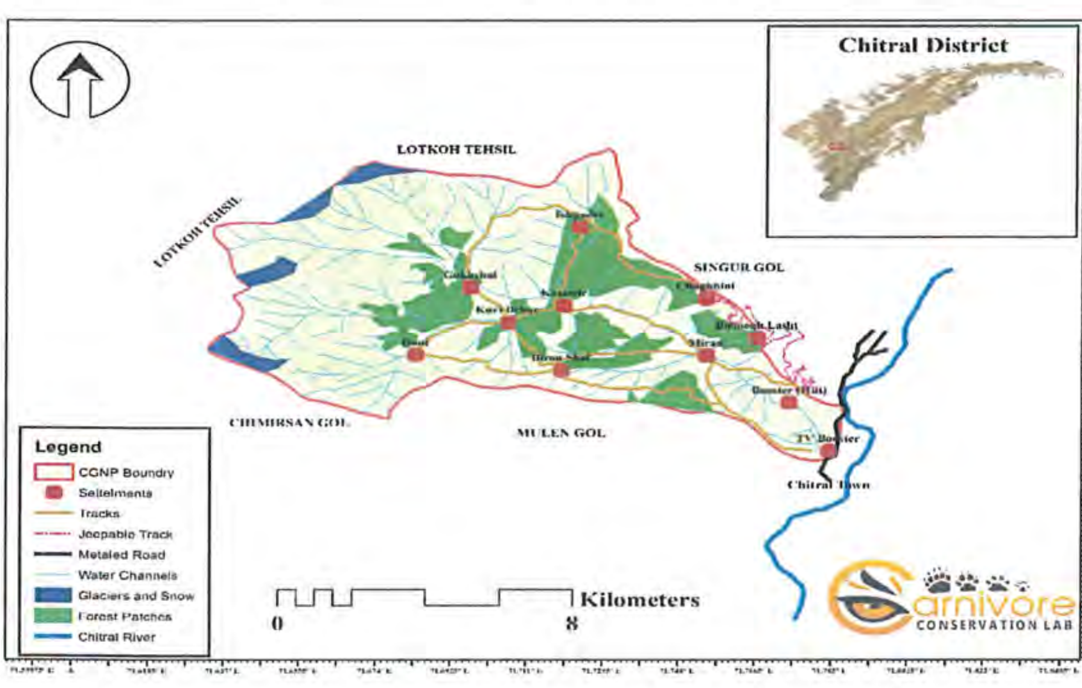

Figure 2.1 Study area map

2.1.2 Flora and Fauna

The documented 31 plant species belonging to 21 families comprises trees, herbs and shrubs (Khan et al. 2011). The dominant trees include deodar (*Cedrus deodara*), Chilghoza Pine (*Pinus gerardiana*), Chir Pine (*Pinus roxburghii*), Oak (*Quercus ilex*), and various flowering trees. The common vegetation of CGNP includes *Rosa webbiana, Artemisia brevifolia, Artemisia maritima*, *Ferula narthex* and *Ephedra gerardiana* (Khan et al. 2010).

The park and its buffer areas are home to some magnificent species, like the national animal of Pakistan -Markhor, which has largest and stable population in the CGNP (Arshad et al. 2012; Ashraf et al. 2014). Snow leopard was first ever photo captured in 1970 in this area by famous zoologist George Schaller (O'Connor 2018). Common leopard (*Panthera pardus*) was also photo-captured in the park during the current study. Similarly, leopard cat (*Prionailurus bengalensis*), Eurasian lynx (*Lynx lynx)*, fox (*Vulpes vulpes*), grey wolf (*Canis lupus*), golden jackals (*Canis aureus)*, pika (*Ochotona species*), weasel (*Mustela* spp), cape *hare* (*Lepus capensis*) are vital mammals residing in the park and its buffer area (Roberts 1997; Sheikh & Molur 2005; Din & Nawaz 2010; Din et al. 2013). It has a diverse avian fauna including Lammergier vulture (*Gypaetus barbatus*), Himalayan snow cock (*Tetraogallus himalayensis*), chukar (*Alectoris chukar*), Himalayan Griffon vulture (*Gyps himalayensis*), golden eagle (*Aquila chrysaetos*) and various other passerine birds (Wildlife of Pakisatn 2019).

2.1.3 Ecological Zone and Habitat Types

Alpine meadows and dry temperate forests are the major habitat types, because of low rain and high elevation. The heavy winter snow falls, and long dry summers result in drought-resistant and cold tolerant vegetation, in and around the park.

2.1.4 Climatic Conditions

Characterized by both seasonal and diurnal extremes, the climate is dry temperate. The hot summer of Arandu in the south, to the arid and cool Broghil in the north, represent the diversity in temperature. The park area is out of the reach of Monsoon and receives 462 mm mean annual rainfall (Hess 2002; Khan et al. 2010). The average maximum temperature during the study period was 36.5°C and minimum temperature was 19.6°C. The average rainfall was 0.5mm and average relative humidity at 0000 UTC was 69% (Table 2.1) (Pakistan Meterological Department 2019)

Table 2.1 Climatic conditions of study area for the study period

2019	maximum (°C)	minimum (°C)	rainfall (mm)	relative humidity 0000 UTC (%)	relative humidity 1200 UTC (%)
June	35.8	17.5	0	63	27
July	36.6	21.2	0.4	70	32
August	37.1	20	1	74	27

2.1.5 Socioeconomic Conditions

There are 12 village conservation committees (VCCs) in core zone, 3 in buffer zone and 2 buffer zone conservancies which surround the park. The total number of households in core zone villages is 2370 with the total population of 18076 and the average literacy rate 64% while its buffer villages have 626 households with the total population of 4895 and average literacy rate is 46%. These custodial villages have same resources and problems with little variation. The main occupation of urbanized villages like Zargarandeh and Jang Bazar is small businesses while residents of other villages like Balach, Mughlandeh and Singoor still depend on agriculture for their livelihoods. The inhabitants of Kalsah and Sheikhans still practice their traditional goat farming and production of cereals. With the increasing population in all these villages, land is becoming scarce which is common problem for them. Pollution and other urbanization related problems are surfacing now which were unknown in the recent past. With the availability of better alternative, some villages have given up the goat herding in park while others did the same voluntarily. Fuel wood is main source of energy as alternance sources are either unavailable or unfordable. Energy problems remain as major problem in these villages especially in cold winters as prices of fuel wood surges in markets.

2.2 Camera Trapping

To collect the field data for lynx, we installed 30 remote cameras for the period of 45 days from 17th June 2019 to 7th August 2019. A map of the study area was developed in ESRI ArcGIS 10.6 with 103 grids of 1×1 km. Cameras were installed in 30 grids which were accessible (Figure 2.2). GPS (Garmin, 62S) and study area map were used to navigate and find the grids and to record the locations of camera stations.

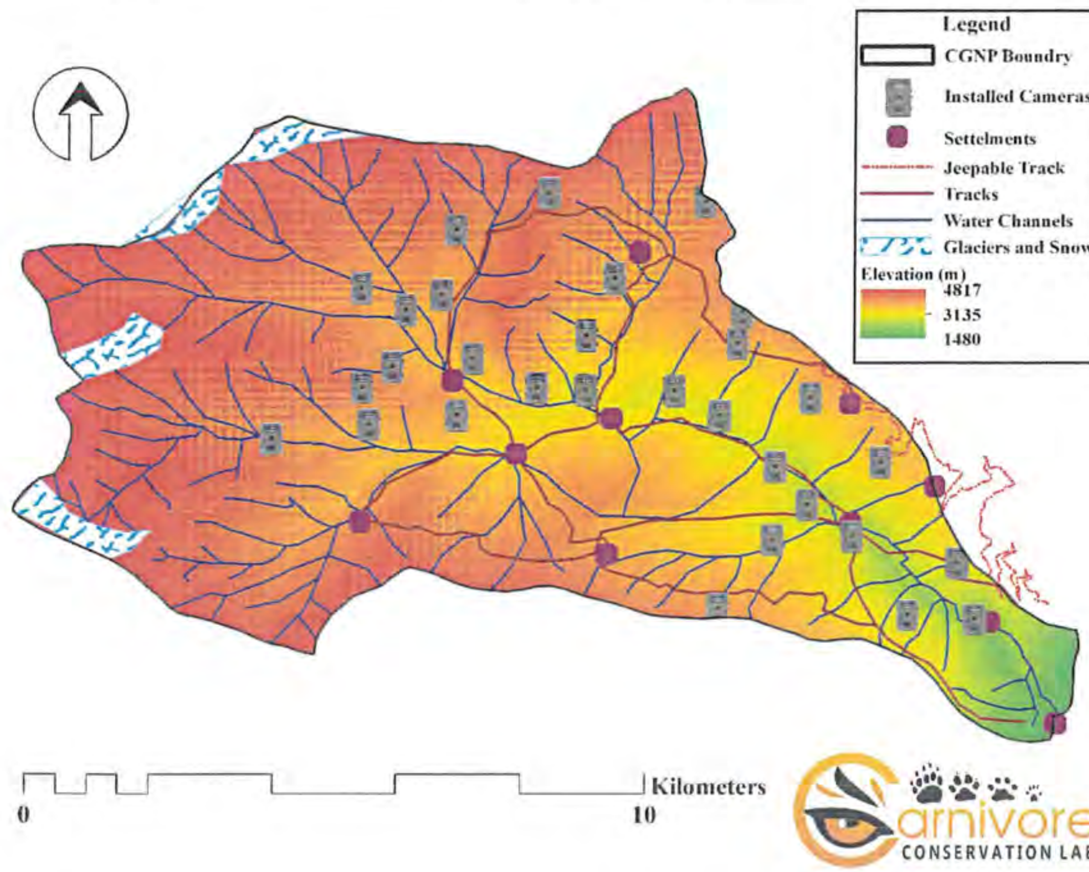

Figure 2.2 Map of Installed Cameras

During the installation of cameras, we avoided to install two cameras on the same track or ridge to reduce the autocorrelation across the cameras. Cameras were randomly installed across the study area to maintain representation of sample from each site and avoid clustering at one site. Some of the cameras were placed at the "T" stands (Figure 2.3) and some were placed on trees at a height of approximately 4-5 feet above the ground (Figure 2.4). Cameras were directed to the north, to avoid direct sunlight, in order to maintain the picture quality and all type of vegetation in the view of cameras was cleared to avoid obstruction and false triggering. Two different camera models were used with 15 Reconyx (Hyperfire2, H2FX) and 15 Apeman (H68). Both models use passive infrared (PIR) with high trigger sensitivity. These cameras were set to take burst of 3 pictures and one video with no delay between two triggers activation. Evanesce Predator Curiosity Lure (Kaatz Bros) and fish oil were used to bait camera stations.

We also recorded the observation covariates for each camera station, during camera installation, which include elevation, habitat type, terrain and substrate.

Figure 2.3 Installing a camera trap on "T" stand

Figure 2.4 An operational camera trap

2.3 Data Management

Colorado Parks and Wildlife (Cpw) Photo Warehouse (Ivan & Newkirk 2016) was used for management, organization and capture history generation for camera trap data. It is Microsoft Access® based free software for camera trap data management, been tailored through Visual Basic® for Applications (VBA) code for this purpose. It provides a Graphical User Interface with point-and-click menu items facilitating the input of details from camera deployments and importing photos with metadata. It stores the data within a relational database. The database can be used for species identification within the pictures, viewing, filtering and sub-setting data based on species of interest, specific study area , or specific season etc., and generating input files for statistical analysis such as animal activity patterns, abundance, density and occupancy. The capability of the application to accommodate multiple observers significantly augments the precision and productivity for managing huge data sets (Ivan & Newkirk 2016).

2.4 Data Analysis

Data was analysed using the currently practiced modelling approaches for abundance and density estimation of unmarked animals. These include the N-Mixture models for repeated counts (Royle 2004), Random Encounter Modelling (Rowcliffe et al. 2008), Space and Time models (Moeller et al. 2018) and Spatial Count models (Chandler & Royle 2013). All the analysis was done in R (Language and Environment for Statistical Computing) (R Core Team 2019) using the base environment and many user developed packages.

2.4.1 N-Mixture Models for Repeated Counts

N-mixture models are essentially generalized linear models (GLMs) requiring repeated counts for a survey period (Royle 2004; Chandler et al. 2011). Assuming a Poisson distribution with mean abundance λ, these models first estimate the local abundance N at site i, and then link the estimate with detection probability p. The given equation summarizes the process

$$O_{ij} \sim binomial\ (N_i p)$$

With Oij observations for site i at survey j. Using a log-link or a logit-link function, effects of covariates can be incorporated into the model associated with λ and p, respectively.

Although they have traditionally been practiced estimating site occupancy and detection for over a decade, they have been recently used for abundance estimation from camera trap data (Furnas et al. 2017; Taggart et al. 2019; Shamoon & Shapira 2019). Due to low detections of lynx for the current study, single day (24hrs) was used as a replicate, generating 45 replicates for the study period. The N-mixture models used for this study were exclusively based on presence/absence data (Royle & Nichols 2003).

The N-mixture models i.e. Poisson, zero-inflated Poisson (ZIP), and negative binomial (NB) were all executed using pcount() function from package Unmarked (Fiske & Chandler 2011). The pcount() requires a data frame created with unmarkedFramePCount() function which includes an *RxJ* matrix of the repeated count data, with *R* number of sites and *J* number of sampling periods per site and a list of site and observation level covariates. The capture history was treated as point count data following Kafley et al. (2019).

First no site or observation level covariates were used. The best suitable model based on lowest Akaike Information Criterion (AIC) value (Zou et al. 2019), was Poisson. This model was further evaluated for effect of covariates on detection. The covariates used were Normalized Difference Vegetation Index (NDVI), ruggedness and elevation. NDVI was calculated for the surveyed grids through Raster Calculator in QGIS 3.10 (QGIS Developent Team 2019) using Landsat 8 Operational Land Imager (OLI)/Thermal Infrared Sensor (TIRS) (Vermote et al. 2016) band-4 and band-5 scenes. Ruggedness was calculated through Raster Terrain Analysis in QGIS 3.10 using Digital Elevation Model (DEM) (Japan Spacesystems & ASTER Science Team 2019). Elevation for each camera station was recorded at installation. All values were standardized using standardize() function from package psycho (Makowski 2018).

Using different combinations of the three covariates 8 variants of the Poisson model were evaluated. Based on lowest AIC value, the model without covariates was selected and analysed for abundance estimation. Using empirical Bayes methods through ranef() function from unmarked, abundance for each sampling site was estimated. Finally, the total estimate was obtained by summing the estimate for all sites.

2.4.2 Random Encounter Modelling

Identification of individuals or marked animals are not required by REM. It can use detector layouts spaced irrespective of the population's home range size. REM estimates the abundance by modelling the detection count, area sampled by the remote camera, and an estimate of movement rate. The modelling assumes a geographic and demographic closure, random placement of cameras and no aversion or attraction to detectors (Rowcliffe et al. 2008; Manzo et al. 2012).

Camera trap data was used to estimate lynx abundance and density as an unmarked population by REM. The gremAbundance() and gremDensity functions from R package RandEM (Caravaggi & Lucas 2019) were used to estimate abundance and density respectively. The arguments needed for gremAbundance() function were detection count, camera detection radius, camera detection angle, 24hrs movement rate of the target species and study area in km^2. Detection count used was 16, camera radius as 15 m, camera detection angle as 42.9̊, movement rate for lynx as 31.2 km/24hrs (Jedrzejewski et al. 2002) and study area as 77.5 km^2.

2.4.3 Space and Time Modelling

The space and time modelling use three different approaches to estimate abundance of unmarked animals, i.e.., the Time-To-Event (TTE), the Space-To-Event (STE) and Instantaneous-Sampling-Estimator (ISE) (Moeller et al. 2018). These models were implemented through R package spaceNtime (Moeller 2019).

All the three models utilize camera deployment and photo information of all the pictures from the study period for abundance estimation, both of which were provided as separate data frames in accordance to the spaceNtime requirements (Moeller 2019). The camera deployment data frame had a unique ID for every camera, the set and pull; date and time of every camera, and area of the camera viewshed, while the photo information data frame had a unique ID for every camera, the date and time of each picture captured and the presence or absence of the target species as "1" or "0'.

Camera viewshed area was calculated using the equation,

$$a_c = \frac{\pi r^2 \theta}{360}$$

Where a_c is area of the viewshed, with camera maximum distance in meters as r and viewshed angle of θ in degrees. The calculated area for single camera was 84.23 m^2, with maximum distance of 15 meters and viewshed angle of 42.9̊.

Many R packages were used to streamline the preparation of required data frames. The read_exif() from package exifr (Dunnington & Harvey 2019) was used to read picture metadata, the capture time and date of the pictures was extracted through the select() function from package dplyr (Wickham et al. 2019). The parse_date_time() function from package Lubridate (Grolemund & Wickham 2011) was used to assign required class "POSIXct" to the date and time. Data was exported as spreadsheets to ease future operations. The reader() function from package reader (Cooper 2017) was used to import data from spreadsheets.

2.4.3.1 Time-To-Event

The time-to-event model quantifies the relationship between detection and encounter rate for abundance estimation (Bischof et al. 2014; Moeller et al. 2018). This model has four assumptions: (i) the number of animals should be Poisson-distributed, (ii) animals move randomly with respect to the cameras, (iii) movement speed of the target species, and (iv) population closure (Moeller et al. 2018; Loonam 2019). These assumptions are similar as that of the Random Encounter Modelling (REM) (Rowcliffe et al. 2008; Moeller et al. 2018). Time-to-event model incorporates repeated measurements of the amount of time that passes before an event of interest occurs to estimate the rate of that event. For abundance estimation from camera traps, a detection is the event of interest and the number of animals per view shed is the rate of interest (Loonam 2019).

The first step for TTE analysis is to specify sampling occasions. Both the sampling occasions and length were set to 1 minute to maximize encounter rate because of the low capture rate of the lynx for the present study. This was done through the build_occ() function from spaceNtime (Moeller 2019). Next step was to build encounter history through the function tte_build_eh(). The encounter history for TTE requires photo information, camera deployment information, sampling occasions and movement rate of the species of interest. The movement rate for lynx was taken as 1.3 km/hr (Jedrzejewski et al. 2002). The final step was to estimate abundance with function tte_estN_fn() by incorporating encounter history and total area of the study area.

This function first estimates density as;

$$\boldsymbol{TTE_{ik} \sim Exp(\lambda)} \qquad \text{Equation 1 (Loonam 2019)}$$

where TTE_{ik} is the time until an event occurs at camera k on occasion i, with an encounter rate λ.

2.4.3.2 Space-To-Event

STE model is just like TTE model, with the exception that it does not require animal movement rate data for estimating abundance. Hence it greatly improves its practical application as this secondary data is usually not available for most species. It also models the number of animals in view of a camera using the Poisson distribution. The difference is it collects data on the amount of space between animals rather than observing the time until an animal is detected.

The first step for STE analysis is to specify sampling occasions. Both the sampling occasions and length were set to 1 minute to maximize encounter rate because of the low capture rate of the lynx for the present study. The build_occ() function was used for this purpose. Encounter history for STE was built through the function ste_build_eh(). The encounter history for STE requires photo information, camera deployment information, and sampling occasions. The final step was to estimate abundance with function ste_estN_fn(), by incorporating encounter history and total area of the study area.

ste_estN_fn(), first estimates density using

$$\boldsymbol{STE_i \sim Exp(\lambda)} \qquad \text{Equation 2 (Loonam 2019)}$$

where λ is encounter rate, and STEi is the number of cameras sampled randomly at each time step, i, until a lynx is observed.

2.4.3.3 Instantaneous-Sampling-Estimator

The ISE is practically more useful for randomly deployed time-lapse cameras. It uses counts of animals for the cameras to estimate abundance. For several Spatio-Temporal replicates, the average count n_{ij} at location $i = 1, 2, \ldots, M$ and occasion $k = 1, 2, \ldots, K$ is an estimate of density (D) when divided by the cameras' viewable area (a_{ij}), as formulated in equation 10 from Moeller et al. (2018)

$$\widehat{\boldsymbol{D}} = \frac{\mathbf{1}}{\boldsymbol{K}} \cdot \frac{\mathbf{1}}{\boldsymbol{M}} \sum_{k=1}^{K} \sum_{m=1}^{M} \frac{n_{ik}}{a_{ik}}$$

This can be converted to abundance through

$$N^{\hat{}}=D^{\hat{}}A$$

The first step for ISE analysis is to specify sampling occasions. Both the sampling occasions and length were set to 2 minutes to maximize encounter rate because of the low capture rate of the lynx for the present study. The build_occ() function was used for this purpose. Encounter history for ISE was built through the function ste_build_eh(). The encounter history for ISE requires photo information, camera deployment information, and sampling occasions. The final step was to estimate abundance with function ise_estN_fn(), by incorporating encounter history and total area of the study area.

2.4.4 Spatial Count Model

Spatial count models are modified N-mixture models with extended fitness to incorporate the underlying spatial process. These models were introduced to estimate density and abundance of unmarked or partially marked animals (Chandler & Royle 2013; Evans & Rittenhouse 2018).

For sampling locations *i*, over sampling occasion *k*, detection counts n_{ik} are modelled as a function of the underlying spatial distribution of individual activity centres *(Si)*, and a function that describes the change in detection probability with change in distance between a sampling location and an activity centre.

Abundance parameters were estimated through Bayesian analysis, using Markov Chain Monte Carlo (MCMC) sampling. Bayesian analysis is the practice of redistributing prior credibility coherent with the observed new data (Kruschke 2013). Possibilities that are close to the data gain more credibility, while those far from the data lose credibility. The highest density interval (HDI) is the most reliable range of possibilities covering 95% of the posterior distribution (Pandong et al. 2018).

A hypothetical abundance of upper bound M is required to a uniform prior distribution on actual abundance N. The value for M should be enough large so that the probability that N=M is 0. For the current analysis M=60 was used.

Values for MCMC chain initiation, were selected by using prior distributions. During each iteration new values for $\boldsymbol{\sigma}$, $\boldsymbol{\lambda 0}$ and $\boldsymbol{Si}$ were proposed through random selection from a normal proposal distribution centred at the current accepted value (Chandler & Royle 2013). A uniform prior for $\sigma \sim U(3{,}000 \text{ m}, 4{,}500 \text{ m})$, based on knowledge of lynx space use, was specified using data from Pesenti and Zimmermann (2013) as no data was available on lynx space use from the study area.

All the analysis was done in R version 3.6.1 (R Core Team 2019) by adopting code from Chandler and Royle (2013). A matrix of the capture history and site coordinates used as SC model inputs were prepared as described in Evans and Rittenhouse (2018). SC model, with a single MCMC chain of 30,0000 iterations, was implemented. The initial 50,000 iterations were treated as training for the model and were discarded for the final analysis. Model output was further investigated through R package wiqid (Meredith 2019).

Posterior probabilities of the MCMC samples were summarized through the mean and 95% highest density interval (HDI). The strength and consistency of the generated estimates was assessed through the overlapping extent of the posterior and prior distributions reported within the wiqid package (Meredith 2019) by the function postPriorOverlap.

Chapter 3

RESULTS

3.1 Photographic Capture Results

Out of the 30 installed camera traps, 5 were removed from the analysis due to malfunctioning. The remaining 25 traps resulted in 1125 trap days capturing a total of 15,266 pictures. Within this time period, lynx was detected 16 times at 6 different sites (Figure 3.1). None of the photo-captured lynx was identifiable as a distinct individual due to unclear fur-pattern (Figure 3.2). The success rate for lynx capture was 0.5%.

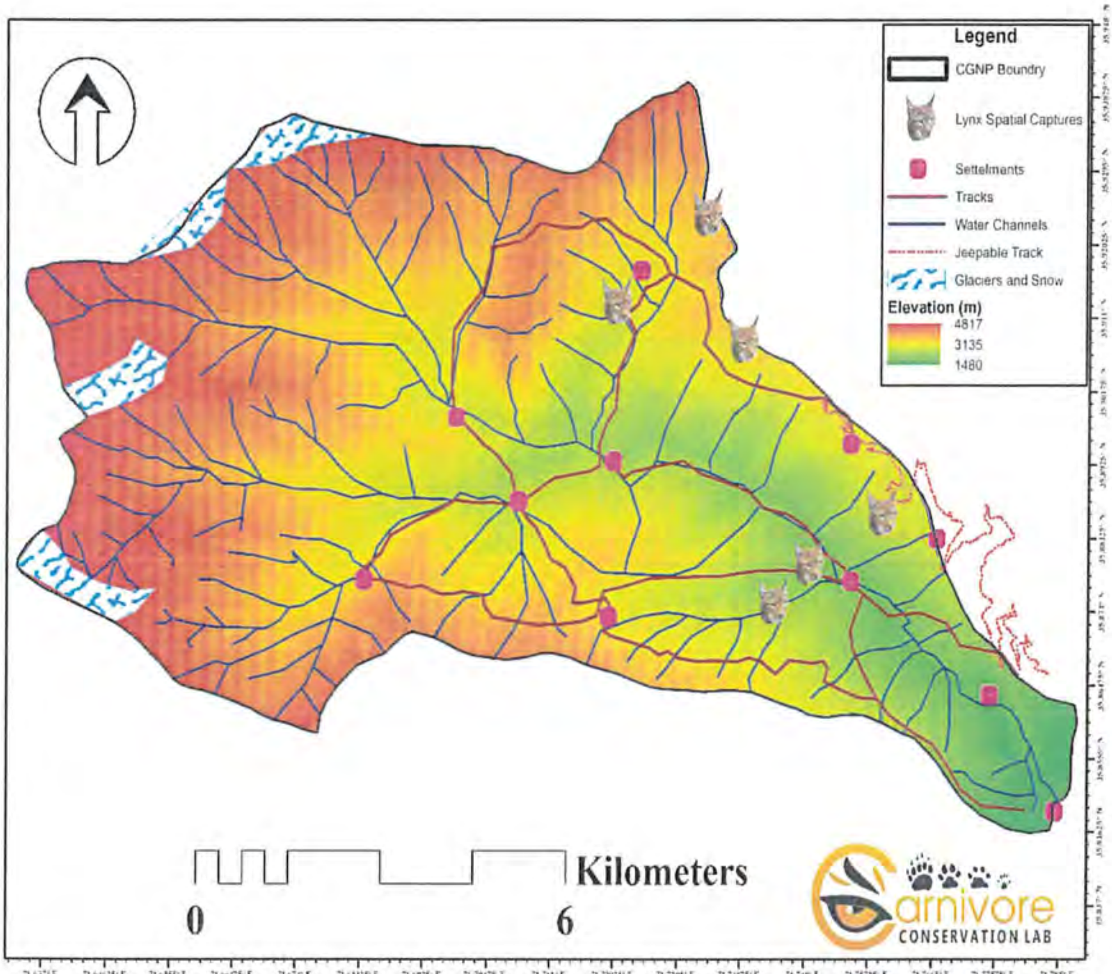

Figure 3.1 Spatial location of lynx photo-captures in Chitral Gol National Park in 2019

Figure 3.2 Lynx photo-captures in Chitral Gol National Park in 2019

3.2 Abundance Estimation

Different modeling approaches were utilized to estimate abundance of the Himalayan Lynx from camera trap data in CGNP. The outcomes of these models are described below.

3.2.1 N-Mixture Models for Repeated Counts

Three variations of N-Mixture approach for repeated counts were tested against the available data. There was no significant difference in abundance estimation. The simple Poisson approach reported 7.88 individuals for the study sites. The estimates reported from Zero-inflated-Poisson and Negative-Binomial were 7.98 and 8.11 individuals, respectively. The best model for this analysis, based on model AIC comparison, was simple Poisson (Table 3.1).

Table 3.1 Abundance estimates for N-Mixture models without covariates

Model	AIC	AICwt	Abundance Estimate
Poisson	151.99	0.58	7.88
ZIP	153.98	0.21	7.98
NB	153.99	0.21	8.11

(AIC=Akaike information criterion, AICwt= Akaike information criterion weightage)

Effects of three site-specific covariates (NDVI, ruggedness, and elevation) were also tested on abundance estimation. Again, based on the AIC comparison, the simple Poisson (without any covariates) stood the best suitable model. Detection probability (p) for the model was 0.0451 ± 0.0161. For confidence limit of 0.975 the confidence interval for model likelihood was 0.12-0.77. The model performance was assessed by calculating mean square error (MSE) which was ± 0.0187.

Table 3.2 presents the results obtained from the Poisson evaluation against different covariates modeled over the detection probability as individually and in various combinations.

Table 3.2 Effect of covariates on detection in N-mixture models for abundance estimation of lynx in CGNP

Model	AIC	AICwt	Abundance Estimate
p(.) lam(.)	151.99	0.295	7.88
lam(.)p(elev)	152.68	0.209	7.83
lam(.)p(rugg)	153.81	0.118	8.37
lam(.)p(ndvi)	153.94	0.111	7.94
lam(.)p(elev+ ndvi)	154.38	0.089	7.89
lam(.)p(elev+ rugg)	154.39	0.089	8.25
lam(.)p(ndvi+ rugg)	155.75	0.045	8.48
lam(.)p(ndvi+ rugg+ elev)	155.79	0.044	8.42

(p=detection, lam=mean estimates, elev=Elevation, rugg=Ruggedness, ndvi= Normalized Difference Vegetation Index)

3.2.2 Random Encounter Modelling (REM)

REM estimated an abundance of 2.69 individuals with a density of $3.47/100km^2$.

3.2.3 Space and Time Modelling

The space and time modelling use three different approaches for abundance estimation i.e., Time-To-Event (TTE), Space-To-Event (STE) and Instantaneous-Sampling-Estimator (ISE). The abundance estimate for TTE was 5.74 individuals with a lower limit of 3.17 and an upper limit of 10.39 and a standard error of 1.78. For STE the estimated abundance was 12.52 with lower limit 7.04 and upper limit of 22.28 and a standard error of 3.76. ISE estimated an abundance of 30.5 with lower limit of 11.9

and upper limit of 78 and a standard error of 15.5. All the estimates were obtained with confidence limit of 0.975. Table 3.3 summarizes the output for the three models of space and time analysis

Table 3.3 Abundance estimates of lynx in CGNP obtained through Space and Time Modelling

Model	N	D (/100km^2)	SE	97.5% LCI	97.5% UCI
TTE	5.74	7.4	1.78	3.17	10.39
STE	12.52	15.8	3.76	7.04	22.28
ISE	30.50	39.35	15.50	11.90	78.00

(N=Number of estimated individuals, D=Density, SE=Standard Error, LCI=Lower Credible interval, UCI=Upper Credible Interval)

3.2.4 Spatial Count (SC) Model

The mean estimated abundance of lynx for SC model was 6.19 individuals with 1 to 15 individuals within the 95% highest density interval (HDI) and a standard deviation of 4.68 (Figure 3.3). Mean density for SC model was 8 lynx/100 km^2.

The 98% overlap for the post-prior efficacy and strength indicate high consistency of the MCMC sampling (Figure 3.4).

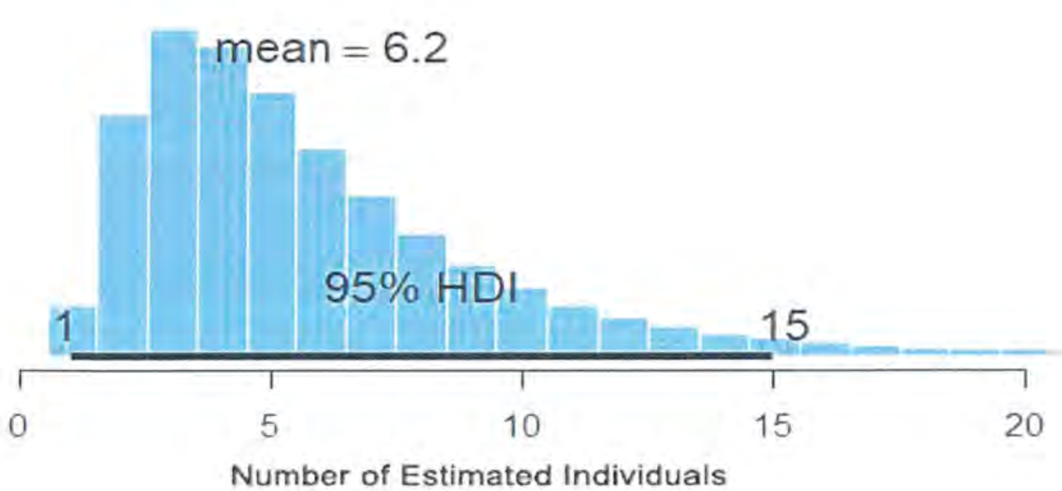

Figure 3.3 Estimated abundance of lynx in CGNP through SC model at 95% highest density interval

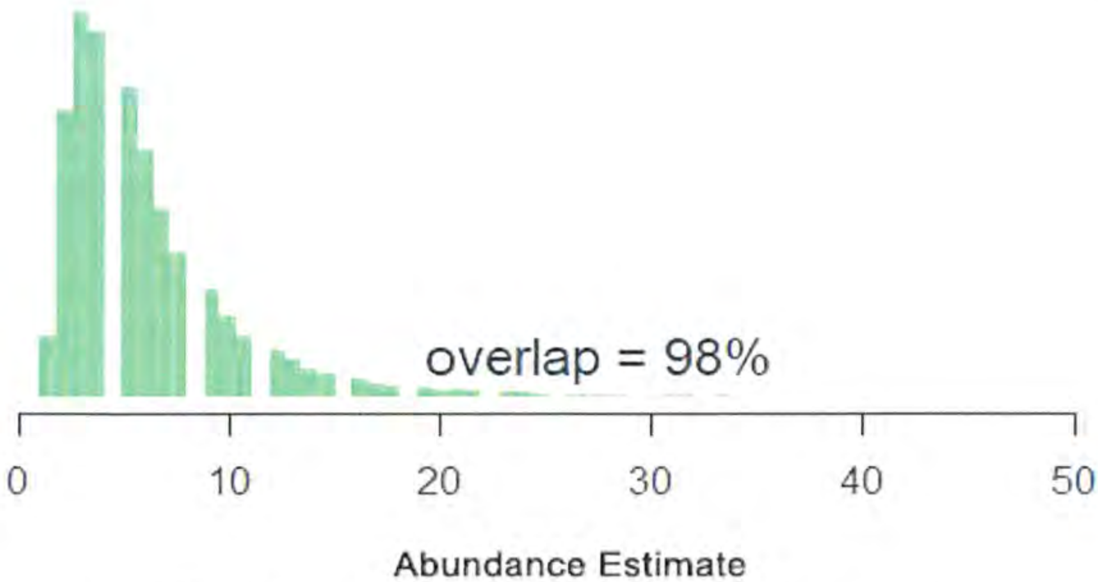

Figure 3.4 Post-prior overlap evaluation of the SC model estimates

3.3 Model Comparison

Highest estimates were reported for ISE with 30.5 individuals followed by STE with 12.5. Estimates for REM were lowest with 2.69 individuals. Estimates for N-Mixture, TTE and SC slightly differed from each other (Figure 3.5).

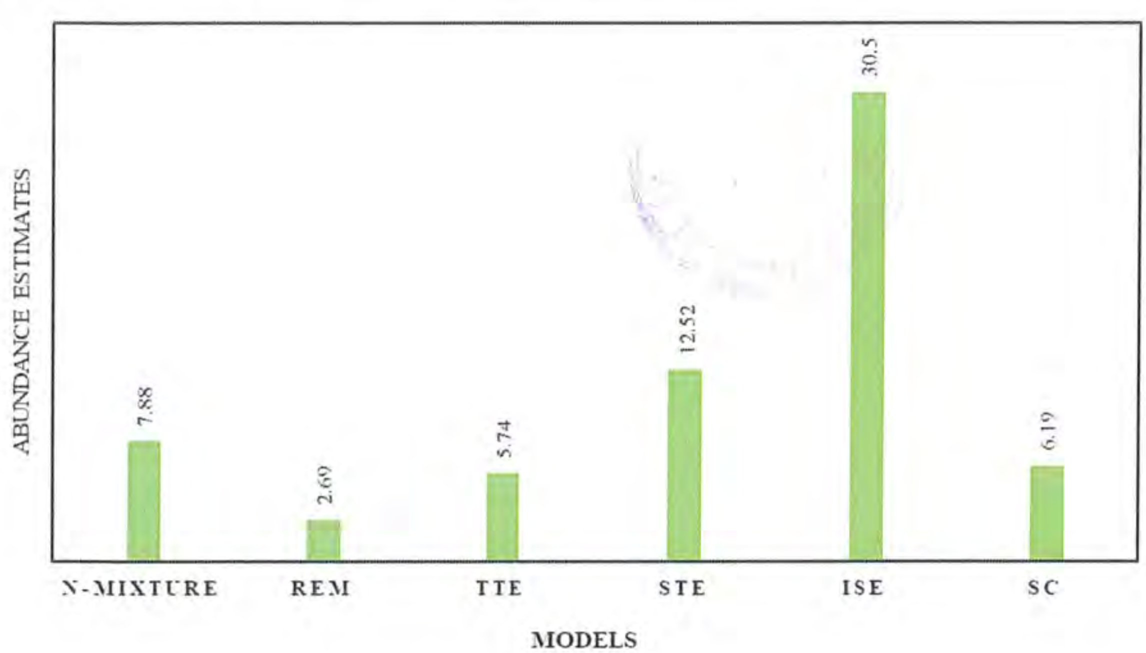

Figure 3.5 Comparative estimates for different unmarked approaches

Chapter 4

DISCUSSION

Robust abundance estimation of wild populations requires good quantitative and qualitative data. Usually limited financial resources and field constrains produce difficulties to obtain this data. Additional hindrance is faced when studying carnivores due to their elusive and nocturnal behavior, and low densities spread over large and remote areas. This means that accurate estimates cannot be guaranteed despite the standardized and consistent protocols.

Current camera trapping study photographed lynx at 6 sites with 16 independent detection events, yielding capture rate of 0.005. The relatively low capture rate is indicative of a small population, however it may also be due to the decreased movement of the nursing females (Nilsen et al. 2012), as this study was conducted just after the current breeding season. None of the captured individuals was uniquely identifiable.

No empirical study based on sound statistical framework has been implemented in Pakistan to assess the lynx population. This study was an attempt to address this question with an emerging unmarked modelling approach, to estimate lynx abundance from camera trap data.

Abundance estimates were highest for Instantaneous Sampling Estimator (ISE) with 30.5 individuals followed by Space-To-Event (STE) with 12.5. Lowest estimates were reported for Random Encounter Modelling (REM) with 2.69 individuals. Estimates for N-Mixture, Time-To-Event (TTE) and Spatial Count (SC) slightly differed from each other.

The lower value of mean square error (MSE) (± 0.0187) for Poisson N-Mixture indicates a best model fit (Arnab 2017). The non-significant change for abundance estimates of models incorporating site specific covariates could be because of the little difference among site covariate values over the study area.

Although REM is a promising approach, there may be some type of bias in the estimates due to wrong measurement of camera detection zone and animal movement rate (Cusack et al. 2015). Also, the model assumption of random camera placement with respect to animal movement, is difficult to meet in case of large territorial carnivores (Foster & Harmsen 2012; Cusack et al. 2015).

The space and time modelling might be a good approach yet need rigorous evaluation with real field data. For the present study, the best space and time approach was TTE model with a smaller standard error of 1.78 as compared to the other two models i.e. STE and ISE. All these approaches were very sensitive to sampling length duration. For a unit increase in sampling length the estimates almost doubled.

The high standard deviation (±4.68) of the spatial count model could be the result of low detection of the species. The large range of 1-15 individuals within the 95% highest density interval (HDI) was due to the over-dispersed data (Evans & Rittenhouse 2018). They could estimate results comparable to the SECR results as suggested by a study on American black bears (Evans & Rittenhouse 2018). To be more useful and reliable they need an assessment through estimation for well-studied species (Burton et al. 2015).

Best estimates reported for the current study within more agreement to the previously reported estimates for the species, were by REM. This might be due to the model requirement for the auxiliary data on the species movement rate (Moeller et al. 2018).

No previous study based on established statistical framework for abundance estimation is available for the area to evaluate the findings of the current study. A camera trap study from the area conducted by Din et al. (2015) identified 6 unique lynx individuals. A study.by Blanc et al. (2013) using SECR on camera trap data from French Jura Mountains estimated abundance at 12.04 individuals. For the same study area using N-Mixture models Blanc et al. (2014) estimated lynx abundance at 9.96 ±1.18. A study from Saihanwula National Nature Reserve of Inner Mongolia, China reported lynx density at 27.14/100 km^2, using SECR on camera trap data (Tang et al. 2019). Zimmermann et al. (2013) using CMR on camera trap data from the northwestern Swiss Alps estimated lynx density at 1.20-3.6/100 km^2. Pesenti and Zimmermann (2013) reported a density of 1.9–2.1/100 km^2 in the northwestern Swiss Alps using SECR on telemetry and camera trap data. A study from Germany utilizing both telemetry and camera trap data reported density at 0.4/100 km^2 and 0.9/100 km^2, respectively (Weingarth et al. 2012). Avgan et al. (2014) from Turkey reported lynx density at 4.20/100 km^2 using SECR for camera trap data. Kubala et al. (2019) using SECR on camera trap data for two study sites from Slovak Carpathians independently reported density at 0.58 ± SD 0.13/100 km^2 and 0.81 ± SD 0.29/100 km^2. A camera trap study using SECR from French Jura and Vosges mountains estimated abundance between 5-29 lynx and density between 0.24-0.91/100 km^2 (Gimenez et al. 2019).

Estimate of the N-Mixture model for the current study cannot be compared to any of the previous studies as this was reported for the surveyed sites only not for the total study area. The estimate for density by REM at 3.47 lynx/100km^2 is comparable with many of the previous studies (Pesenti & Zimmermann 2013; Zimmermann et al. 2013; Avgan et al. 2014). Estimates for TTE, SC and STE are slightly higher for the current study. ISE estimated the highest density ever reported for the species at 30.5 lynx /100km^2 surpassing the previously reported highest density at 27.14 lynx/100 km^2 by Tang et al. (2019).

Conclusion and Recommendations

As suggested by the findings of the current study unmarked modelling could be powerful alternatives to the CMR approach. Yet the different approaches of unmarked have some shortcomings. The N-Mixture models can reliably estimate site specific abundance for the surveyed sites but cannot be used to reliably estimate abundance at the unsurveyed sites. Use of REM is limited by the requirement of auxiliary data on animal movement rate. Also, determining the camera detection zone radius and detection angle, need great care as the estimates are very sensitive to these parameters. All three approaches of Space and Time modelling are very sensitive to length of sampling occasions. Specifying a wrong length may lead to either overestimation or underestimation. Estimates of SC models are vulnerable to animal space use specification. Both the Space and Time and SC require prolonged execution time and higher computational resources and will not perform on low end machines.

To increase their reliability unmarked approaches, need an evaluation against SECR, for estimates of marked population. Also, the estimates can be improved when applied for combined data from different survey tools i.e. camera tarps, genetic sampling and radio telemetry.

This study is the first of its kind for the study area, addressing an important parameter for conservation, monitoring and management of wildlife populations. Robust population estimates are integral to conservation efforts. Estimates from the current study can help the authorities to define conservation strategies for lynx population.

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Annexure-I

CAMERA TRAP STATION SHEET

Set by:

STATION ID	

e.g. MISGAR-WS1-CAM1

WATERSHED	

e.g. MISGAR-WS1

Decimal degrees

N	☐☐.☐☐☐☐☐☐
E	☐☐.☐☐☐☐☐☐
ELEVATION	meters

CAMERA ID	

LURE TYPE	☐skunk + fish oil	☐castor + fish oil	☐fish oil	☐none	
HABITAT	☐scrub	☐forest	☐pasture	☐barren	☐agric.

(in immediate surroundings)

TERRAIN	☐ridge	☐cliff base	☐draw	☐valley	☐saddle	☐plateau
SUBSTRATE	☐sand	☐soil	☐rock/ gravel	☐snow	☐vegetation	

Station potential ☐good ☐medium ☐poor

Sign in buffer area ⇨

STATION VISIT ⇩

	DATE	TIME	SIGN AT STATION	SD CARD	Camera Operational	NR NEW PHOTOS
SETUP						
RE-BAITING					☐ YES ☐ NO	
TAKE-DOWN					☐ YES ☐ NO	

Comments: (use backside, if needed)

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