URBAN CARNIVORES IN ASIA: SURVIVAL, OCCUPANCY AND CONSERVATION

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ABSTRACT

Increased urbanization has led to significant changes in land use types and landscapes, resulting in substantial negative impacts on biodiversity. This thesis aims to analyze which urban environmental and biological factors affect the survival and occupancy of urban carnivores in Asia and how to develop corresponding habitat connectivity programs to restore carnivore populations.

Chapter 2 investigated the potential for carnivore survival and presence in different Asian cities through a literature search. The effects of different urban environmental variables and species' physiological traits on urban carnivore survival were assessed. Chapter 3 compares leopard cat occupancy in two cities, Shenzhen and Hong Kong, by camera trapping, and patterns of occupancy variation were assessed according to different urban environmental factors. Chapter 4 investigated the distribution range and temporal activity patterns of leopard cats in Shenzhen using kernel density estimation. The spatiotemporal distribution pattern overlap of humans, domestic cats, and dogs, as well as the interactions and potential ecological niche competition between leopard cats and the different species, are explored. In Chapter 5, potential leopard cat ecological corridors in Shenzhen were simulated using the minimum cumulative resistance (MCR) model (with cost analysis), considering the distribution density of leopard cats and environmental factors such as topography and human and social factors.

Chapter 2 found that the percentage of urban built-up area was the most critical factor

affecting urban carnivore survival, with a significant coefficient of -0.5 in my regression model ($p<0.05$). Regarding physiological characteristics, the urban survival rate of non-diurnal carnivores was 28.4% higher than that of diurnal carnivores. The urban survival rate of solitary carnivores was 45.1% lower than that of group carnivores.

Chapter 3 found that the occupancy rate of leopard cats in Shenzhen was 18.8% lower than in Hong Kong ($p < 0.01$). The number of leopard cats decreased with increasing distance from the road, and in Shenzhen, the occupancy probability of leopard cats was higher in agricultural areas (0.75) than in urban areas (0.35).

Chapter 4 found that the spatial overlap between leopard cats and humans, domestic cats, domestic dogs, and wild pigs was > 55%. Regarding temporal overlap, leopard cats had the highest overlap with domestic cats, with peak activity times at 06:00 and 23:00, 3-4 hours later than the peak activity times of domestic cats (02:00 and 20:00).

In Chapter 5, combined with the estimated spatial extent of core leopard cat density, this study proposes 118 potential ecological corridors in Shenzhen to aid leopard cat migration. The 23 most important major ecological corridors were identified. The average width of the best corridors was 727 m, with a total corridor area of 97.7 km^2 and an estimated total cost of RMB 55.74 billion.

This study increases our understanding of urban carnivores in East Asia, confirms the importance of natural habitats for urban carnivores, and provides evidence that will be useful for prioritizing carnivore species conservation, controlling alien species and human disturbance, and restoring carnivore habitat connectivity.

Based on the results of this thesis, I suggest conserving remaining native habitats, restricting the expansion of urban built-up areas (for example, by laws of ”ecological red lines”), and giving priority to the conservation of heavy, nocturnal, and solitary carnivores. Increased surveillance and stern action against poaching are also needed. In order to avoid

endangering wild predators, my results also suggest reducing the amount of stray cats and dogs in urban habitat regions. Additionally, recreational users' access to carnivore habitats has to be controlled in order to lessen the negative effects of growing human-wildlife distributional overlap. Finally, it is important to identify the hotspots where wildlife is most active in cities, as well as the key connecting corridors that these animals use through the urban matrix. Fences that point animals in the direction of ecological corridors should also be created. When building the corridor habitat, native plants should be useful, and managers should also pay attention to the quality of the vegetation.

DEDICATION

Dedicated to my parents

ACKNOWLEDGMENTS

I would first like to express my gratitude and reverence to my advisors Luke Gibson, Tom Matthews, and Jon Sadler, whose advice and encouragement have helped me gain a deeper understanding of conservation biology. I am also very grateful to all my friends and classmates who helped me during the preparation of this thesis. In addition, I would like to thank my parents, Ying He and Feihu Song, for their unfailing love and unwavering support.

You have to let it all go, Neo

Fear, doubt, and disbelief

Free your mind

"The Matrix"

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Acronyms

AIC Akaike information criterion. 23

CI Confidence Interval. 23

DEM Digital Elevation Model. 47

GLMs Generalized Linear Models. 22

MCR Minimum Cumulative Resistance. 12

NDVI Normalized Difference Vegetation Index. 47

OSM Open Street Map. 47

SDM Species Distribution Model. 39

Chapter One

Introduction

1.1 Introduction: the impact of urbanization on biodiversity in cities

Currently, over 50% of the world's population lives in urban areas, which is expected to increase to 68% by 2050 (UN-Habitat, 2022). Increased urbanization (Behnisch et al., 2022); (X. Li et al., 2021) has led to significant changes in urban land-use types and landscapes (Dadashpoor et al., 2019), a process that is taking place in regions with high biodiversity (Seto et al., 2012), such as several large Asian megacities (Fei and Zhao, 2019). The resultant modifications to and loss of natural habitats in cities have significantly negatively impacted biodiversity (McKinney, 2008; Knapp et al., 2021), especially compared to patterns found in natural habitats (Aronson et al., 2014).

The process of urbanization is characterized by increased disturbance (Grimm et al., 2017), increased landcover change and turnover of landcover (Dadashpoor et al., 2019), habitat fragmentation (Dickman, 1987), reduced connectivity (LaPoint et al., 2015) and modified biogeochemical processes (Kaye et al., 2005), and local climates (Kumar, 2021).

This wholescale modification of once pristine natural environments affects biodiversity at both inter- and intraspecific levels (Knapp et al., 2021). Overall, and across numerous taxonomic groups, the pattern is one of reductions in species richness (McKinney, 2008; Aronson et al., 2014) and increased biotic homogenization (McKinney, 2006), as well as both functional responses (Williams et al., 2009; La Sorte et al., 2018) and the loss of phylogenetic and genetic diversity (Knapp et al., 2021). Urbanization thus filters regional biotas, yet with the addition of nonnative species (Gaertner et al., 2017) in local species pools, linked to the creation of anthropogenic habitat, it can also create novel species assemblages (Merckx and Van Dyck, 2019).

The recent increase in global, cross-taxa comparisons of the impact of urbanization (Aronson et al., 2014) point to geographical and taxonomic biases in the comparisons, with the bulk of the studies focusing on plants, insects and birds in temperate regions. Research on other groups (e.g. non-bird vertebrates) and tropical and Asiatic regions lacks (X. Li et al., 2021). This is especially true of mammals and mammal carnivores in urbanized regions, although recent reviews indicate both behavioural (Ritzel and Gallo, 2020) and physiological (Hantak et al., 2021) plasticity in their responses to urbanization.

This chapter seeks to examine the changes and influencing factors affecting urban biodiversity both before and after urbanization. It uses facts and figures to show that habitat loss, fragmentation, and increased species competition caused by urbanization have negative impacts on species survival. Thus, it provides evidence for governments and NGOs to further slow down or prohibit urban expansion. It shows that urban top predators are not always critically endangered species and may, in fact, be widely distributed in urban nature reserves. The reason for studying urban biodiversity, then, is that urban biodiversity indicates the integrity of urban ecosystems and the integrity of urban food webs and symbolizes the normal functioning of urban ecosystems. Cities rely heavily on the resilience of urban ecosystems, especially in the face of natural disasters and the need for clean air and water.

1.2 Urbanization and carnivores

Large carnivores require large habitat areas, and habitat loss can lead to their local extinction. However, many studies have also demonstrated that habitat fragmentation per se, leaving aside habitat loss, affects biodiversity in numerous ways (Crooks K.R., 2002). Fragmentation leads to an increase in the length of habitat edges and a decrease in the size of habitat core area (Ewers and Didham, 2007). Carnivores sensitive to urbanization and human activity will reduce in abundance due to edge effects (Łopucki et al., 2019). The presence of roads in many anthropogenically fragmented landscapes results in roadkill being a leading cause of carnivore mortality in cities (Bateman and Fleming, 2012). Isolated habitat patches prevent genetic exchange between carnivores, reducing the probability of reproduction and increasing the likelihood of local extinction (Lino et al., 2019). The reduction in gene exchange due to isolation has a number of additional severe consequences. For example, when an originally intact carnivore population is divided into multiple isolated meta-populations, the genetic diversity of each subpopulation is reduced, potentially triggering increased "genetic drift", which reduces the adaptive capacity of the population to respond to disturbance (Santangelo et al., 2018). In addition, the continuous inbreeding within a small habitat makes the offspring highly susceptible to genetic diseases (Saccheri et al., 1998; Bijlsma and Loeschcke, 2012). Habitat loss and fragmentation due to urbanization reduces and concentrates the distributions of numerous carnivore species, leading to the temporal and spatial overlap of wildlife, and thus increased competition between and within species (Zanni et al., 2021). For example, in Mumbai, India, leopards (*Panthera pardus*) and hyenas (*Hyaena hyaena*) have been observed close to human settlements, adapting to urban environments by using food sources such as domestic animals and garbage, leading to increased overlap between the two species, both spatially and temporally (Athreya et al., 2013).

Several carnivore species are known to adapt to urbanization by changing their behavioural patterns (Ritzel and Gallo, 2020). For example, some diurnal animals increase their nighttime activity to avoid frequent human disturbance during the day. Some animals change their spatial range and try to avoid the edges of habitat patches. In contrast, others prefer the transition zone between habitat patches and urban areas, and home ranges are denser in that area. The increase in the number of small mammals, such as rodents, in the natural habitat-urban transition zone increases the attraction of these sites for carnivores. Many animals have adapted to a life that relies more on human waste as a food source and have even changed their diet by accepting direct human feeding. A study in northern Utah demonstrates that coyotes (*Canis latrans*) and their prey have adjusted their activity times, increased the frequency of nocturnal activity and increased the temporal overlap in species' activities (Zanni et al., 2021). Many species depend on human feeding or litter, and species that would originally occupy different ecological niches and thus would be unlikely to compete in the wild are in the same urban-habitat transition zone, competing for the same food sources (Theodorou, 2022).

Domestic cats, an exotic species that would not naturally be present in urban habitats, are a lethal threat to many native urban species, including birds (on which they prey) and small mammals such as wild leopard cats, which occupy a similar ecological niche (Doherty et al., 2017; Zapata-Ríos and Branch, 2016). Similar competitive relationships exist for other species. Canids, such as coyotes, foxes, and wolves, can adapt to urban life due to their opportunistic behaviour and diet. They can easily adapt to the food supply in urban areas, including garbage and pet food. In addition, they can quickly learn to avoid humans and hide in small spaces, making them difficult to detect (Leflore et al., 2019; Gil-Fernández et al., 2020; Breck et al., 2019). Urban red foxes in Bristol, UK, have been observed successfully breeding and raising their young in urban parks and gardens (Baker and Harris, 2004).

Due to their solitary nature and ability to hunt small prey, Felidae and Viverridae are generally well-adapted to urban life. Bobcats have been observed in urban areas, such as the suburbs of Washington, D.C., where they are known to prey on rabbits and other small mammals (Poessel et al., 2014). The small Indian civet (*Viverricula indica*) has been observed in urban areas of India, where it feeds on garbage and fruit trees (Balakrishnan and Sreedevi, 2007). Due to their opportunistic feeding habits and ability to live in small spaces, Mustelidae and Herpestidae can also adapt to urban life (Wright et al., 2022). They are often found in urban areas with waterways or parks with suitable habitats for hunting and hiding. Weasels (*Mustela nivalis*) have been observed in urban environments in Europe, where they prey on small mammals such as mice and rats (Zalewski, 2001).

1.2.1 Asian urban carnivores

The rapid development in many Asian cities has been detrimental for many carnivore species. For example, tigers were once widespread in China, but by the 1950s, they had been driven extinct in southern China. The leading cause of extinction was the hunting and poaching of tigers for their skins, bones, and other body parts. The Chinese government banned tiger hunting in 1979, but it was too late to save the tiger population in southern China. The South China tiger was last seen in the wild in the 1990s. This subspecies is considered extinct in the wild, with only a few individuals being kept in captivity (Huang et al., 2021).

However, there are still some carnivores in many cities in Asia that are adapted, to some degree, to urbanized life. For example, fishing cats (*Prionailurus viverrinus*) are found in Sri Lanka (Santiapillai et al., 2003). Their home range is about 2-5 km, and they prey on fish, frogs, small mammals, and even domestic animals such as poultry and small pets (Santiapillai et al., 2003). However, fishing cats are still considered endangered, facing threats such as habitat loss and poaching (Ganguly and Adhya, 2022). To take another example, the presence of leopards (*Panthera pardus*) was found in Mumbai. Their home range is approximately 30-40 km (Athreya et al., 2013), and they have been facing major threats due to habitat fragmentation, poaching, and human-leopard conflict, with several cases of leopard poaching occurring in Mumbai recently (Athreya et al., 2013). The increasing number of sightings may indicate that the leopard population is on the rise in urban areas of Mumbai. However, the exact number of leopards in Mumbai's urban areas is still unknown. As a final example, raccoon dogs (*Nyctereutes procyonoides*) are an invasive species in Tokyo (Saito et al., 2016). They have a home range of 1-3 km and prey on small mammals, such as rabbits and rodents (Saito et al., 2016). Raccoon dogs are not considered an endangered species and are not affected by poaching. In conclusion, records of carnivorous species in urban areas of Asia indicate that these species face various threats, such as habitat loss, poaching, and human-wildlife conflict. However, while the aforementioned examples are informative, it is clear that there are large data gaps in terms of what is known about the presence and ecology of carnivores in Asian cities. It is thus crucial that more is done to monitor populations of these species with the aim of better-informing conservation measures.

Carnivores have also been recorded in many Chinese cities, indicating that, despite the rapid development in recent years, cities have retained space for wildlife to survive, at least for now. According to a study by the Shanghai Wildlife Conservation Society, the most common carnivores in the city are civets, leopard cats, and Chinese skunks: the study found approximately 1,500 civets, 1,200 leopard cats, and 800 Chinese skunks in the city(Zhang et al., 2019). The home range of the civets was approximately 1.2 square kilometres, while the home range of the leopard cats was approximately 3.8 square kilometres. The home range of the Chinese weasel badger is smaller, about 0.5 km^2. Masked palm civets feed mainly on insects, fruits, and small mammals. While leopard cats feed on rodents, birds, and reptiles, Chinese ferret badgers feed on insects, small mammals, and fruits. The association's report shows that poaching is rampant in Shanghai, and many animals are killed and sold on the black market (Zhang et al., 2019).

Carnivores in Asian cities, including Chinese cities, will be under significant threat in the future. First, these cities are still undergoing rapid development, with human population numbers and built-up areas still expanding (UNESCAP, 2013). Second, governments may put economic development ahead of ecological conservation while the economy is still developing. There is no appropriate policy and reasonable funding to protect natural habitats in many Asian cities, nor the wildlife living in them. In addition, conservation organizations and scientific institutions do not have enough funding to support their research on urban carnivores, leading to gaps in research and background surveys. Finally, there is a lack of conservation awareness among the public, who often do not have a correct concept of wildlife protection.

1.2.2 Leopard cat (*Prionailurus bengalensis*)

One of the few remaining Asian urban carnivores, the leopard cat, is known to occur in the centre of a rapidly developing megacity, Shenzhen. Why have they adapted to city life without becoming extinct? Are there other carnivores in the city, and what kind of city is more suitable for carnivores to survive? These are all questions the present thesis will address.

Leopard cats exhibit a broad distribution and are found in several regions in southern Asia. Its range encompasses eastern Afghanistan and the northern and coastal areas of India, Myanmar, Sumatra, and eastern China. According to the IUCN Red List, the leopard cat is currently classified as "Least Concern" and exhibits a population that remains steady, although this is across its whole range and not specifically in relation to urban areas. Leopard cats demonstrate exceptional hunting abilities and are known to prey on tiny vertebrate pests inside rural and agricultural regions (Leopard Cat Foundation, 2009; Prof. Paul's Guide to Mammals, 2011; Chan, 2010; Sanderson et al., 2008.).

Leopard cats inhabit various ecosystems, including tropical and temperate forests, coniferous forests, shrubland habitats, and grasslands (Grassman,2006). It is found in these habitats at elevations between 0 and 4400 metres. The species demonstrates high adaptability to diverse vegetation and climate conditions but is usually associated with dense vegetation. The habitat of the leopard cat also typically encompasses water sources.

The species can deal with small amounts of human disturbance, as seen by the stable populations in secondary forests and disturbed environments. Additionally, the species is frequently observed in close proximity to agricultural land and rural towns. The species exhibits proficient swimming abilities, which potentially accounts for its presence on various islands (Leopard Cat Foundation, 2009; Prof. Paul's Guide to Mammals, 2011; Fernandez Degia, 2011; Mukherjee et al., 2010; Watanabe, 2009).

In China, leopard cats were widely hunted for their furs, which are used to make clothing and accessories. The demand for their coats has led to a decline in their population, estimated to have declined by about 50% in the last 20 years. The density of leopard cats in their natural habitat has also been reduced by as much as 80% in some areas (Mohd-Azlan, 2022). In 2015, China upgraded the protection status of leopard cats from second to first-class national protected animals. Another essential strategy for their protection is establishing protected areas for the leopard cat and its habitat. The year 2015 saw the establishment of the National Leopard Cat Nature Reserve in Fujian Province, the first reserve dedicated to this species in China. The reserve covers 12,000 hectares and includes critical leopard cat habitat.

According to a 2022 survey (Yang,2022), there are an estimated 5,000 to 7,000 leopard cats in China, and their population density has increased in some areas due to conservation efforts. However, they are still at substantial risk from human-wildlife conflict, particularly in urban areas (J. Wu et al., 2020; G. Li et al., 2022). For example, many zoonotic viruses increase wildlife mortality, and wildlife species are hunted to avoid city-dwellers' exposure to these viruses (Lo, 2021). Pollution of water sources in cities, the increase in solid waste such as plastic, and increases in construction and industrialization in urban areas, amongst other things, are also a great threat to leopard cats (J. Wu et al., 2020). Continued habitat destruction, fragmentation and degradation, mortality from traffic collisions, and the impacts of introduced species (e.g. domestic cats) are all anthropogenic activities that are also known to negatively affect leopard cats in China (J. Wu et al., 2020).

However, despite leopard cats being the focus of numerous governmental conservation initiatives, people know very little of their ecology within urban environments, partly due to the lack of fine-scale field data relating to their presence/absence within habitat patches within cities.

1.3 Urbanization in China

Urbanization in China has a long and complex history, with the process accelerating significantly after 1978 when the country began its economic reforms. This was due to several factors, including the opening-up of the Chinese economy, the shift towards a market-oriented economy, and the growth of the manufacturing and service sectors (Lu, 2014; Gu, 2010). There have been three main stages of urbanization in China:

1) During the early period (1978-1990), urbanization was slow, focusing on developing basic infrastructure in urban areas. Urbanization was concentrated in the coastal areas and major cities, such as Beijing and Shanghai. During this period, the built-up area in cities was around 5%, and the urban population was around 17% of the total population (Gu, 2017). 2) The developmental period (1990-2000) saw a rapid increase in urbanization,

with the construction of new cities and the expansion of existing ones. New cities, such as Chengdu and Chongqing, were constructed in inland areas. In this period, the built-up area increased to around 10%, and the urban population increased to around 30% (Gu, 2017). 3) The peak period (2000-present) has seen a continued increase in urbanization, focusing on the development of high-tech industries and the growth of megacities. It has seen the growth of megacities, such as Guangzhou and Shenzhen. In this period, the built-up area has increased to around 20%, and the urban population has increased to around 60% (Gu, 2017). 4) China's urbanization is expected to continue in the coming years, focusing on sustainable development and integrating urban and rural areas. The Chinese government has set a target of 70% urbanization by 2035 (Gu, 2017).

Urbanization has led to significant changes in urban land-use types and landscapes. For example, in natural green areas, agricultural land has decreased, and the size of construction land and artificial green regions has increased (Winkler, 2021). Due to the early rise of the logging industry in many cities, the existing forests are mainly planted forests and secondary forests (Pataki, 2021). Natural habitats for wildlife have been lost and fragmented due to the destruction of natural green spaces. Human activities such as hiking and poaching are frequent in various habitat patches, with often substantial impacts on wildlife. There is even illegal use of natural green areas for construction, further exacerbating habitat loss and fragmentation (Chen, 2020).

1.4 Aims and research questions

The above review of the literature illustrates that, while there are a number of city and species-specific case studies, our knowledge of carnivore species in Asian cities is fragmentary, and we have little fine-scale ecological data for individual urban carnivore species of conservation concern, such as the leopard cat. Combining secondary data with primary camera trapping data, this thesis aims to investigate the survival of carnivores in Asian cities:

First, the objective of Chapter 2 is to investigate the urban environmental and species-level variables that have the greatest impact on the survival of carnivores in cities. The objective of Chapter 3 is to compare the distribution of leopard cats in the urban centre habitat of Shenzhen and Hong Kong's county parks to determine which city has a broader distribution of leopard cats. The most important factors contributing to the different distribution status of leopard cats in Shenzhen and Hong Kong will be identified. The objectives of Chapter 4 are to investigate whether other wildlife species overlap with the leopard cat in urban habitat patches in Shenzhen in space and time, to calculate the degree of overlap, and to compare which wildlife species overlap with the leopard cat the most, and thus have the greatest impact on the survival of the leopard cat. Chapter 5 is focused on the design of ecological habitat corridors to increase connectivity for leopard cats. It has five objectives: 1) to select suitable ecological core areas in Shenzhen; 2) to determine the location and number of potential ecological corridors in Shenzhen; 3) to prioritize the construction of different ecological corridors; 4) to establish a systematic methodology for determining the most effective width of ecological corridors; and 5) to calculate the cost of ecological corridor construction to provide theoretical support for the construction of urban ecological corridors in Shenzhen.

1.5 Thesis Structure and overview

The present chapter (Chapter 1) provides a brief introduction to urbanization, urban carnivores, and the overall study framework of the PhD thesis. Chapter 2 compares the survival rates of carnivores in different cities in East and Southeast Asia and the urban factors and carnivore species characteristics that affect urban survival rates. Chapter 3 involves the collection of field data on leopard cat presence in Shenzhen and uses this to compare the occurrence probability of leopard cats in two megacities, Shenzhen and Hong Kong, and assesses how urban environmental factors affect the occurrence probability in the two cities. Chapter 4 compares the spatial and temporal distribution of leopard cats in Shenzhen with other carnivore species, such as domestic cats, and calculates the overlap rate between species. Chapter 5 uses the fieldwork data collected in earlier chapters to design an ecological corridor network to increase the connectivity between urban habitat patches for leopard cats in Shenzhen using the minimum cumulative resistance (MCR) method. The final chapter, Chapter 6, provides a discussion and contextualization of the findings and provides recommendations for further study. A brief chapter structure is shown below (Figure 1.1).

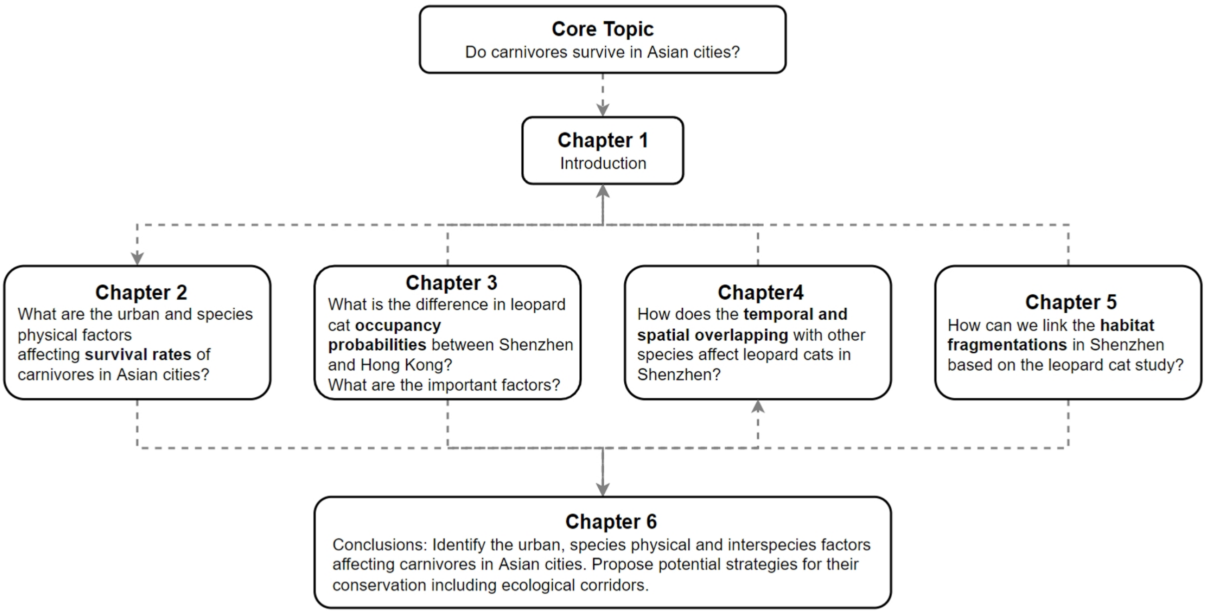


Figure 1.1: Thesis structure. The core scientific issues addressed by the thesis as a whole, as well as the individual chapters, are shown, as are the hierarchical relationships between the chapters.

Chapter Two

A cross-sectional comparison of carnivore survival in Asian cities: Which urbanization factors and species characteristics significantly impact survival?

2.1 Abstract

Urbanization has led to the extinction of many urban predators. Large carnivores have become extinct, with only small and medium-sized carnivores remaining, thus representing keystone species in urban ecosystems. However, urban carnivore research has been concentrated in Western countries in Europe and the United States. Few studies have been conducted on urban carnivores in Asia, especially in East and Southeast Asia, where urbanization is proceeding rapidly.

This study collected reports of carnivore presence in different Asian cities by searching scientific papers, government websites, and published documents, as well as the list of possible carnivores in different Asian cities through the IUCN database. The effects of different urban environmental variables (e.g., population density, GDP per capita, and the ratio of built-up area to urban area, etc.) and physiological characteristics of species (e.g., body weight, diet, lifespan, and activity pattern) on the survival rate (reported carnivore diversity/potential carnivore diversity) of carnivores in the cities was then assessed.

The percentage of built-up urban area was the most important factor influencing the survival of urban carnivores, with a significant coefficient of -0.5 ($p<0.05$) in the regression model. Canidae and skunks were the two groups of carnivores with the highest survival rates in urban areas, with survival rates greater than 70%. Regarding physiological characteristics, the urban survival rate of non-diurnal carnivores was 28.4% higher than that of diurnal carnivores, and the urban survival rate of solitary carnivores was 45.1% lower than that of group carnivores.

This study explores the urban environmental variables that have the most significant impact on the survival of carnivores in cities and provides recommendations for biodiversity conservation in Asia and other rapidly growing cities. The sensitivity of carnivores in cities to urbanization and development is ranked to facilitate more targeted government policies. This study suggests protecting existing habitats, limiting the expansion of urban built-up areas (e.g., ecological red line policies), and prioritizing the protection of heavy, diurnal, and solitary carnivores.

2.2 Introduction

Urbanization and its resulting habitat loss and fragmentation have a significant negative impact on urban biodiversity, especially with regard to carnivores (Riley, 2003; Smith, 2018). The last century has seen the increased movement of people towards urban areas, and thus, the impacts of urbanization on biodiversity have increased (Grinder, 2001). These include the impacts of land use change, as well as more subtle impacts such as those from recreational activities such as hiking and biking in urban country parks (Lewis, 2015). However, as urbanization continues, to aid sustainable development and maintain the stability of urban ecosystem functions, governments must develop and enforce more conservation policies and increase funding to restore urban biodiversity (Ciach, 2019; Fitzgibbon, 2011).

Carnivores are the top predators of ecosystems and are thus essential components of food chains, controlling the abundance of species in lower trophic levels, including many birds and small mammals (Thomas et al., 2018). As such, they represent keystone species. However, most are highly sensitive to human activities and habitat alteration, and their numbers have declined dramatically during urbanization (Makelainen, 2016; McKinney, 2006). A few species, particularly medium to small carnivores, can adapt to urban life, possibly because large carnivores have become extinct in cities. This is called the "Mesopredator Release" theory (Soulé, 1988; Crooks, 1999). It has also been argued that omnivores with a broader ecological niche are best adapted to cities, especially carnivores that can use human waste as food (Gehrt, 2020).

Other studies indicate that animals shift their activity time and increase their nighttime activity to avoid human disturbance (Prange, 2004; Gehrt, 2009). Many studies in North America have found that coyotes (*Canis latrines*), bobcats (*Lynx rugs*), and red foxes (*Vulpes values*) are adapted to urban life (Atwood, 2004; Salek, 2014). In Los Angeles, bobcats and coyotes maintain crepuscular activity patterns, but they become less active in the daytime than in rural populations (Lourraine, 2002). Researchers also found that human food accounted for 14-25% of total coyote food in areas with high human activity, while it only accounted for 0-3% in areas with minimal human activity (Fuller, 2001).

However, as evidenced by the abovementioned studies, urban carnivore research mainly focuses on Europe and North America. A review article also highlighted the lack of studies in Asia and South America (Bateman, 2012). Urbanization in Europe and the United States began in the 1830s and proceeded relatively slowly (e.g., it took the United Kingdom about 150 years to increase its urbanization rate from 20% to 80%, Solene 2008; Stanley, 2011). In contrast, urbanization in Southeast and East Asia accelerated from the 1950s (e.g., China took only 40 years to increase its urbanization rate from 20% to 60%, Revision of World Urbanization Prospects, https://population.un.org/wup/). There are many Asian cities where the human population has risen from a few hundred thousand to tens of millions in decades, resulting in substantial threats to urban biodiversity (Garden, 2010). However, we know very little about the response of Asian carnivores to urbanization and thus cannot develop appropriate conservation strategies.

Here, for the first time, a comparative study of urban carnivores in Asia is presented. Secondary data on the presence of carnivores in different cities were collected and compared with their potential geographical distribution. The objective of this study was to investigate the urban environmental variables that have the greatest impact on the survival of carnivores in cities and the species characterization variables that have the greatest impact on the survival of carnivores in cities.

It was hypothesized that urban vegetation cover, GDP per capita and population density would significantly affect the survival of carnivores in cities. Urban vegetation cover, GDP per capita and survival rate were predicted to be positively correlated. Population density was hypothesised to be negatively correlated with survival rate. In addition, it was hypothesized that body size, activity patterns and dietary habits would significantly affect the probability of survival of carnivores in urban areas. It was predicted that medium-sized, non-diurnal, omnivorous carnivores would have the highest probability of survival in the city. A brief chapter structure is shown below (Figure 2.1).

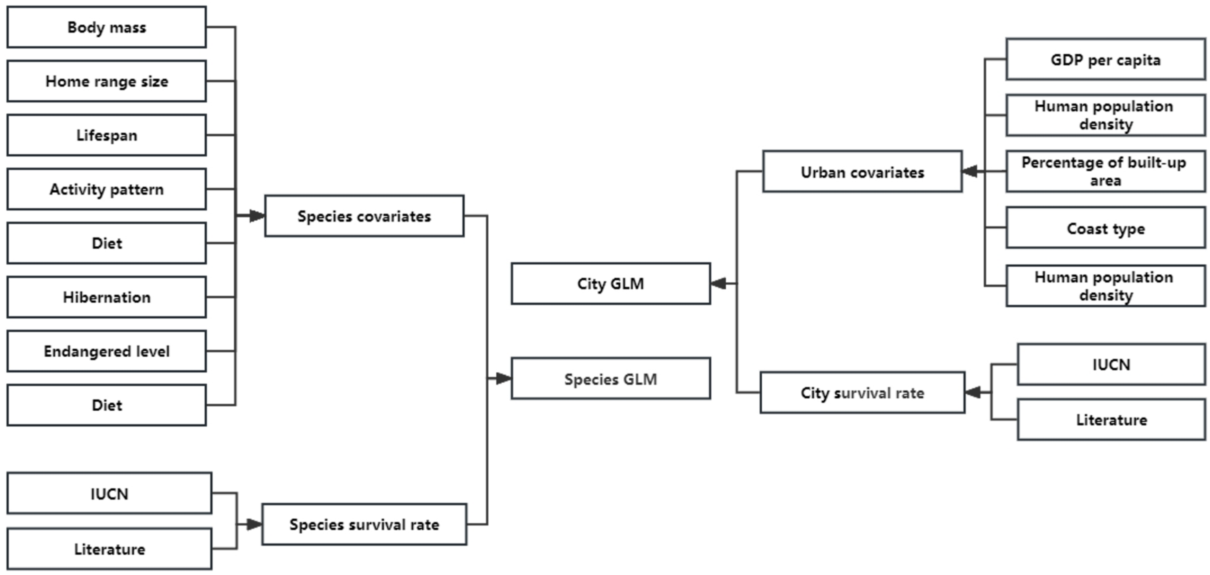


Figure 2.1: Method flow chart for Chapter 2. The main methods and variables inputted into the models used in this chapter and the links between the methods and variables, are shown.

2.3 Materials and methods

2.3.1 Methods structure

The research methodology of this chapter can be divided into three parts: 1) Collecting data: selecting the East Asian cities to be studied, collecting scientific literature or government documents in the selected cities, looking for detection reports of carnivores; screening the IUCN geographical distribution range maps of species to find the carnivores that could potentially be found in the studied cities; collecting the urbanization characteristics of the studied cities; collecting the biological characteristics of the studied species; 2) Calculate the survival rate: The survival rate of carnivores in different cities and the urban survival rate of different carnivores; 3) Modelling: use GLMs to identify the urbanization variables and species' biological characteristics most relevant to carnivore survival rates (Figure 2.1).

2.3.2 Study Area

Large Asian cities were first selected to understand carnivores' presence and determine which urban environmental variables significantly impact carnivore survival. Nine mainland Chinese cities in the top 25 of WTO's "Top 100 Asian Cities List" were selected: Shanghai, Beijing, Guangzhou, Shenzhen, Chengdu, Hangzhou, Chongqing, Nanjing, and Suzhou, and five cities in the top 25 from other parts of China and overseas, including Tokyo, Singapore, Hong Kong, Kuala Lumpur, and Taipei. Some cities, such as Jakarta and Manila, were not selected because carnivore data was challenging. The distribution of cities is given in Figure 2.2. Mainland Chinese cities' latitude and longitude, area, population, GDP, and percentage of the built-up area were sourced from the 2021 government statistical yearbook. For non-mainland Chinese cities, latitude and longitude data were sourced from Google Earth (https://earth.google.com/, and area data were from (https://unstats.un.org/unsd/demographic-social/products/dyb/). Population data were taken from the United Nations Statistics Division (UNSD) Demographic Yearbook System 2020. I then calculated population density (PD) and GDP per capita (GDPp). Information on the percentage of built-up area (BU%, the ratio of the land area of the remotely sensed land use type of built-up area to the area of the city's administrative boundaries) was obtained from government websites: Hong Kong (The Government of the Hong Kong Special Administrative Region, https://www.pland.gov.hk/pland_en/index.Html/), Tokyo (http:

//atlasofurbanexpansion.org/cities/view/Tokyo), Singapore (http://atlasofurbanexpansion.org/cities/view/Singapore), Kuala Lumpur (http://atlasofurbanexpansion.org/cities/view/KualaLumpur), Taipei (http://atlasofurbanexpansion.org/cities/view/Taipei). The topographic map was used to determine the coastal type (CT). All city variables are shown in Table A.1, including whether the city is inland, coastal, or on an island.

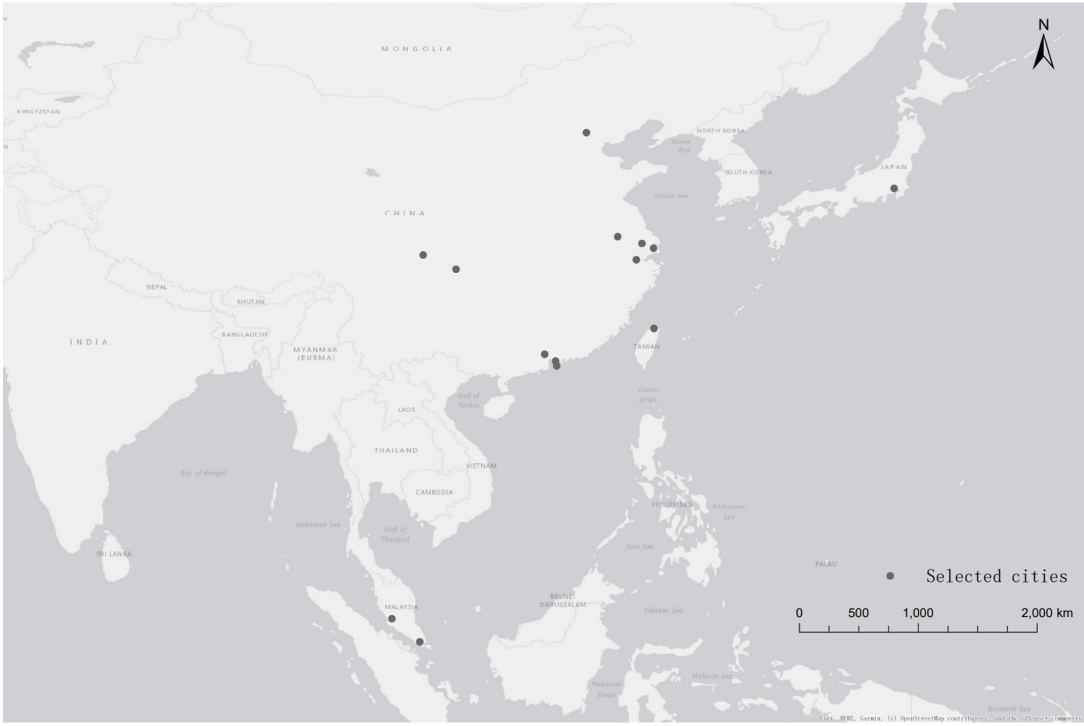


Figure 2.2: Location of the studied cities in East and Southeast Asia. I selected nine mainland Chinese cities in the top 25 of WTO's "Asia's Top 100 Cities List" - Shanghai, Beijing, Guangzhou, Shenzhen, Chengdu, Hangzhou, Chongqing, Nanjing, and Suzhou - as well as five cities in the top 25 from other parts of China and overseas, including Tokyo, Singapore, Hong Kong, Kuala Lumpur and Taipei

2.3.3 Species Detections

To obtain data on the presence of carnivores in the target cities, I undertook a literature survey. I made use of the Web of Science database to search literature from 2000 to 2022 for articles containing abstracts with the following keywords: (Shanghai OR Beijing OR Shenzhen OR Guangzhou OR Chongqing OR Suzhou OR Chengdu OR Hangzhou OR Nanjing OR Tokyo OR Singapore OR Hong Kong OR Taipei OR Kuala Lumpur) AND (Arctonyx OR Aonyx OR Lutra OR Lutrogale OR Martes OR Meles OR Melogale OR Mustela OR Ailuropoda OR Ursus OR Arctogalidia OR Paguma OR Paradoxurus OR Prionodon OR Viverra OR Viverricula) NOT (fossil OR zoo OR captive).

Firstly, by reading the titles of the articles, I excluded articles from non-biogeographical disciplines (e.g., laboratory work, microbiological studies of carnivores as hosts, etc.). A total of 57 studies matched the search criteria. Then, by reading the content of the articles, I excluded articles unrelated to the target city and carnivore species. Finally, 41 papers describing reports of the corresponding carnivores in the city were filtered out.

Carnivore records were also extracted from species lists published by the forestry department websites of Tokyo (2020 Red List of Threatened Species in Tokyo: https://www.kankyo.metro.tokyo.lg.jp/nature/animals_plants/red_data_book/redlist2020.html/), Singapore (https://www.nparks.gov.sg/biodiversity/wildlife-in-singapore/species-list/mammal), Hong Kong (Checklist of Terrestrial Mammals of Hong Kong, https://www.afcd.gov.hk/english/conservation/hkbiodiversity/speciesgroup/speciesgroup_mammals.html), Kuala Lumpur, and Taipei (https://biodiv.gov.taipei/Home/additional). In mainland Chinese cities, records of carnivore occurrences were confirmed by searching Chinese species list books. For each city, I identified the species list of Carnivora that occurred, and for each species, I identified the city list covered by its distribution.

All records of carnivore presence in the city were combined from two data sources: research articles and government websites. Data from citizen science projects were not used, as the accuracy of these data types is unknown. If the government website only had records of presence described in the local language, an e-mail was written to the local government asking for the data.

2.3.4 Potential Carnivore Presence

The IUCN Red List website (https://www.iucnredlist.org/) was used to identify the potential geographical distribution of carnivore species. I first restricted the occurrence of carnivore species to Asia and then exported the geographic distribution shape file for all selected species. I downloaded the boundary shape files for the above cities. The city boundary file was intersected with the carnivore distribution file in ArcGIS (version 10.4.1) to obtain the potential distribution of carnivore species in each city and, thus, the cities where each carnivore species might be discovered. I calculated the survival rate for each city and species by the following equations:

$$\phi_c = N_c t / N_c p \tag{2.1}$$

Here, ϕ_c is the species' survival rate of a city. $N_c t$ is the reported species richness in a city, and $N_c p$ is the potential species richness in a city.

$$\phi_s = N_s t / N_s p \tag{2.2}$$

Here, ϕ_s is the survival rate of a species. $N_s t$ is the number of cities that reported a given carnivore, and $N_s p$ is the number of cities that potentially contain a given carnivore.

For cases where a species is present in a city but its IUCN range does not cover the city, I did the following: 1) If the IUCN range is close to the city, I consider that the IUCN range underestimates the species' distribution and consider it to be potentially distributed in the city. 2) Sometimes, the IUCN range is far from the city and does not intersect with the target city. For example, the raccoon (*Procyon later*), whose geographic distribution is in North America, should not be present in Tokyo but has been reported as being detected in Tokyo (likely as a human introduction or an incorrect record). In this case, the city is not considered a potential range for the species.

2.3.5 Carnivore Characteristics

Numerous physiological and other characteristics of carnivores were considered. Body mass (BM) data were available from the Body Mass of Late Quaternary Mammals (MOM v3.3). From the IUCN Red List website, I obtained the endangerment level (ED), habitat type (how many types of habitats a species is found in, HT), and elevation range (ER) for each target species. From Animal Diversity Web (ADW, https://animaldiversity.org/, University of Michigan, Museum of Zoology), I extracted the home range size (HR), lifespan (LS), sociality (SC), activity pattern (AP), diet (Diet), and hibernation status (HB) (Table A.1).

2.3.6 Data analysis

Generalized linear models (GLMs, family: binomial) were built for cities and species, respectively (N1=14 and N2=44, N1 for the number of cities, N2 for the number of species). The Pearson correlation test was performed on all numerical predictor variables, and the correlation coefficient between all variables was less than 0.7. The z-scores of the variables were computed. A full subset regression of urban city survival rates was then performed using the "dredge()" function from the R package MuMin. I ranked all alternative models and selected the top models as those with ΔAICc (Akaike information criterion) less than two. The coefficients and confidence intervals (CIs) from these models were stored. Another full subset regression of species survival rates was calculated using the "dredge()" function. In this case, I also ranked all alternative models and selected the top models as those with ΔAICc less than two.

For each covariate in the best models, mean variable coefficients and corresponding confidence intervals were calculated to determine the direction of the covariate's effect on urban carnivore survival and the absolute value of the level of effect. The p-values for each environmental covariate were used to confirm statistical significance. Response graphs between survival rate and city/species variables were plotted.

2.4 Results

The results of the full subset regression using the city-level response variable show that only one model has ΔAICc less than 2 (table 2.1). The top model contains only one urban environmental variable, the percentage of built-up area. The second-ranked model, the null model, has ΔAICc equal to 2.78. The results indicate that GDP per capita, population density, and coastal type are not significant factors influencing the survival of urban carnivores at the city scale. In the best model, the BU% has a coefficient of -0.52 (95% CI: -0.96 to -0.08, $p = 0.025$, Table 2.2, Figure 2.3).

ID	Model	K	logLik	AICc	ΔAICc	wi
13	Built-up + Constant	3	5.76	-3.10	0.00	0.58
4	Constant	2	2.71	-0.30	2.78	0.14
	...					
47	scale(PD) + scale(GDPp) + CT + Built-up + Constant	6	0.24	-0.03	61	0.11

Table 2.1: Competing models of urban factors affecting carnivore survival. Full subset regressions of urban variables and survival were performed using the "dredge()" function. All alternative models were ranked, and the models with ΔAICc less than 2 were selected.

Coveriates		**SE**	**t**	$\mathbf{p(>\lvert t\rvert)}$
Constant	0.64	0.07	9.16	0
Built-up	-0.52	0.20	-2.56	**0.0251**

Table 2.2: The best model of urban factors affecting carnivore survival. The coefficients and corresponding confidence intervals for each covariate were calculated to determine the direction of the effect of the covariate on urban carnivore survival and the absolute value of the level of effect. P-values were used to confirm statistical significance.

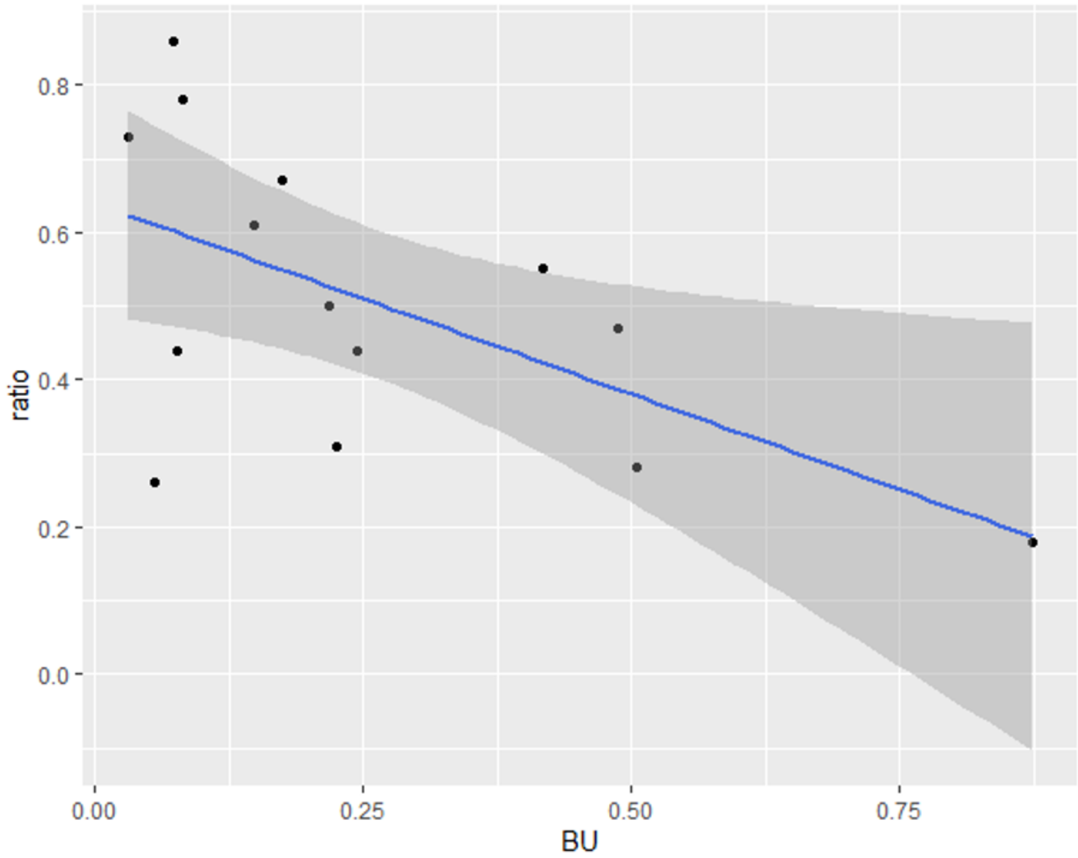


Figure 2.3: Predicted urban carnivore survival ratio as a function of Built-up (BU) area. Grey-shaded areas are 95% confidence intervals around the regression line.

The results of the full subset regression with the species-level response show that three models have ΔAICc less than 2 (table 2.3), and the top model contains three variables: Body mass, activity pattern, and sociality. The models indicate that home range size, lifespan, diet, endangered level, habitat types, elevation range, and hibernation had no significant effect on the carnivore survival rate in this study. In the best model, body mass has a coefficient of -0.13 (SE = 0.06, p = 0.03, Table 2.4, Figure 2.5). Survival rate was 45.1% lower for solitary compared to social carnivores (95% CI: -80.3% to -9.94%, $p<0.05$) and 28.4% higher for carnivores that were not active in the daytime compared to species that were active in the daytime (95% CI: 1.7% to 55.0%, $p<0.001$, table 2.4). Among social carnivores, the average

ID	Model	K	logLik	AICc	ΔAICc	wi
37	scale(BM) + AP + SC + Constant	5	4.67	-2.86	0	0.64
11	AP + SC + Constant	4	2.95	-1.73	1.13	0.19
3	Constant	2	1.46	-0.88	1.98	0.14
	...					
72	ED + HT + scale(ER) + scale(HR) + scale(LS) + Diet + HB + scale(BM) + AP + SC + Constant	12	0.12	-0.07	32	0.01

Table 2.3: Competing models of urban factors affecting carnivore survival. Full subset regressions of species variables and survival were performed using the "dredge()" function. All alternative models were ranked, and the models with ΔAICc less than two were selected.

probability of urban survival was 0.65 for those active in the daytime and 0.71 for those not active in the daytime. Among solitary carnivores, the average probability of urban survival was 0.55 for those active in the daytime and 0.61 for those not active in the daytime (Figure 2.4).

Covariates		**SE**	**t**	**p(>|t|)**
(Intercept)	0.64	0.15	4.25	0.28
scale(BM)	-0.13	0.06	-2.23	0.03
diurnon-diur	0.28	0.14	2.09	0.05
sociality	-0.45	0.18	-2.51	0.02

Table 2.4: The best model of urban factors affecting carnivore survival. The coefficients and corresponding confidence intervals for each covariate were calculated to determine the direction of the effect of the covariate on urban carnivore survival and the absolute value of the level of effect. P-values were used to confirm statistical significance.

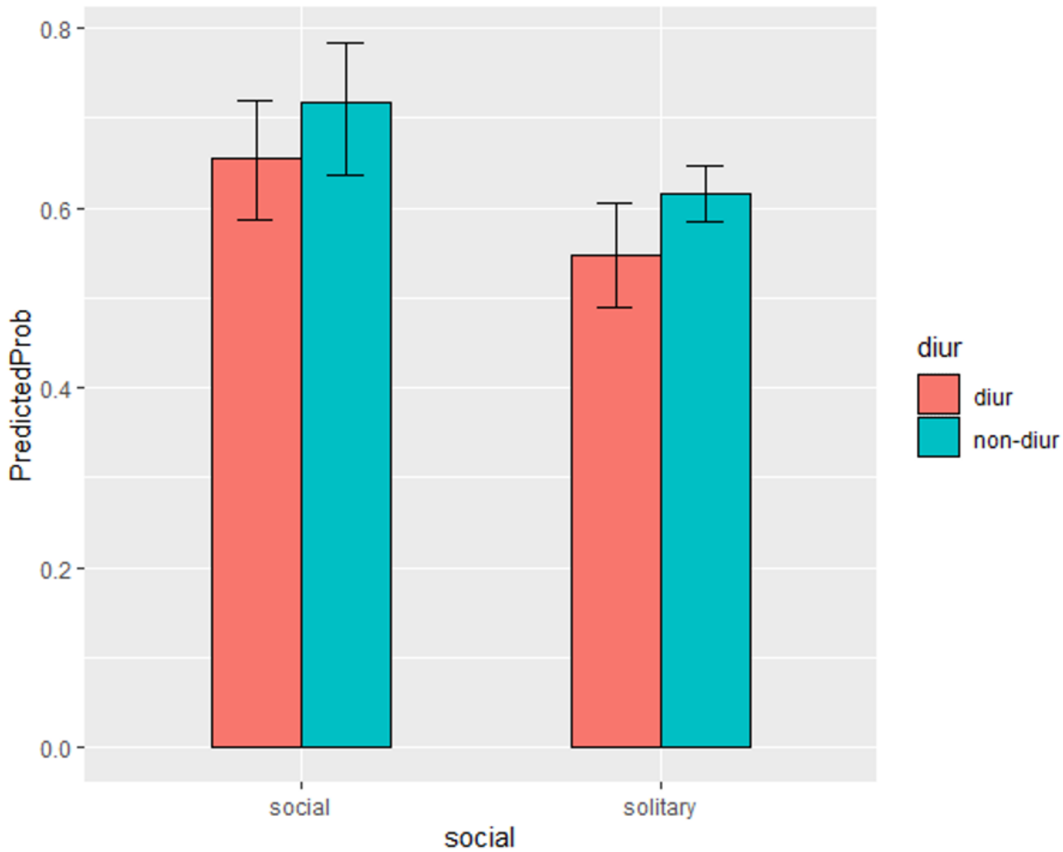


Figure 2.4: Predicted urban carnivore survival ratio as a function of species sociality (SC) and activity pattern (AP). Error bars represent the 95% confidence intervals.

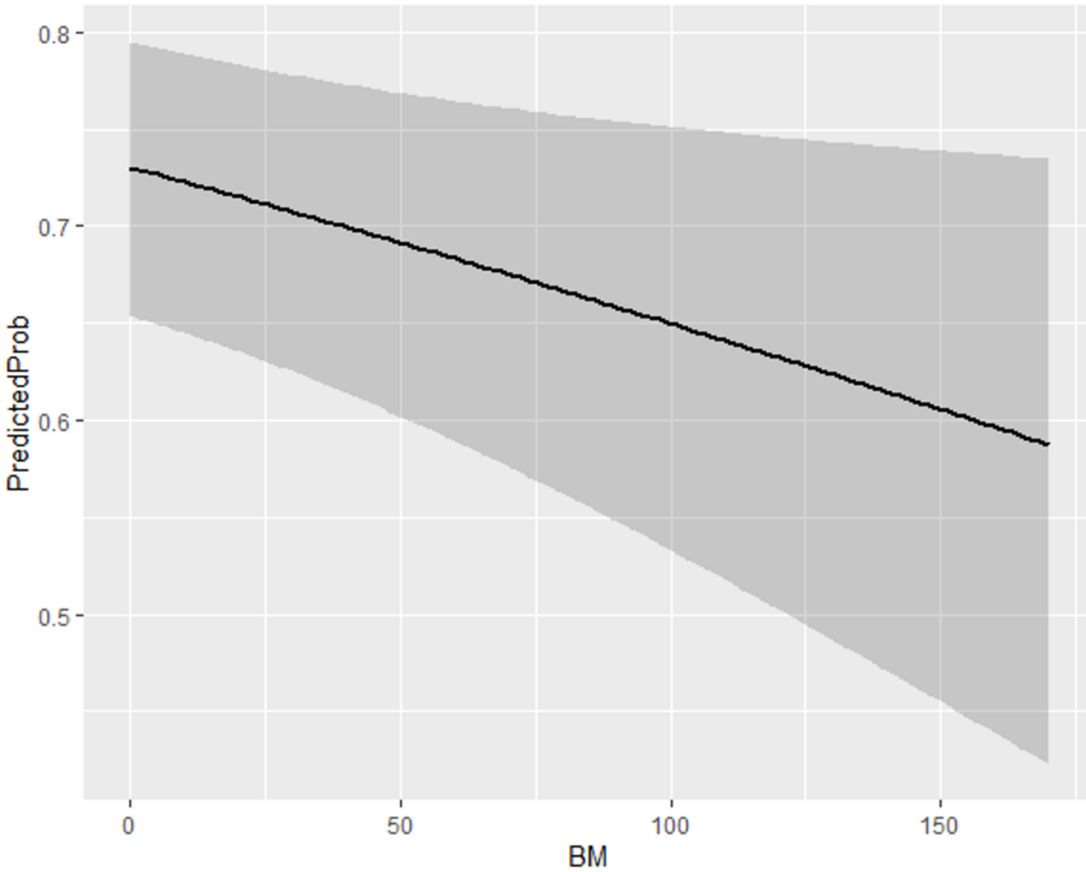


Figure 2.5: Predicted urban carnivore survival ratio as a function of species body mass (BM). Grey areas are the 95% confidence intervals.

The survival rates of different carnivore families in cities varied. As shown in Figure 2.6, Canidae and Mustelidae were the two families with the highest survival rates in cities, with median values of 0.74 (0.71, 0.76) and 0.88 (0.65, 1), respectively. The lowest survival rates were found in Felidae, Ailuridae, and Ursidae, with median survival rates of 0. The median survival rates were 0.3 (0.15, 0.45) for Herpestidae and 0.45 (0.25, 0.56) for Viverridae, which indicates that they can adapt to urban life to a certain extent.

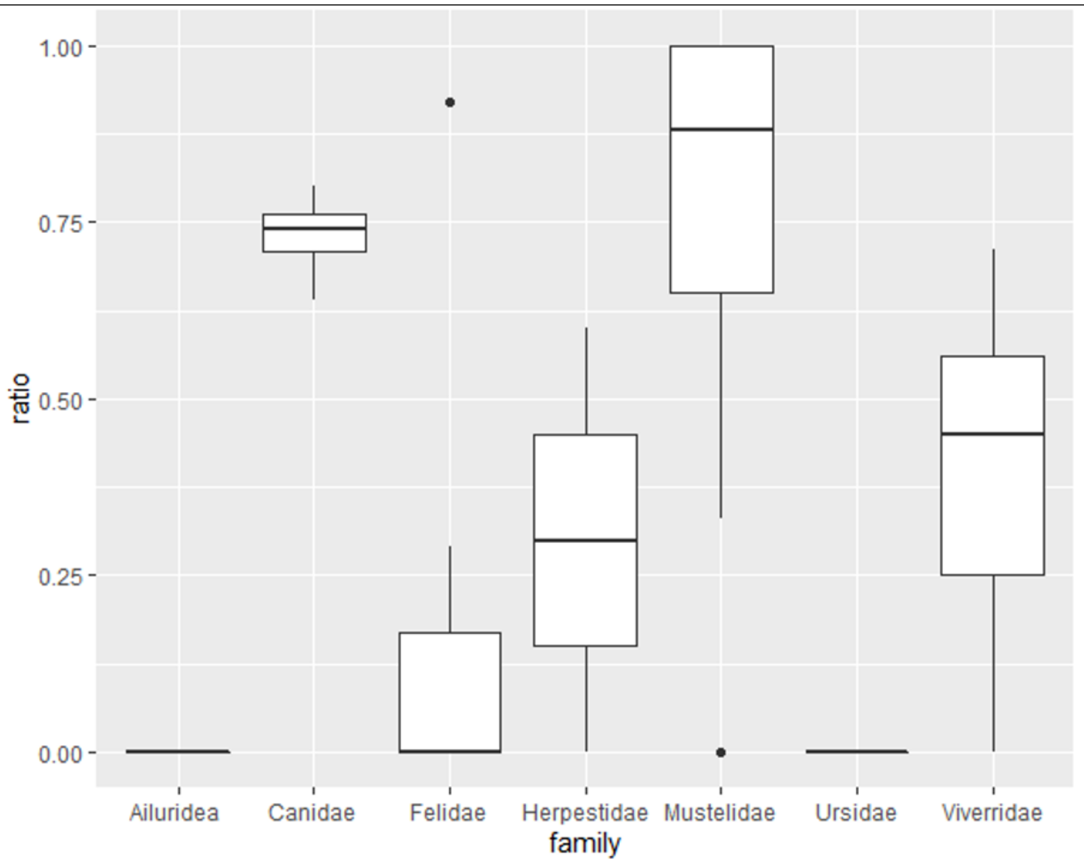


Figure 2.6: Survival ratio of different urban carnivore families.

2.5 Discussion

The results support a classic biogeographic pattern, the species-area relationship: the larger the habitat patch, the higher the species richness (*The Species–Area Relationship: Theory and Application* 2021). Previous studies have also demonstrated a correlation between patch size and isolation, with the largest habitat patches in cities being the least isolated; these larger patches also have higher species occupancy and/or population density (Prugh et al., 2008; Maseko et al., 2020; Ryser et al., 2019). The results presented here provide further evidence of the importance of patch size for the persistence of biodiversity in urban areas (Crooks K.R., 2002; Beninde et al., 2015; Diao et al., 2022). Other studies have reported findings similar to those of this paper. For example, Lee recorded 14 mammal species in 33 selected forest patches in Daejeon Metropolitan City (South Korea, Lee, 2017). In that study, mammal species richness was positively correlated with vegetation area, and habitat frequency was positively correlated with vegetation area. Numbers of Siberian chipmunks (*Tamias sibiricus*), leopard cats (*Prionailurus bengalensis*), Korean hares (*Lepus coreanus*), roe deer (*Capreolus capreolus*), and wild boars (*Sus scrofa*) were also positively correlated with vegetation area. As a further example, Mcalpine (2006) examined the occurrence of koalas in urban and rural landscapes in south-east Queensland, Australia, and found that occurrences increased with forest habitat area and the proportion of dominant eucalypt species and decreased with increases in mean closest distance between forest patches, forest patch density, and closed road density. Higher species richness in urban habitat patches implies higher food chain integrity and greater stability of urban ecosystems (Hanski, 2015). In addition, the larger the habitat patch size, the larger the population size of each species and, thus, the correspondingly lower chance of extinction and higher chance of mating success. Combining these previous findings with those of this chapter, it can be concluded that maintaining and restoring habitat areas in cities is essential for the recovery and persistence of urban biodiversity (Elmqvist et al., 2015; Ribeiro et al., 2022). Ideally, the area of natural habitat in cities would be increased, as patch size is one of the most important drivers of urban species occupancy; however, in most cases, there are considerable economic (and sometimes social) costs associated with doing so (Koch Kuser 2000, (Sustainable and Development, n.d.; Nesbitt et al., 2017); this issue will be discussed in more detail in Chapters 5 and 6.

There is a dearth of research on the species richness and survival of urban carnivores in Asia and the impacts of urbanization on these species, with more studies focusing on developed countries in Europe and the United States (Soultan et al., 2021; RILEY, 2006;

Šálek et al., 2014; Bateman and Fleming, 2012). In particular, few in-depth studies have been conducted on the leopard cat, a charismatic species of conservation concern (Izawa, 2009). While many previous studies have discussed the effects of urbanization on species in terms of urban environmental factors (Theodorou, 2022; Diao et al., 2022), the novelty of this chapter lies in also assessing the response of species traits to urbanization, i.e., which species and which species traits contribute to their survival in urban areas.

The results are consistent with the conjecture that the smaller the percentage of built-up area (often negatively correlated with vegetation cover) in a city, the greater the probability that a predator will live there. This result reiterates the importance of vegetation in the landscape for biodiversity (Lombardi, 2017; Saito, 2013). Possible reasons for this are that urban vegetation provides a rich food source for carnivores, such as rodents, birds, and other small mammals (Sorace, 2001). Vegetation types such as forests and shrubs give carnivores shelter, enabling them to effectively avoid disturbance from other predators and human activity.

To protect its remaining natural habitat, the Chinese government released the National Ecological Protection Program in 2000, which introduced the concept of a ”bottom line for ecological protection,” the beginning of China's ”Ecological Redline” policy. The first Chinese city to draw an ecological redline in urban areas was Shenzhen, which proposed a primary ecological control line in 2005. In a newly revised regulation in 2021, the Shenzhen government stipulated the core protected areas of the nature reserves. In principle, it prohibited any activities within the ecological protection redline. Further areas are banned from developmental and productive construction activities. My study confirms the need for ecological red-line policies to protect remaining habitats within cities, but whether urban biodiversity has recovered after implementing the policies in China has not been verified.

One theory is that urban animal diversity first declines before subsequently increasing as the economy grows (Otero, 2020; Trauger et al., 2003). The reason is that when the level of municipal economic development is low, the habitat area is relatively large, which benefits the maintenance of biodiversity. Moreover, when the efficiency level continues to develop to a particular stage, the government's economic investment in ecology will increase, and the public will be more aware of wildlife conservation. Contrary to my conjecture, however, I did not find a significant effect of GDP per capita on the survival rate of urban carnivores. This may be due to the number of cities selected, or GDP per capita is too coarse a variable to measure these effects.

The results also support the findings of Reilly (2017), who found that an animal has an increased probability of survival in cities if it is nocturnal or crepuscular, this way avoiding the effects of many anthropogenic activities (e.g. recreation). Also, with lower traffic volumes on city roads at night, this shift in activity patterns reduces the risk of being struck by vehicles. In addition, if a carnivore is a social animal, it is more likely to survive in the city. Possible reasons for this are as follows: first, social animals have better warning mechanisms to avoid natural predators and human activities (Wilson, 2020; Townsend, 2012). Second, social animals have an advantage in competing with other carnivores that have a similar ecological niche and are more likely to capture prey (Elbroch, 2017). Third, social animals have a larger population size, meaning they will suffer less risk of local extinction (McKinney, 2002; Griffen,2008).

2.5.1 Conservation suggestions

As mentioned above, the size and connectivity of habitats in cities have a significant positive effect on urban biodiversity, especially the diversity and abundance of carnivores (Lepczyk et al., 2017; Nor et al., 2017). Therefore, protecting remaining habitats in cities and restoring connectivity between patches is one of the essential measures to conserve urban biodiversity (Stewart et al., 2019; Nor et al., 2017). Establishing protected areas, the most common method of large-scale conservation is mostly unfeasible in cities. First, cities are densely populated, and land is a scarce resource. Second, the proximity of concentrated human settlements to these habitat patches makes restricting human activities in these habitats costly and unnecessary (Fang et al., 2022). China's approach to habitat conservation in urban areas is to use "ecological red lines" to limit urban expansion by delineating areas where urban construction is prohibited to protect habitats and biodiversity (Bai et al., 2018).

It is imperative to reevaluate the conservation objectives for the extant carnivore species within Asian cities, prioritising the conservation of species that exhibit higher sensitivity to urbanisation. For instance, carnivorous species with larger body mass are more prone to encountering conflicts with humans and becoming targets of hunting activities. These species require large daily food intake and resource and territorial demands and thus necessitate prioritisation in conservation efforts. In addition, they possess the capacity to function as umbrella species, the conservation of whom will potentially safeguard a more extensive array of other species. In a similar vein, diurnal carnivorous species have a greater number of encounters with humans and are also a higher priority for conservation. Solitary carnivores are also at greater risk.

2.5.2 Limitations

The following limitations of this chapter, all related to data availability, are relevant: 1) cities are only distributed in East and Southeast Asia, and there is still a gap in the study of urban carnivores in other parts of Asia. The main cities studied are concentrated in mainland China, and there is a lack of balance between countries; 2) potentially useful data were not always available/accessible, such as those published in local languages or restricted government notices; 3) many data on carnivore species are out-of-date, for example, those from the 1990s: many cities were less urbanized at that time, and some species may have already become locally extinct.

2.5.3 Future research

It is noted that there is a lack of data on the presence/absence of carnivores and their general ecology in relation to Asian cities. Therefore, there is an urgent need to establish systematic urban biodiversity monitoring programs in rapidly developing Asian countries, such as China and India, with centralized linkages through online accessible databases to aid future urban biodiversity research. Future Asian urban carnivore studies, based on long-term wildlife surveys and focused on carnivore species richness, population size, movements, and food composition, are needed to understand the impacts of urbanisation better.

2.6 Summary and synthesis

This study is the first comparative study of urban carnivores surviving in multiple Asian cities. It was concluded that the larger the area of natural habitat within a city, the higher the survival rate of carnivores; carnivores living in groups and inactive during the daytime have a higher survival rate in Asian cities. More citizen science and sustained government efforts to monitor the distribution of urban wildlife are encouraged. This study also recommends setting ecological red lines for urban ecological habitats and limiting urban development. Governments should prioritize conservation policies for urban carnivores, as they are critical to urban ecosystems, and many are endangered.

Chapter Three

Modelling the distribution of Leopard Cat (*Prionailurus bengalensis*) in Shenzhen and Hong Kong

3.1 Abstract

Shenzhen and Hong Kong are adjacent and share a similar species composition, but the two cities differ in the timing and intensity of urbanization. Shenzhen has undergone shorter but more intense urbanization and has a smaller area of remaining habitat. I aimed to determine whether Shenzhen's later and faster urbanization than Hong Kong has reduced the probability of leopard cat occupancy. I also tested which urbanization factors have the most significant effect on the probability of leopard cat occupancy. In this study, I investigated the difference in leopard cat occupancy probability in Shenzhen in 2019-2021, compared to Hong Kong in 2002-2006, using an infrared camera survey method to parameterise generalized linear models of leopard cat occupancy probability. Predictor variables included land use type, age of urbanization, distance to built-up land, and distance to roads. I found that eight

carnivores were extinct in the centre of Shenzhen compared to Hong Kong. I also found that the occupancy rate of leopard cats in Shenzhen was 18.8% lower than that in Hong Kong ($p < 0.01$). The number of leopard cats decreased with increasing distance from the road, and, in Shenzhen, the occupancy probability of leopard cats was higher in agricultural areas (0.75) than in urban areas (0.35).

The study results indicated that the leopard cat has a higher occupancy rate in Hong Kong than in Shenzhen, highlighting the impacts of urbanization rate and extent on leopard cat presence. Based on the results of the analyses presented in this chapter alongside a review of the relevant literature, I recommend that, in order to protect the last top predator in the core area of Shenzhen, the Shenzhen Municipal Government should 1) stick to the ecological red line policy and ban any permanent building that changes the land use type within the red line; 2) persist in monitoring and cracking down on poaching; and 3) construct ecological bridges and corridors in suitable sites, such as roadways where road kills commonly occur, to allow the leopard cat to disperse safely between different habitats patches.

3.2 Introduction

Global urbanization has been a primary driver of habitat loss and fragmentation for decades, leading to biodiversity declines. Carnivores, often the top trophic level in urban ecosystems, have typically been most affected (Mckinney, 2008; Ordenana, 2010; Bateman, 2012; Sálek, 2015). Fragmentation of habitats has decreased the availability of their prey, such as small mammals and birds, sometimes even changing their diet structure (Rich, 2017; Smith, 2018). Meanwhile, buildings and roads impede movement, mating, and gene flow, causing the fragmentation of carnivore populations (Lee, 2012; Adducci, 2020). Fragmentation also results in an increased amount of habitat patch edges; thus, humans, domestic dogs and cats are more likely to enter natural habitats and disturb carnivores.

Under such circumstances, however, some carnivores can still adapt to anthropogenic landscapes and, in some cases, thrive. One of the most successful examples of a species adapting to urbanization is the coyote, which has thrived in the cities of North America. They have expanded their dietary niche and altered their activity patterns to adjust to the novel environment provided by cities (Gehrt, 2011; Lombardi, 2017; Larson, 2020). Bobcats, previously considered sensitive to urbanization, have also been found to behaviorally adjust to cities (Young, 2019; Dunagan, 2019). However, most research on urban carnivores is concentrated in North America, lacking in other parts of the world (Bateman, 2012).

Asia has 73 species of Carnivora in 12 families (IUCN red list), and China has 61 species of Carnivora in 10 families. According to the results of Chapter 2, there are seven families and eight species of Carnivora in South China, including Shenzhen, Hong Kong, and Guangzhou: Javan mongoose (*Herpestes javanicus*), crab-eating mongoose (*Herpestes urva*), Eurasian otter (*Lutra lutra*), yellow-bellied weasel (*Mustela kathiah*), small-toothed Ferret Badger (*Melogale moschata*), leopard cat (*Prionailurus bengalensis*), masked palm civet (*Paguma larvata*), large Indian civet (*Viverricula indica*), greater Hog Badger (*Arctonyx collaris*). The results of Chapter 2 show that out of all the families of carnivores, the Feline and Ursidae have the lowest survival rates in the city, while the large-bodied Ursidae have long been extinct in the city. Therefore, it was decided that small and medium-sized cats represented the best focal taxon. According to news reports, leopard cats have been found several times in habitat patches in the central urban area of Shenzhen, while the rest of the feline species have not been reported. Therefore, the leopard cat was selected as the focal species for analysis.

Leopard cats exhibit a similar size range as that of larger domestic cats: the mean weight falls within the range of 3 to 7 kg. The length of the body exhibits a range of 44.5

to 107 cm (McDade, 2004; Smith Xie, 2008). This species' primary dietary preference is carnivorous, focusing on consuming small terrestrial animals, including rodents and lizards. Due to their adept climbing abilities, leopard cats also possess the capacity to occasionally capture avian and Chiropteran prey (Nowak, 2005). The home range of the leopard cat exhibits considerable variation, ranging from 2 square kilometres to more than 10 square km. However, it often falls between the range of 2.5 to 5.4 square km (Sanderson,2011). Leopard cats have a solitary behaviour pattern as they traverse their entire habitat in search of prey. Their elusiveness and small size make them challenging to observe. Individuals tend to avoid social interactions and tend to retreat when faced with encounters involving other individuals (Watanabe, 2009). Leopard cats frequently inhabit the peripheries of rural and agricultural regions. They are active in both the morning and evening. However, leopard cats also typically exhibit nocturnal behaviour, potentially serving as a way to reduce the likelihood of predation (Nowak, 2005). The usual lifespan of a leopard cat in its natural habitat is approximately four years. However, in captivity, it has the potential to survive for up to twenty years (McDade, 2004; Smith Xie, 2008; Nowak, 2005).

The Guangdong-Hong Kong-Macau Greater Bay Area (GBA, also known as the Pearl River Delta, PRD) in China is the largest concentration of humans on the planet, where most natural habitats have been destroyed or isolated into small, isolated patches. The GBA comprises nine megacities, including the adjacent cities of Shenzhen and Hong Kong, which have different urbanization histories. Hong Kong became one of the world's busiest ports in the 19th century and today is one of the hubs of trade and finance, whereas Shenzhen only recently transformed from a small town to one of the largest cities in the world.

Hong Kong's urbanization process started in 1842, but the massive urban population growth had not started until 1945 (Figure 3.1). Since then, Hong Kong's territory coverage by urban areas has doubled (from 12.24% to 25.1%; Yang and Huang, 2021), and the human population has increased by 47.8% (5.06 to 7.48 million; Hong Kong Census and Statistics

Department, 2022). In Hong Kong, the urban development strategy is to concentrate construction on specific areas, particularly flat areas such as Kowloon and the northern coast of Hong Kong Island. Meanwhile, many natural habitats outside the core urban areas have been spared from development, and today, 40.1% of the territory has been preserved as county parks.

Shenzhen's story is substantially different; although it borders Hong Kong, urbanization in Shenzhen only started in 1985 in response to the Economic Reform and Open-up policy. Over the past 40 years, the coverage of the built-up area in Shenzhen increased by 15 times (3% to 49%, Figure 3.1; Yang and Huang, 2021), and the human population increased by 52 times (0.33 to 17.63 million; Statistics Bureau of Shenzhen Municipality, 2022). Many formerly continuous habitats were broken into isolated patches during the urbanization process. Additionally, these patches are used for human entertainment rather than biological conservation. Their simple structure and the large amount of artificially planted secondary forests provide little support for native biodiversity.

Here, the effects of urbanization on leopard cats in China were studied for the first time. I used camera trapping and species distribution models (SDM) to compare the occupancy probabilities of leopard cats in Shenzhen and Hong Kong and to identify the urban and environmental factors that affect occupancy. I hypothesize that 1) Hong Kong has a higher carnivore species richness than Shenzhen; 2) Hong Kong has a higher occupancy probability of leopard cats; and 3) leopard cat occupancy probabilities will increase with increased distance from the built-up area and roads, lower human population density, and the duration of urbanization.

By identifying differences in the ability of the leopard cat to persist in these two cities, my findings can guide strategies to continue urban development while preserving natural biological communities as far as possible. As urbanization continues to modify natural habitats worldwide, sustainable development and the conservation of natural resources will become increasingly important. Below is a brief chapter structure (Figure 3.2).

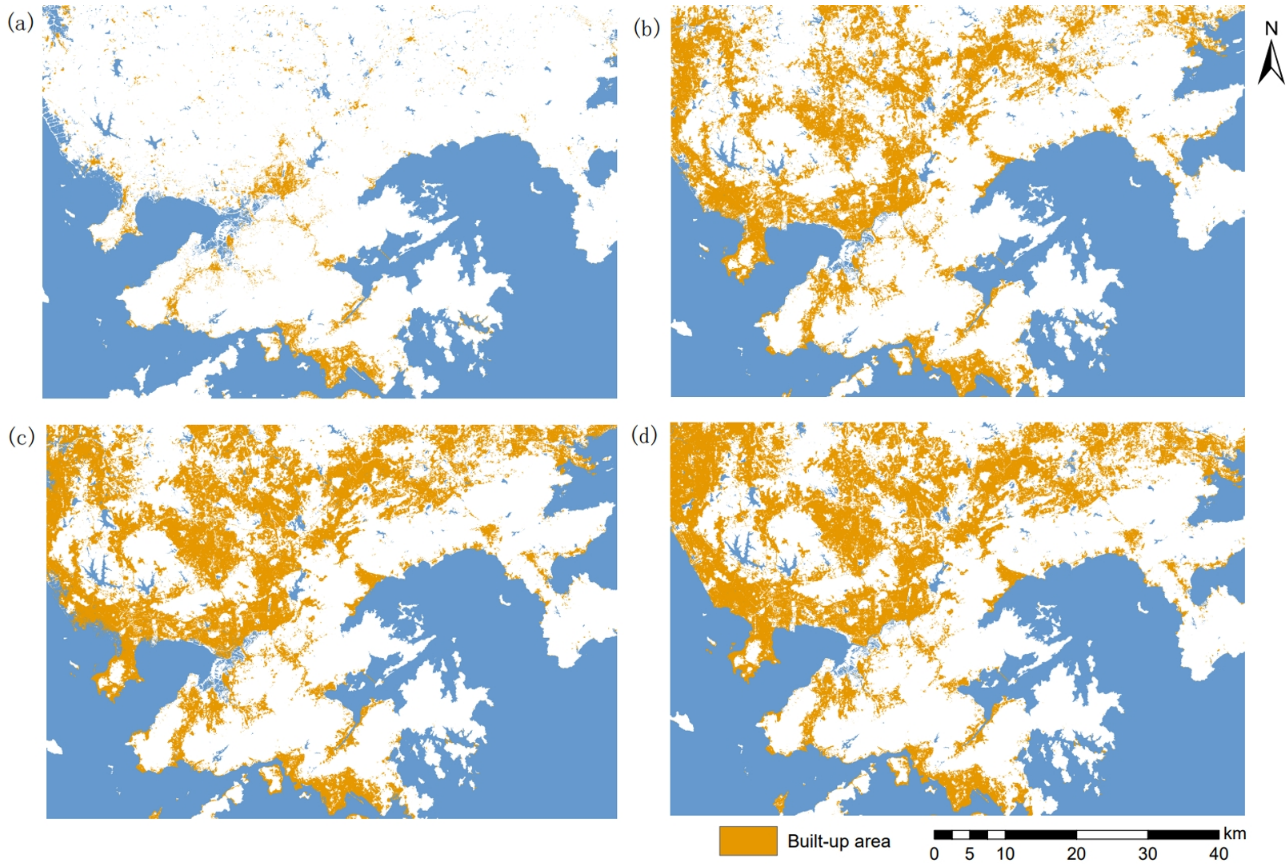


Figure 3.1: Expansion of the built-up area in Shenzhen and Hong Kong. Data extracted from Yang and Huang's (2021) dataset.

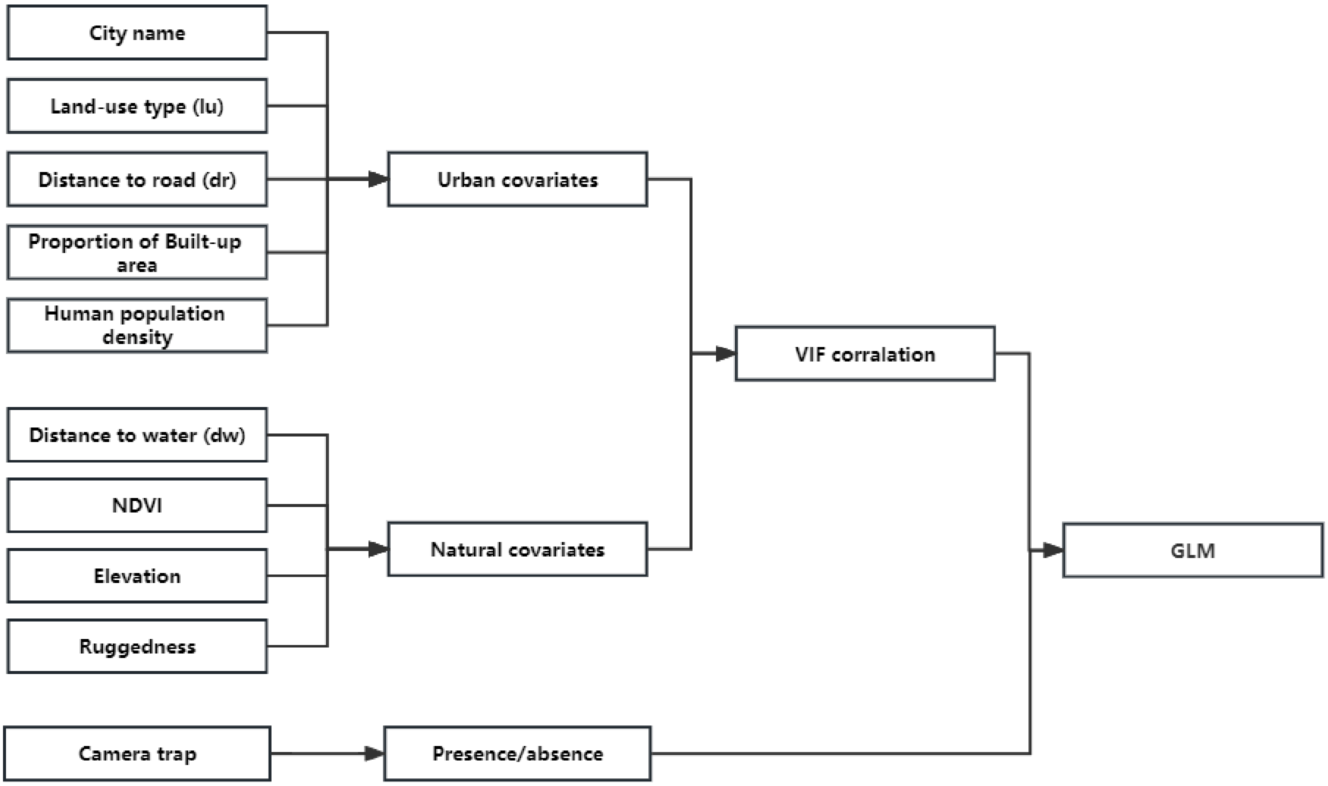


Figure 3.2: Methods flow chart for Chapter 3. The main variables and methods used in this chapter and the linkages between the different components.

3.3 Materials and methods

3.3.1 Methodological structure

The research methodology of this paper can be divided into four parts: 1) setting up the sampling method in Shenzhen and placing the infrared cameras; 2) recovering the infrared cameras and calculating the probability of leopard cat occupancy; 3) collecting data for a range of environmental factors at each sampling point; and 4) building a GLMM model to determine the predictors that significantly affect the distribution of leopard cats, and comparing these results across the two cities (Figure 3.2).

3.3.2 Study Area

The distribution and activity patterns of leopard cats in Shenzhen (22°24'-22°52' N, 113°46'-114°38' E) and Hong Kong (22°9'-22°33' N, 113°50'-114°26' E) in southern China were analyzed (Figure 3.3). These cities have an area of 1997 km^2 and 1110 km^2, a built-up area ratio of 46.46% and 25.1%, a green area ratio of 50.7% and 70.1%, a resident population of 17.63 million and 7.48 million, and a built-up area population density of 18997 and 26842 people/km, respectively (Shenzhen Statistical Yearbook 2021, Hong Kong Yearbook 2021, Hong Kong Planning Department). The region has an annual average temperature of 23.0°C, and annual precipitation is 1935.8 mm (Meteorological Bureau of Shenzhen Municipality, 2021). The natural vegetation types are tropical evergreen seasonal rainforest and southern subtropical seasonal broadleaf evergreen forest, but all forests are secondary forests and fruit tree plantations (Chen, 1985). Based on the IUCN species range maps and a literature review (see Chapter 2), Shenzhen and Hong Kong have sixteen and twelve species of potentially extant carnivores, respectively. Survey records show that eight species of carnivores are present across the two cities, of which the crab-eating mongoose (*Herpestes urva*) is found only in Hong Kong. In contrast, the Siberian weasel (*Mustela sibirica*) is found only in Shenzhen (Table 3.3).

3.3.3 Fieldwork/Camera traps

Six habitat patches were selected to survey leopard cats in Shenzhen. I used ArcGIS (10.4.1) to map 1 km × 1 km grid cells covering the target habitat patches, with 108 cells selected (Figure 3.3, Table 3.1). Surveys lasted from September 2019 to August 2021; camera traps (EREAGLE Trail Camera E1B) were placed at a randomly selected point inside each cell and were active for at least 60 days. Five plots were sampled simultaneously in each patch, and cameras were rotated to other plots in the patch every two months until all plots in all target patches had been surveyed (Figure 3.4). Cameras were mounted at the base of tree trunks near wildlife trails, 10-20 cm above the ground, facing the animal trails. The cameras were spaced at least 500 m apart. If any cameras were stolen, I increased the sampling effort within the same patch during the next survey rotation. At the end of the survey, I examined photographs and videos and compiled records of leopard cat species. The camera capture rate was defined as the ratio of independent photographs to the number of capture days (the number of 24-hour cycles in which the camera operated, i.e., until the film was full or the camera was retrieved) and multiplied by 100. These data were compared with data collected in Hong Kong by the Agriculture, Forestry and Conservation Department (AFCD) in Hong Kong, which provided a 1 km leopard cat presence/absence grid shapefile with 346 cells surveyed between 2002 and 2006 (Figure 3.3).

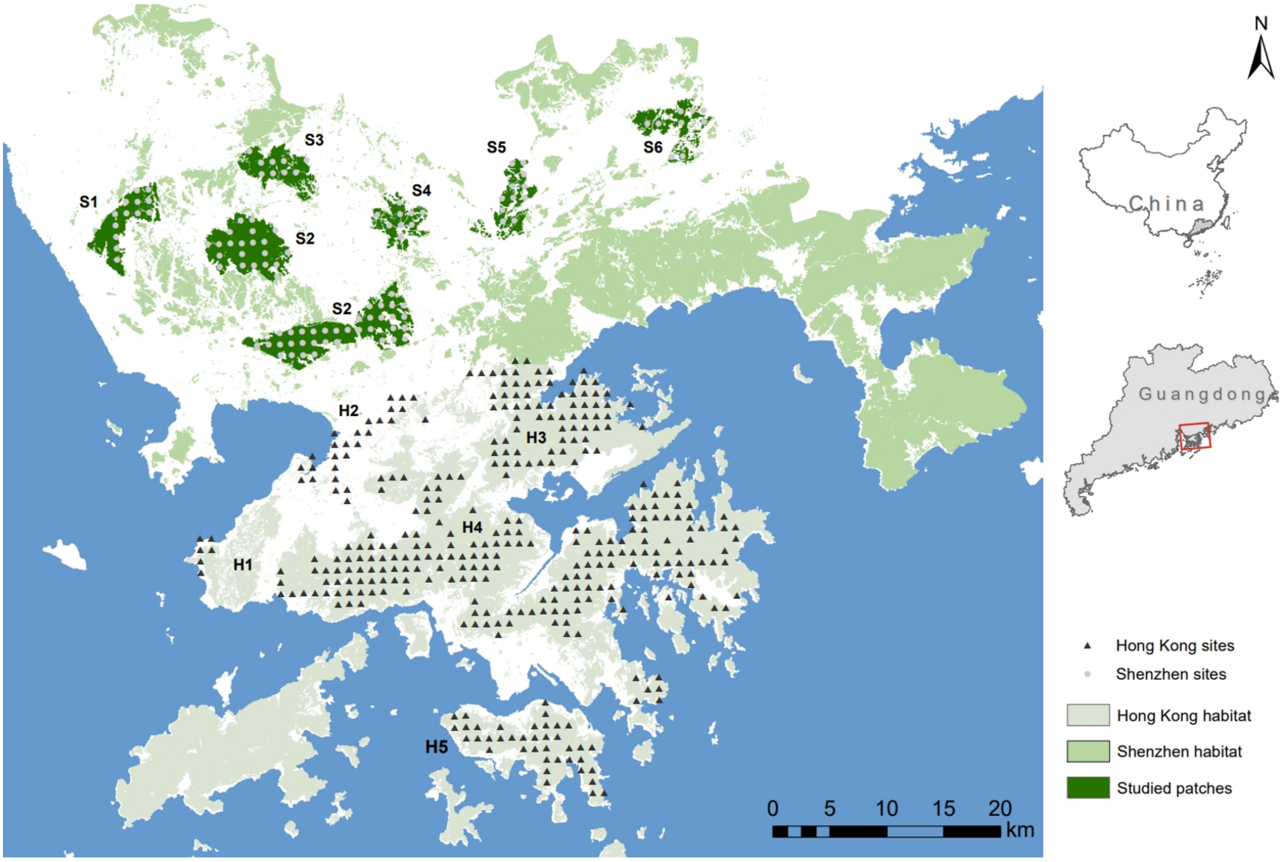


Figure 3.3: Location of camera traps and ecological core areas in Shenzhen and Hong Kong. Six habitat patches were selected for leopard cat surveys in Shenzhen, with a total of 108 1 km × 1 km grid cells. Surveys lasted from September 2019 to August 2021; camera traps (EREAGLE Trail Camera E1B) were placed randomly within each cell and were active for at least 60 days.

City	**Patch**	**Area (km^2)**	**N-Cameras**	**Time**	**Trap nights**	**Trap nights/site**	**Adjacent distance (m)**
Shenzhen	S1	171.48	55	21/09/2019-16/08/2021	4682	88.3	750.2
	S2	17.24	9	07/12/2019-26/07/2021	989	109.9	917.3
	S3	21.38	17	22/08/2019-29/08/2021	1709	100.5	781.2
	S4	32.43	9	11/12/2020-20/06/2021	641	64.1	722.6
	S5	25.07	9	15/12/2019-03/08/2021	1057	96.1	784.5
	S6	24.37	9	21/12/2019-23/07/2021	471	52.3	941.6
Total		732.64	108	22/08/2019-29/08/2021	9549	87.6	800.0
Hong Kong	H1	42.22	6	2002-2006			
	H2	33.43	27	2002-2006			
	H3	97.69	73	2002-2006			
	H4	323.01	202	2002-2006			
	H5	57.95	39	2002-2006			
Total		554.3	347				

Table 3.1: Information related to the patches and camera traps. The table shows the area of each habitat patch, the number of camera traps in each patch, sampling duration, trap nights, and the average distance between neighbouring cameras.

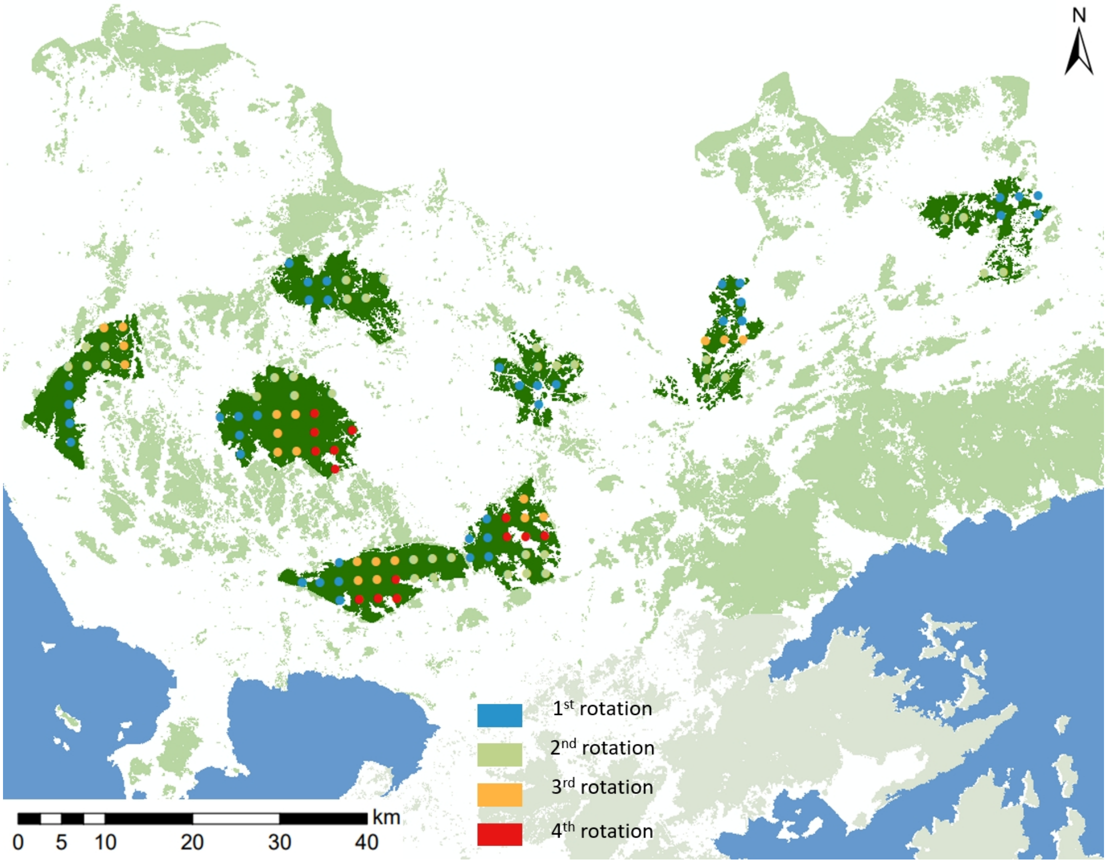


Figure 3.4: Camera site rotation map in Shenzhen. Five cells were sampled simultaneously in each patch, and cameras were rotated to other cells in the patch every two months until all cells in all target patches had been surveyed.

First, I used raster datasets to extract the environment variables. If not specified, the following datasets were for both 2006 and 2015. All raster data with a resolution of 30 m were reclassified to a resolution of 250 m. Since only grid data are available for Hong Kong, I did not use the actual point locations of the Shenzhen camera traps. Instead, I used the centres of the Hong Kong and Shenzhen sampling cells to maintain consistency in the data structure. All Hong Kong environmental variables raster maps were clipped using Hong Kong boundary data (resource and environmental science and data centre, 2015, https://www.resdc.cn/data.aspx?DATAID=201/) to remove the influence of the ocean and the Shenzhen-Hong Kong boundary (which the leopard cats could not pass).

The annual dataset of land use in China from 1990 to 2019 was first downloaded (Yang and Huang, 2021). This dataset has a resolution of 30 m and is classified into eight categories: agricultural, forest, grassland, shrub, urban, bare, water body, and wetland. I defined agricultural land, forest, grassland, and shrubs as natural habitats of leopard cats. Using the "Extract by Points" tool, each camera point's land use type (LU) was extracted. The land use types were defined by remote sensing satellite images, which were inverted according to the reflectivity bands. The city's built-up area, cropland and green land (including forests, shrubs and grasslands) were defined. From the spatial distribution of land use types in Shenzhen, the cultivated land is generally around the built-up area, and further out, it becomes green land. Therefore, the suburb area is defined as the outer 1km of the built-up area.

The boundaries of the urban and habitat polygons were extracted, and the habitat loss polygons were calculated by subtracting the 1985 habitat polygons from the 2019 and 2006 habitat polygons. I obtained road data from the Open Street Map (OSM, http://download.openstreetmap.fr/extracts/asia/), keeping only main roads and their connections (roads with IDs beginning with 511 and 513) and removing tunnels. I calculated Euclidean distances between camera trapping points and the nearest urban polygon boundary (DU), habitat boundary (DH), road (DR), and water body (DW, the closest distance to "water" or "waterways"). The Normalized Difference Vegetation Index (NDVI, Deering, 1978) from the NASA Earth DATA website (https://lpdaac.usgs.gov/products/mod13q1v006/) was used, with plate number "h28v06". The NDVI images of Shenzhen and Hong Kong started on 19/12/2019 and 30/9/2006 (corresponding to the survey periods) and lasted 16 days. I used the 30 m Aster Global Digital Elevation Model map (DEM, NASA, 2020,https://lpdaac.usgs.gov/products/astgtmv003/#tools/) for elevation. Based on the DEM file, I used the Geomorphometry and Gradient Metrics (version 2.0) toolbox (Evans, 2014, https://github.

com/jeffreyevans/GradientMetrics) to calculate the Terrain Ruggedness Index (Rug, Riley, et al., 1999; Blaszczynski, 1997) with the "Roughness" tool. In each 1 km grid, I calculated the grid-averaged NDVI, elevation (Ele), and ruggedness (Rug) using the "Tabulate Area" tool.

A "raster calculator" tool was used for each urban pixel to determine its first year as an urban land use type. I calculated the time between 2019 and 2006 (for Shenzhen and Hong Kong, respectively) as its age of urbanization. I also calculated the last year as a natural habitat for pixels once natural habitats, representing the age of habitat loss. I used the 250 m Global Human Settlement Population grid map (GHS-POP, European Commission, 2015, https://ghsl.jrc.ec.europa.eu/download.php?ds=pop). A buffer zone of 2 km was generated around the camera trap points to measure the influence of surrounding environmental variables. Since the farthest distance from the city of Shenzhen is 1825 m, I chose 2 km as the maximum distance that urban environmental variables can affect the camera sites. In each 2 km buffer, I used "Tabulate Area 2" from Spatial Analysis Supplemental Tools (SAST, Noman, 2013) to calculate the percentage area of urban (U%). I used " Zonal Statistics 2" in SAST to calculate the age of the urban land use (AU) and habitat loss (AHL) as well as the average population density (Pop). The reason for using the SAST kits is to address the issue of buffer overlap. A brief description of the environmental variables can be found in (Table 3.2).

3.3.4 Data analysis

Firstly, Shenzhen and Hong Kong were studied together. To determine which urban and environmental factors significantly affect the occupancy probability of leopard cats, I used General linear mixed-effects models (GLMMs) to model the species' occupancy (logistic regression: binomial family, logit link). The data from Shenzhen and Hong Kong were

Name	Methods	Year	Resolution	Resource	Unit	Moran's I
Land-use type (LU)	Categorical variable	2006, 2019	30 m	Yang and Huang, 2021		
Distance to urban (DU)	Euclidean distance	2006, 2019	30 m	Yang and Huang, 2021	m	0.17
Distance to habitat edge (DH)	Euclidean distance	2006, 2019	30 m	Yang and Huang, 2021	m	
Distance to Road (DR)	Euclidean distance	2014, 2021	250 m	Open Street Map	m	0.10
Distance to water (DW)	Euclidean distance	2014, 2021	250 m	Open Street Map	m	0.11
Normalized Difference Vegetation Index (NDVI)	1 km^2 grid mean	2006, 2019	250 m	MODIS/Terra Vegetation Indices 16-Day L3 Global 250m SIN Grid V006	/	0.12
Elevation (Ele)	1 km^2 grid mean	2021	30 m	ASTER Global Digital Elevation Model V003	m	
Terrain Ruggedness Index (Rug)	1 km^2 grid mean	2021	30 m	ASTER Global Digital Elevation Model V003		0.08
Proportion of urban (U%)	2 km round buffer mean	2006, 2019	30 m	Yang and Huang, 2021	m	
Age of urbanization (AU) Age of habitat loss (AHK)	2 km round buffer mean	1985-2019	30 m	Yang and Huang, 2021	year	
Human population density (Pop)	2 km round buffer mean	2000, 2015	250 m	Global Human Settlement Layer	$/km^2$	

Table 3.2: Environmental variables information. The table shows how each environmental variable was obtained, its resolution, source, unit, and amount of spatial autocorrelation (measured using Moran's I; no values were significant).

combined, and the data from the two cities were divided into two separate sub-datasets. All covariates were replaced by their z-scores. The Moran's I value for the environmental variables revealed no significant spatial autocorrelation within each variable. Pearson's correlations were used to test for multicollinearity between predictors. Based on this, I removed "DH" in all datasets (as it had a high Pearson's correlation with DU) and "DR" and "Ele" in the Shenzhen dataset (high Pearson's correlation with DU).

GLMMs were run using the "glmer()" function of the lme4 package in R to test models with different random effects structures. For the SZHK dataset, "patch" was used as the random factor for the intercept and as a random slope for DU and DR to create combinations of (1 + DU|Patch), (1 + DR|Patch), and (1 + DU + DR|Patch). The best random effect structure was determined by finding the minimum Akaike's information criterion (AIC). Then, a full subset regression with all covariates was undertaken using the best random effects structure and the "dredge ()" function of the MuMin package in R. The models with ΔAIC less than two were considered to have equal support, and all best models were averaged using the "model.avg()" function. The p-values for the model as a whole and each environmental covariate were examined to assess statistical significance. I calculated the R-squared and adjusted R-squared of the best model, indicating the percentage of the variance explained by the variables. The residuals of the best model were plotted to confirm whether they were randomly distributed and to determine the model's goodness of fit. For each covariate, the coefficients and corresponding confidence intervals were calculated to determine the size and direction of influence of the covariates on the probability of occupancy of leopard cats.

Due to the low explanatory power of the GLMMs when using the combined dataset and the fact that the City variable (i.e., whether a data point related to Shenzhen or Hong Kong) was the most important variable affecting the probability of leopard cat occupancy, the data from the two cities were then divided into two independent sub-datasets. A range of different species distribution models was then fitted to the two sub-datasets using the getevaluation()

function in the R package biomod2: Artificial Neural Network (ANN), classification tree analysis (CTA), flexible discriminant analysis (FDA), generalized additive models (GDA), generalized additive models (GAM), generalized linear models (GLMs), and multivariate adaptive regression splines (MARS). Model fits were analyzed and compared using the Area Under Curve (AUC) and True Skill Statistic (TSS). The bmVariablesImportance() function was used to determine the relative importance of different variables for Shenzhen and Hong Kong models separately. The bmPlotResponseCurves() function was used to plot the response function between the occupancy probability and the importance factors of different leopard cats in Shenzhen and Hong Kong.

3.4 Results

Two carnivore species were recorded in the 2019-2021 surveys in Shenzhen: leopard cats (*Prionailurus bengalensis*, 196 independent records) and ferret-badgers (*Melogale moschata*, two independent records, Table 3.3). In contrast, eight carnivore species have been recorded in the suburbs of Shenzhen and Hong Kong. In the IUCN Geographic Distribution Map of Carnivores, there are 16 potential carnivore species distributed in Shenzhen and 12 in Hong Kong. Of all 455 sites across Shenzhen and Hong Kong, 297 leopard cats were recorded, with a naïve occupancy value of 0.65. In Shenzhen, leopard cats were found in 55 sites (out of 108 in total), with a naïve occupancy of 0.51. In Hong Kong, leopard cats were found in 242 (346 in total) sites with a naïve occupancy of 0.70 (Table 3.4). In Shenzhen, the mean RAI of all camera sites where leopard cats were present was 0.044, indicating a low frequency of leopard cats in the survey. The patch with the highest RAI was S2 (RAI = 0.109), while the lowest RAI was found in patch S5 (RAI = 0.014, Table 3.4).

The GLMM random effect analysis results indicated that the best model did not contain

Species	**Shenzhen center**	**Shenzhen suburbs**	**Hong Kong**	**SZ IUCN**	**HK IUCN**
Herpestes javanicus		1	1	1	
Herpestes urva			1	1	1
Lutra lutra		1	1	1	1
Martes flavigula				1	
Mustela kathiah		1	1	1	
Meles leucurus					1
Melogale moschata	1	1	1	1	1
Mustela sibirica		1		1	1
Neofelis nebulosa				1	
Nyctereutes procyonoides				1	1
Prionailurus bengalensis	1	1	1	1	1
Paradoxurus hermaphroditus				1	
Paguma larvata		1	1	1	1
Panthera pardus				1	1
Viverricula indica		1	1	1	1
Vulpes vulpes				1	1
Viverra zibetha				1	1

Table 3.3: Carnivore species richness in Shenzhen and Hong Kong, and whether a carnivore species has a potential distribution in Shenzhen and Hong Kong (IUCN columns, data from Chapter 2).

City	Patch	N cameras	N presence	Naïve occupancy	N Records	Photographic Capture Rate Index
Shenzhen	S1	55	26	0.47	83	3.5±3.4
	S2	9	5	0.56	59	10.9±15.5
	S3	17	8	0.47	13	1.9±2.4
	S4	9	8	0.89	25	5.3±6.5
	S5	9	3	0.33	4	1.4±1.1
	S6	9	5	0.56	12	7.0±6.1
Total		108	55	0.51	196	4.4±6.2
Hong Kong	H1	6	5	0.83		
	H2	27	22	0.81		
	H3	73	54	0.74		
	H4	202	141	0.70		
	H5	39	20	0.51		
Total		347	242	0.70		

Table 3.4: Results from the camera traps. The table shows the total number of camera sites in each habitat patch, the number of camera sites where leopard cats were found, Naïve occupancy probability, and camera capture rate.

any random effect structure. Among all six competing models, the full model without random variables was the best (AIC = 534.41, Df = 10, weight = 0.603). Given this, the model selection was then undertaken using binomial GLMs rather than GLMMs. The full subset regression of the fixed-effect model (i.e., a binomial GLM) showed that six candidate models met the condition of ΔAIC<2 (Table 3.5). Since these six models have similar support, I averaged them to obtain the final model (Table 3.6). It was found that "city" was the environmental variable with the lowest p-value.

ID	**CITY**	**Lu**	**scl(dw)**	**scl(rug)**	**df**	**logLik**	**AICc**	**ΔAICc**	**weight**
20	+	+	0.18		4	-257.92	523.90	0.00	0.23
2	+				2	-260.11	524.20	0.30	0.20
18	+		0.16		3	-259.10	524.30	0.32	0.20
4	+	+			3	-259.17	524.40	0.45	0.19
84	+	+	0.17	0.07	5	-257.73	525.60	1.66	0.10
68	+	+		0.08	4	-258.92	525.90	2.00	0.09

Table 3.5: The best GLMs for the combined occupancy data of Shenzhen and Hong Kong. Full subset regression was used, and all models with ΔAIC less than 2 were considered to have similar support.

The city was the most important variable explaining the occupancy probability of the leopard cat ($p < 0.01$, Table 3.6). Compared with Hong Kong, the occupancy probability of leopard cats in Shenzhen was 81.2% lower (Table 3.6). According to the model predictions, the probability of leopard cat occurrence was 0.48 ± 0.10 in Shenzhen and 0.69 ± 0.05 in Hong Kong (Figure 3.5). Land-use type, a categorical variable, was non-significant in the Shenzhen-Hong Kong leopard cat model ($p = 0.148$) but indicated that the presence probability odds in the forest was 41.9% less than in cropland. The distance to water bodies and terrain ruggedness index were negatively and non-significantly related to the probability of leopard cat presence (p = 0.138 and 0.512, respectively), with β coefficients of 0.167 and 0.077 (Table 3.6).

Varibales	**Beta**	**SE**	**Adjusted SE**	**z value**	**Pr(>|z|)**
(Intercept)	1.135	0.381	0.382	2.970	**<0.01**
citySZ	-0.812	0.248	0.249	3.266	**<0.01**
luF	-0.543	0.374	0.375	1.448	0.148
scale(dw)	0.167	0.112	0.113	1.485	0.138
scale(rug)	0.077	0.117	0.117	0.656	0.512

Table 3.6: Results of the best GLM for the combined Shenzhen and Hong Kong analysis. For each environmental covariate in the best model, the coefficient and corresponding confidence intervals were calculated, and p-values were inspected.

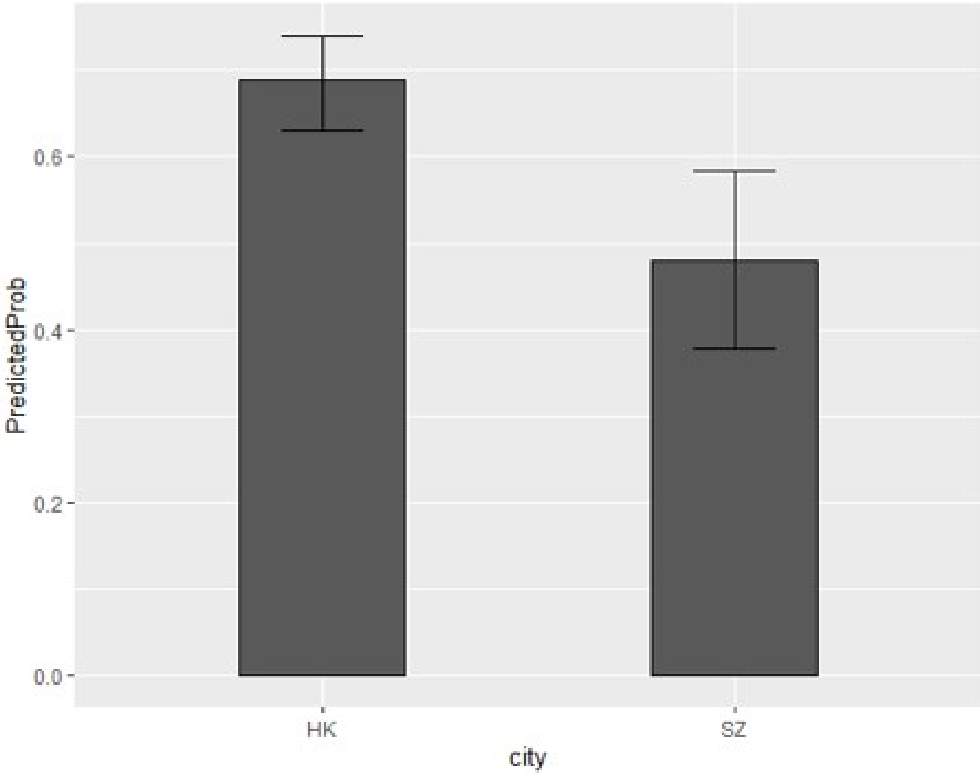
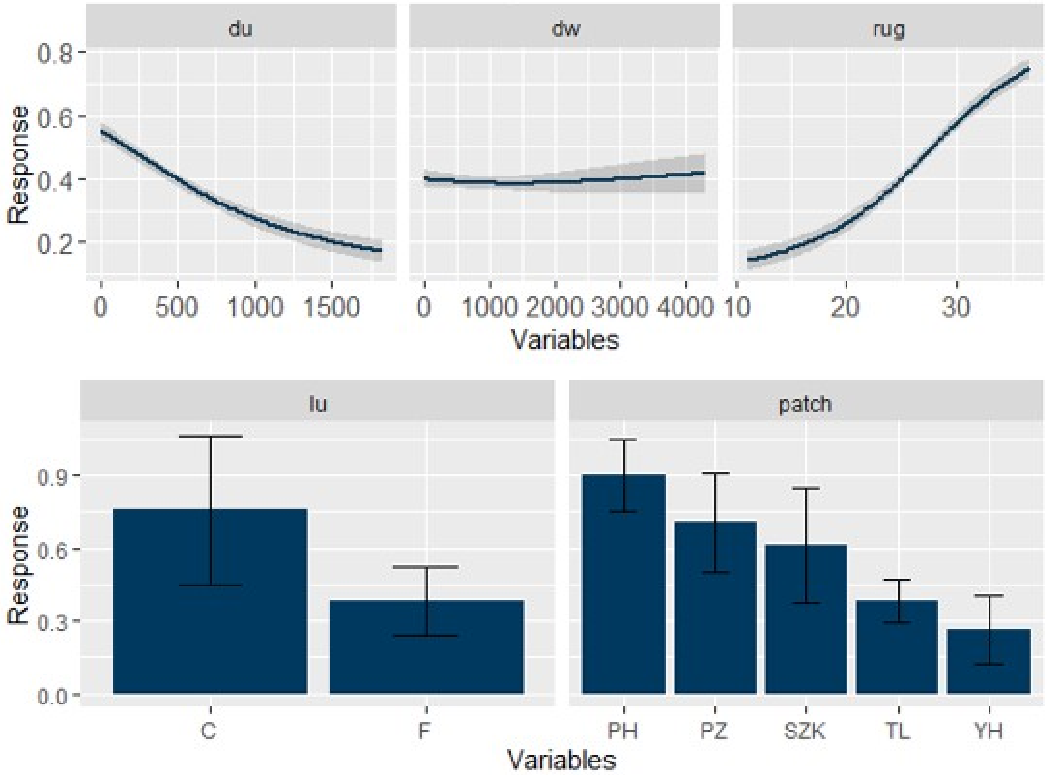


Figure 3.5: Predicted occupancy probability of leopard cat in Shenzhen and Hong Kong. Error bars represent the 95% confidence intervals.

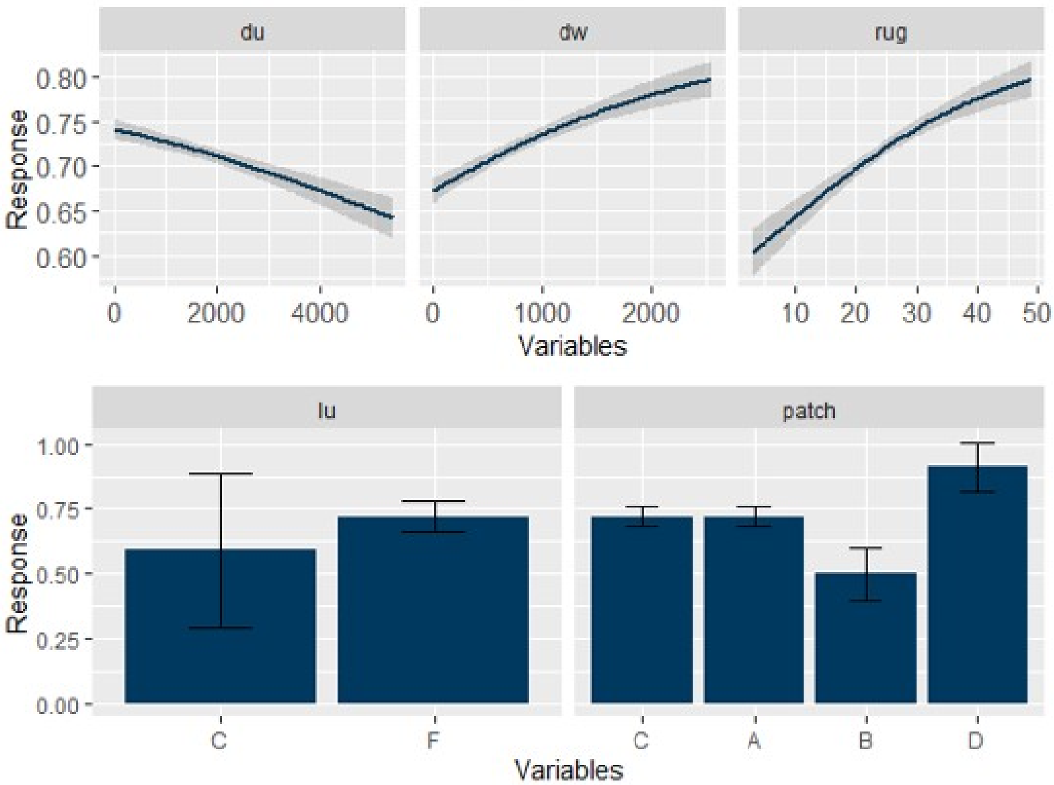
The bootstrap tests on the separate city datasets showed that all SDM algorithms had AUCs less than 0.575 and TSSs less than 0.175, indicating that none of the models could accurately predict the distribution probability of leopard cats. Among them, the GLM method with TSS of (0.16) and AUC of (0.56) was the best, and GLM was selected as the algorithm that obtained the most variable response curves. In Shenzhen and Hong Kong, Patch is the most critical environmental variable, with an importance of 0.65. The variables with importance greater than 0.1 are DU, Rug, and LU in Shenzhen and DW, Rug, and LU in Hong Kong.

In Shenzhen, the occupancy probability of leopard cats decreased from 0.55 at 0 m to

0.17 at 1750 m distance to the urban edge, and in Hong Kong, it decreased from 0.75 at 0 m to 0.65 at 3000 m distance to the urban edge (Figure 3.6). In Shenzhen, the occupancy probability of leopard cats is not related to the distance to water sources, while in Hong Kong, the occupancy probability increased from 0.675 at 0 m to 0.8 at 2500 m distance from the water source. In Shenzhen, leopard cat occupancy probability increased from 0.15 at 10 to 0.75 at 35 in terms of the ruggedness of the land, and in Hong Kong, from 0.61 at 5 to 0.8 at 50. Among the land use types in Shenzhen, the probability of occurrence is higher for cropland (0.75) and lower for forest (0.35). In Hong Kong, the results are reversed, with a higher probability of forest occurrence (0.7) and a lower probability of cropland occurrence (0.55, Figure 3.6).



(a)



(b)

Figure 3.6: Predicted occupancy probability in (a) Shenzhen and (b) Hong Kong. The bmPlotResponseCurves() function of the Biomod2 package was used to plot the response function of the occupancy probability of different leopard cats in Shenzhen and Hong Kong with respect to different variables. C refers to cropland, and F refers to forest. Error bars represent the 95% confidence interval.

3.5 Discussion

The chapter compares, for the first time, the occupancy of carnivore species in two rapidly growing Asian megacities and analyzes the effects of different anthropological, ecological, and animal physiological factors on carnivore survival. Based on my field data from Shenzhen and other data for Hong Kong, two geographically similar cities were analyzed, which differed in terms of timing of urbanization and percentage composition of land use types. This research system thus provides an ideal model for studying the effects of urbanization processes on carnivore occupancy and extinction risk.

The results demonstrate that urbanization decreases carnivore occupancy and diversity. Only two carnivores were sampled with my camera traps in the centre of Shenzhen (habitat area: 225.7 km^2), and the remaining eight native carnivores present in the Shenzhen suburbs and Hong Kong (habitat area: 425.7 km^2 and 745.6 km^2, respectively) have disappeared from the landscape. There are several possible explanations for this. First, the smaller natural habitat size in Shenzhen centre means less area for carnivores to hunt and less food (Crooks, 2002). Second, habitat patches in Shenzhen centres are relatively isolated from neighbouring patches. Carnivores are unable to travel across the city to find mates during the mating season and thus have fewer options for mates (Banks, 2007; Adducci, 2020). Third, disturbance from human activities such as hiking and biking is higher in Shenzhen centres compared to the Shenzhen suburbs and Hong Kong County parks.

This study also found that the occupancy probability of leopard cats in Shenzhen was lower than that in Hong Kong, which is consistent with my hypothesis. The explanation above for the low carnivore diversity in Shenzhen centre can also explain the low occupancy probability of leopard cats, which is, for example, due to a small amount of natural habitat. Nevertheless, the occupancy probability of leopard cats in Shenzhen (0.48) is still relatively high compared to other Chinese cities such as Shanghai. The Shenzhen government released the ”Ecological Redline” policy in 2005, and construction activities are prohibited within redlines. The Shenzhen government also released the ”Notice on the Prohibition of Hunting of Terrestrial Wildlife” in 2014, stating that the administrative area is a no-hunting area. Eleven animal traps were cleared in Dapeng New District, Shenzhen, in 2020. These policies have likely contributed to the observed leopard cat occupancy rate compared to other Chinese cities.

Hong Kong had an even higher leopard cat occupancy rate, likely due to its large amount of natural habitat. According to Section 10 of the Hong Kong Country Parks Ordinance, any proposed development within a country park is prohibited unless prior consent has been obtained from the Agriculture, Fisheries and Conservation Department (AFCD). Hong Kong encompasses a land area of around 1,108 square kilometres, with a mere 25% of this territory being developed, while the remaining 75% retains its natural state. Country parks are officially recognised areas that fall under the jurisdiction of the Country Parks Ordinance (Cap. 208). Encompassing an expansive land area of 44,300 hectares, which accounts for 40% of Hong Kong’s total land area, national parks and special areas represent the most extensive classification of protected places. These regions serve as crucial habitats for a significant majority, namely over 98%, of Hong Kong’s diverse flora and fauna.

The GLM and SDM results indicate that leopard cats are relatively insensitive to certain urbanization impacts and are perhaps even urban adapters. This result contradicts my hypothesis that the probability of leopard cat presence increases with distance to the built-up area. Notably, patch S4 in Shenzhen, which ranks 7th among all patches in terms of area, has the highest occupancy probability of leopard cats. However, the results match those of previous studies (Mata, 2017; Poessel, 2014) that found that some carnivores have higher occupancy closer to roads and the edge of the built-up area. There are two possible reasons for the above. First, leopard cats have a more mixed diet and a wider ecological niche than other carnivores. 56.7% of their prey are rodents (Fan, 2020). On the urban fringe, the population density of rodents increases due to increased human waste, enriching the food supply for leopard cats. My results found a higher probability of leopard cat presence in cropland (i.e., in locations with large amounts of cropland in the surrounding buffer) than in forests in Shenzhen, and more rodents in cropland may be the explanation. Second, leopard cats are crepuscular animals, occasionally coming out to feed at midday. Such activity patterns may help them avoid human activities (Kamler, 2020; Poessel, 2016; Suraci, 2019).

3.5.1 Conservation suggestions

Based on the results of my analyses presented in this chapter, it appears that leopard cats are relatively insensitive to urbanisation as a general process, which is promising regarding their long-term survival prospects. However, given the results presented, in addition to the findings of other related studies (e.g., Mladenoff et al., 2015; Izawa, 2009; Gregory, 2021), the following recommendations are made: 1) A crackdown on poaching is required. This could be achieved by organizing a team of forest rangers, conducting training, and strengthening patrols around important habitats. Cameras and surveillance networks should be used for real-time biodiversity monitoring. 2) As outlined in Chapter 5, ecological bridges and corridors should be constructed in suitable places, such as roads where roadkill often occurs, to help leopard cats migrate between different habitats. 3) The construction of roads in the remaining natural habitats should be avoided. As road density increases, the availability of suitable habitat for many species declines. For example, Mladenoff et al. (2015) showed that road densities > 0.6 km/km^2 make landscapes unsuitable for wolves in Wisconsin, US, and road avoidance behaviours have eliminated more than 10.9 million hectares of potential elk habitat in the United States (Izawa, 2009).

3.5.2 Limitations

This work has the following limitations: 1) The surveys in the two cities were undertaken in different periods (2006 and 2019). The survey duration was also different. Hong Kong took place over five years, while Shenzhen was only three years. This may help explain Shenzhen's lower leopard cat occupancy probability; 2) The R-squared of the GLMs and the AUC and TSS values of the SDMs are small, indicating that the explanatory power of the models is low. One reason for the lower explanatory power may be the number of camera traps used in Shenzhen, which was lower than initially planned due to the COVID-19 pandemic. 3) due to limitations in camera resolution and the clarity of nighttime photos, I was unable to identify individual leopard cats, and thus, I could not accurately calculate the population size and density; 4) since I could not obtain detailed detection history data and latitude/longitude of the camera sites from the HK survey, I simplified the data for Shenzhen to ensure consistency in the data from the two cities. For each Shenzhen camera site, only the centre point of the grid square in which the camera was located was taken, thus throwing away fine-scale location information; and 5) the effect of environmental variables on the probability of detection was also not considered, potentially biasing the estimation of leopard cat occupancy probability.

3.5.3 Future research

Future research should focus on urban carnivores' food chains, activity trajectory, and genetic connectivity. 1) For the food chain, the faecal composition of leopard cats should be analyzed to determine the food sources of leopard cats and the extent to which leopard cats use human waste. In addition, the relationship between the main prey (mainly rodents) and the distance from the built-up area should also be analyzed to determine whether the prey of leopard cats increases due to urbanization. 2) Activity trajectories need to be monitored by

GPS collars to show whether leopard cats are crossing built-up areas to interact with other isolated patches for population communication. In addition, the dividing roads between patches need to be monitored. Does roadkill occur when leopard cats cross these roads, and how does it affect their population size? 3) Finally, genetic connectivity needs to be explored to see if there are already isolated metapopulations across isolated patches in the Shenzhen centre and whether populations still communicate genetically. If not, when did populations become isolated, and how did this isolation relate to the process of urban construction?

3.6 Summary and synthesis

In conclusion, this chapter examined the effects of urbanization on leopard cats in the Pearl River Delta and compared two megacities in the region. This study is the first to examine urbanization's effects on the Asian leopard cat. Although the models had low explanatory power overall, it was found that the species was relatively insensitive to urban environmental variables, was an urban adaptor, and could survive in the central area of Shenzhen, whereas the other eight carnivores could not. Hong Kong has extensive and well-connected agricultural and forested habitats, so leopard cats have a higher occupancy rate. While the species is able to survive in urban areas, natural habitat in cities is still important, and thus, it is imperative to protect the only remaining habitat of this top predator in central Shenzhen to ensure its long-term persistence.

Chapter Four

Urbanization exacerbates the spatial and temporal overlap of carnivores with other species

4.1 Abstract

Urbanization has reduced species' home ranges, population sizes and distributions and increased the spatial and temporal overlap among species. Urbanization has also involved the introduction of exotic species, such as stray cats and dogs, which has exacerbated this overlap and the resultant increases in competition. The aim of this chapter was to assess the extent to which leopard cats have spatio-temporal overlap with humans, stray cats, stray dogs, and wild boars in the urban centre of Shenzhen. Using survey data from infrared cameras, I studied the distribution and temporal activity patterns of leopard cats in Shenzhen using kernel density estimates. The spatiotemporal distribution patterns of leopard cats overlapped with those of humans, stray cats, and stray dogs and potentially indicated increased ecological niche competition between leopard cats and these other species. Overall,

I found that leopard cats had $> 55\%$ spatial overlap with humans, stray cats, stray dogs, and wild boars. Regarding temporal overlap, leopard cats had the highest overlap with stray cats, with peak activity times at 06:00 and 23:00, 3-4 hours later than the peak activity times of stray cats (02:00 and 20:00). The results indicate that the rates of competition between carnivores and introduced species in cities may be high and that carnivores may adopt strategies to avoid such species in space and/or time. This study advises reducing the number of domesticated animals in urban areas, primarily by sterilization, and introducing rules to reduce the abandonment of domesticated animals. There is also a need to control recreational users' access to natural habitats to minimize the negative implications of human-wildlife overlap.

4.2 Introduction

Urbanization adversely affects urban biodiversity. First, urban expansion has reduced natural habitat size and increased habitat fragmentation. In turn, this has significantly reduced the distributions and population sizes of many species, as well as increasing the amount of spatial and temporal niche overlap between species, which leads to increased competition, e.g., for food (Cartera, 2012; Smith, 2018; Prigioni, 2014). Second, urbanization has dramatically increased the number of stray cats and dogs in cities, which tend to have very high birth rates (Reilly, 2017; Lacerda, 2009). This creates competition with other carnivores in the system. In addition, many of the remaining fragmented patches of natural habitat in cities are used as municipal country parks, with an increased frequency of human disturbance activities such as hiking, biking, and even poaching. These activities can trigger changes in urban wildlife activity time as well as other life history characteristics, such as diet (Wang, 2015; Riley, 2003).

Increased competition due to increased ecological niche overlap is arguably one of the key drivers of urban biodiversity decline. However, it is relatively understudied. The escalation of spatial and temporal convergence between human and wildlife populations has been observed to amplify the occurrence of human-animal conflict. These conflicts, characterised by confrontations between humans and wildlife due to their proximity, frequently culminate in instances where humans resort to self-defensive measures by killing animals or engaging in preemptive or retaliatory killings. Such actions have the potential to contribute to the extinction of certain species. Increased competition for food can also result in predators less adapted to urban life being disadvantaged during predation.

Carnivores adjust their spatial ranges and activity times to reduce the effects of spatial and temporal overlap on species populations (Gaynor, 2018; Lourraine, 2002). Two species with high spatial overlap would each adjust their active time and avoid each other to hunt (George, 2006). Two carnivores with abnormal temporal overlap will avoid each other's activity ranges to reduce the probability of an encounter (Grinder, 2001). Appropriately altering their diets to capture different prey is also a common strategy to decrease the adverse effects of spatial and temporal overlap (Wang, 2015).

Worldwide studies on carnivores' spatial and temporal overlap primarily focus on large carnivores and protected areas (Smith, 2018). Similarly, spatial and temporal overlap studies of carnivores in China have focused on tigers, snow leopards, and panthers. In contrast, only 29% of studies have focused on small and medium-sized carnivores (Li, 2021). In addition, little is known about the spatial and temporal overlap of carnivores in inner-city habitats and how it affects their survival.

Here, I study the effects of spatial and temporal overlap between leopard cats and other species in one of the youngest megacities in southern China: Shenzhen. The spatial and temporal overlap of distinct species with leopard cats was analysed using camera trapping monitoring and kernel density estimation. My scientific questions were (i) which species overlap with leopard cats in urban habitat patches, (ii) what the pattern of overlap is, (iii) how the spatial and temporal distribution of leopard cats is affected by this overlap, and (iv) how it may change in future. I hypothesize that stray cats have the highest spatial and temporal overlap with leopard cats and that human activities cover all leopard cat distribution areas. I also predict that leopard cats adjust their spatial and temporal distribution and diet to reduce competition from stray cats and disturbance from human activities. A brief chapter structure is shown below (Figure 4.1).

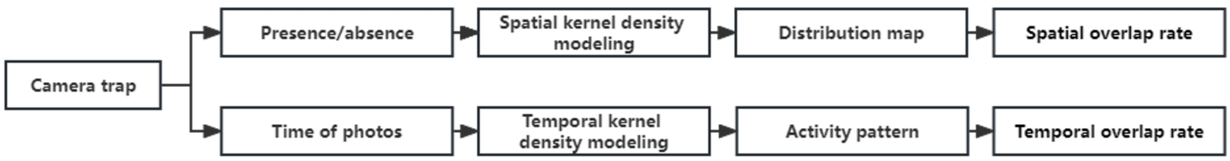


Figure 4.1: Methods flow chart of Chapter 4. The main methods used in this chapter and the links between the methods are shown.

4.3 Materials and methods

4.3.1 Method structure

The research methodology of this paper can be divided into four parts: 1) calculating the spatial distribution range of each species using kernel density estimation based on the location and number of photographs taken by the infrared cameras; 2) calculating the activity patterns of each species based on the time of day when the infrared cameras took photographs; 3) calculating the spatial distribution overlap rate of the leopard cat with that of other wildlife as well as with that of human activities; and 4) calculating the temporal distribution overlap rate of the leopard cat with that of other wildlife as well as with that of human activities (Figure 4.1).

4.3.2 Spatial overlap

The study area and the information on the camera trapping deployment are described in detail in the methodology section of Chapter 3. In this chapter, captured photos and video files were also processed. Firstly, the focus in this chapter only relates to the camera trapping survey in Shenzhen from 2019 to 2021. Secondly, I no longer use the centroids of the sample grid but use the actual locations of the cameras for analysis. Finally, I examined the photos to filter out files containing humans, stray dogs, stray cats, and wild boar. Using the ”readexif(,)”, I extracted the function in the ”exifr” package of the R language, the time when the camera took each image file. I used the ”mutate()” function in the ”dplyr” package to filter files with a shooting time interval larger than 30 minutes as a separate occurrence record for each species.

To study the overlap of the geographic distribution of leopard cats with other species, the first step is to determine the ranges of the different species. As analyzed in Chapter 3, urban and natural environmental variables such as topography and vegetation do not accurately predict the distribution of leopard cats in this study system. Therefore, I chose a simple kernel density estimation (Teng,2011; Karelus, 2016) method to calculate the distribution of the species. In ArcGIS, I utilized the ”Kernel Density Estimation” function to estimate a ”heat map” of species distribution using the presence/absence of data for different species at the camera loci. I chose the median of all density values, and the area larger than the median is the probable distribution range of the species (Teng, 2011). I calculated the size of the distribution range. Using the ”intercept” function, I overlapped the distribution ranges of humans, stray cats, stray dogs, and wild boar with the potential distribution range of leopard cats. I then calculated the size of the overlapping areas. To calculate the spatial overlap rate between species I used the following equation:

$$O_{ab} = \sqrt{H_{ab}{}^2/(H_a H_b)} * 100 \tag{4.1}$$

Where O_{ab} is the spatial distribution overlap rate of species a and b, H_{ab} is the area of the overlapping range of the two species, and H_a and H_b are the areas of the distribution range of species a and b, respectively (Gould, 2013).

4.3.3 Temporal overlap

To study the temporal overlap between species, I used the ”overlapPlot()” function from the ”overlap” package to plot the change in the activity level of each species over 24 hours. The activity curves of humans, cats, dogs, and pigs were overlaid with the leopard cats. I assessed the peak activity times for different species. The temporal overlap coefficient between two species was estimated using the ”overlaps()” function in the ”overlap” package. Due to the small amount of recorded data, I utilized the ”Δ4” coefficient for the simulation using a ”bootstrap()” with 1000 cycles (Ridout Linkie, 2009; Schmid Schmidt, 2006). The 95% confidence interval was estimated using the ”bootCI()” function (Berger, 2007). I repeated the same procedure for both human and wild boar species.

4.4 Results

The recording of leopard cats has been outlined in detail in the results section of Chapter 3. In addition to this, for the camera trapping survey in 2019-2021, I obtained additional photos of 212 leopard cats, 13,965 humans, 579 stray cats (without owners), 1660 stray dogs(without owners), and 178 wild boars. The numbers of independent records were 1214 stray dogs, 106 wild boars, 491 stray cats, 7774 humans, and 188 leopard cats, respectively. Of all 108 camera loci, the number of camera loci recorded in the above species was 75 for stray dogs, 30 for wild boar, 49 for stray cats, 106 for humans, and 55 for leopard cats. Eleven of these loci recorded only one species, five camera loci located in S1, and one camera loci in the S3 patch recorded all five species, including leopard cats and wild boar. Leopard cats had the highest probability of being present in the same locus as stray cats, with 29 units and a probability of 26.9The distribution areas of leopard cats, humans, stray cats, stray dogs, and wild boars in western Shenzhen's principal urban habitat segment were 96.4, 91.0, 84.1, 129.6, and 39.1 km^2, respectively (Figure 4.2, Figure 4.3, Figure 4.5, Figure 4.4). The distribution range of leopard cats was mostly concentrated in the densely populated patches S1 and S3 near the south, with 43.9 and 17.5 km^2. The distribution area in patch S4 was also larger at 13.6 km^2. There was no distribution in S5 and S6 in the east. The distribution range of human activities is comparable to that of leopard cats, except there is less distribution in patch S2 and more in patch S5 (area 7.5 km^2). The distribution of stray cats was also comparable to that of leopard cats, increasing the distribution in northern patch S2 (area increased by 5.7 km^2) and decreasing the distribution in southern patches S3 and S4 (area decreased by 18.5 km^2). There was an increase and decrease in the distribution area for patch S1. Stray dogs had a larger distribution area than all the above species, increasing their distribution in patch S5 (area 11.6 km^2). The distribution of wild boar only included the southern patches S1 and S3 (areas 24.7 and 12.9 km^2, respectively) (Figure 4.4).

The distribution ranges of humans and leopard cats largely overlapped, except for patch S5. The distribution range of leopard cats contained that of stray cats, with a different pattern to patch S1. The distribution range of stray dogs contains that of leopard cats, with distinct patterns in the northern part of S1. The distribution range of leopard cats contains that of wild boar. The range of human activity contains the range of wild boar (Figure 4.4, Figure 4.5). The spatial overlap rates of leopard cats with human activities, stray cats, stray dogs, and wild boar were 0.69, 0.63, 0.73, and 0.58, respectively. The highest overlap rate was with stray dogs. The spatial overlap rate between human activities and wild boar was 0.63 (Figure 4.6).

The activity pattern of leopard cats showed a strong nocturnal morning and evening pattern. The peak activity times were 6:00 and 23:00. Human activity was focused at 9:00 and 15:00, and activity in the afternoon was more frequent than in the morning. The activity pattern of stray cats is very close to that of leopard cats, which are also nocturnal in the morning and evening. However, their peak activity was a few hours earlier than leopard cats, at 2:00 and 19:00. Stray dogs showed morning, evening, and daytime activity, with peak activity at 7:00 and 16:00. Wild boar activity was mostly concentrated at 9:00. The variation throughout the day was not so great as in the other four species (Figure 4.2, Figure 4.3, Figure 4.4, Figure 4.5).

The greatest time overlap between leopard cat and human activity was at 7:00 and 16:00, with less time overlap of 0.22 (0.10-0.27), the smallest overlap among the four species. The time overlap between the leopard cat and the stray dog was similar to that of humans, although the overlap was higher at 0.37 (0.22-0.48). Leopard cats and stray cats had the highest overlap rate of 0.68 (0.45-0.79), concentrated at 21:00 and 23:00-5:00. The overlap time between leopard cats and wild boar lasted from 19:00 to 8:00 the next day, with a time overlap coefficient of 0.54 (0.39-0.77). The overlap time for humans and wild boar was from 8:00 to 18:00. The overlap coefficient was 0.57 (0.44-0.76) (Figure 4.8).

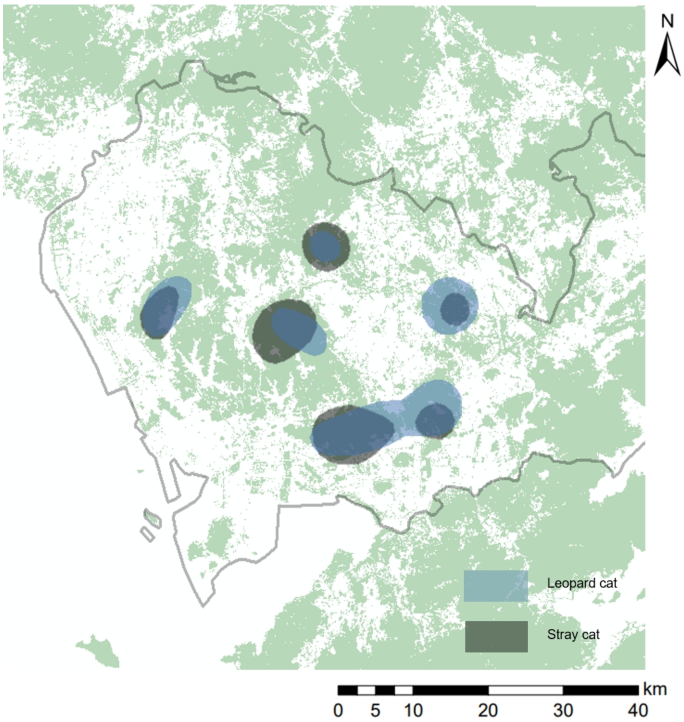


Figure 4.2: Spatial overlap between leopard cat and stray cats. Using the ”Kernel Density Estimation” function, presence/absence data for different species from camera-located points were used to estimate a ”heat map” of species distributions. Here, the distribution of stray cats overlaid with the likely distribution of the leopard cat.

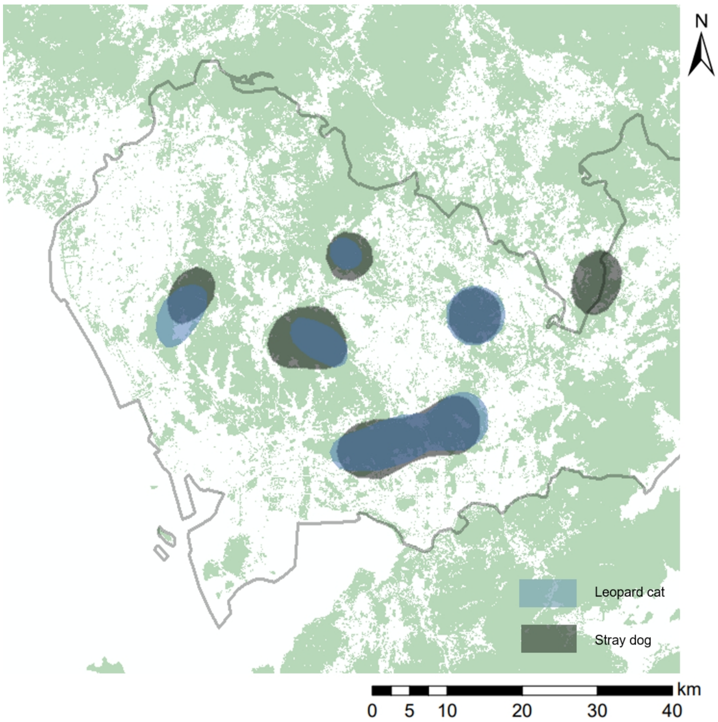


Figure 4.3: Spatial overlap between leopard cat and stray dogs. Using the ”Kernel Density Estimation” function, presence/absence data for different species from camera-located points were used to estimate a ”heat map” of species distributions. Here, the distribution of stray dogs overlaid with the likely distribution of the leopard cat.

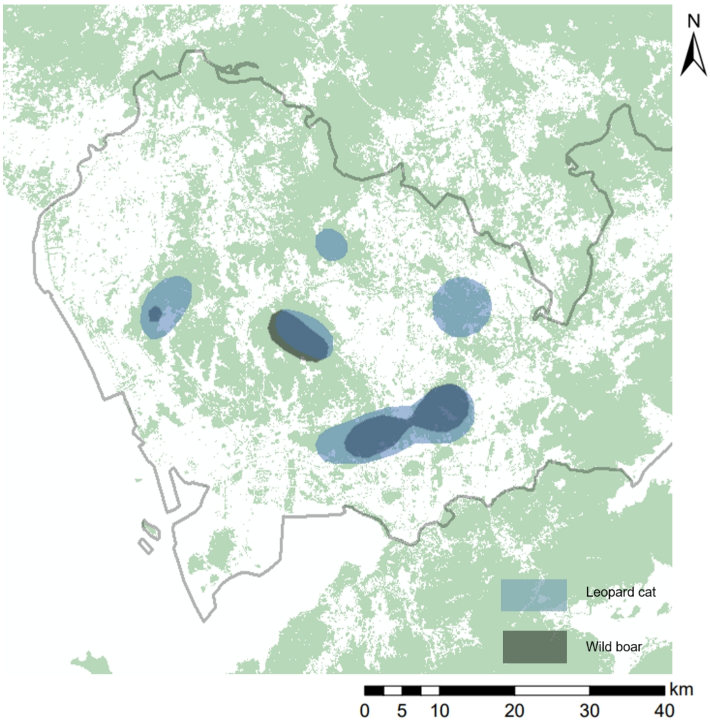


Figure 4.4: Spatial overlap between leopard cat and wild boars. Using the "Kernel Density Estimation" function, presence/absence data for different species from camera-located points were used to estimate a "heat map" of species distributions. Here, the distribution of wild boars overlaid with the likely distribution range of the leopard cat.

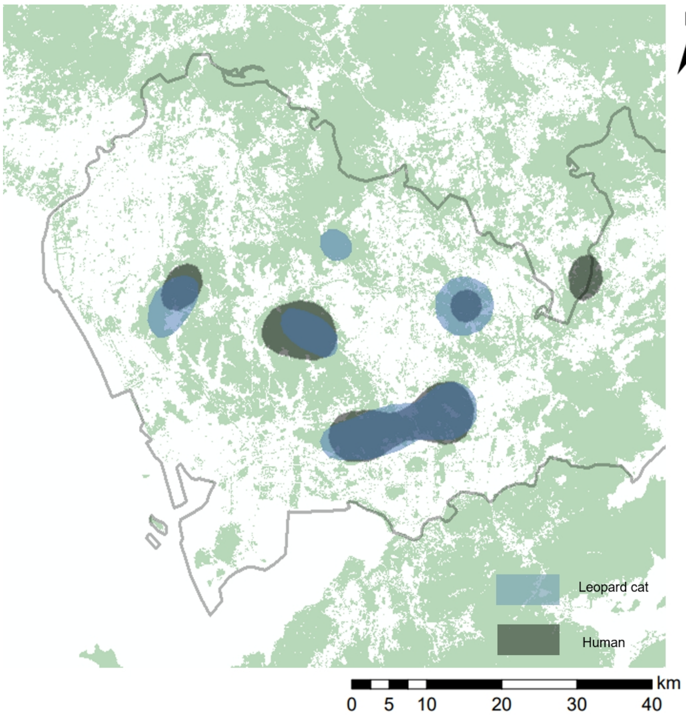


Figure 4.5: Spatial overlap between leopard cat and human beings. Using the "Kernel Density Estimation" function, presence/absence data for different species from camera-located points were used to estimate a "heat map" of species distributions. Here, the distribution of human beings overlaid with the likely distribution of the leopard cat.

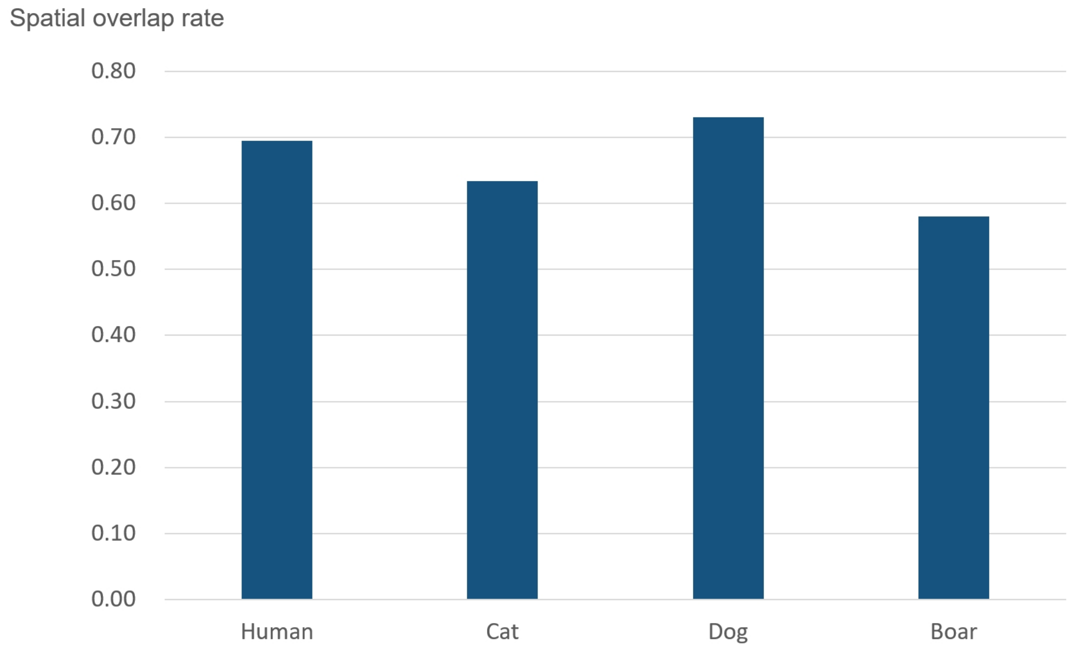


Figure 4.6: Spatial overlap rate between leopard cats and four other species: humans, stray cats, stray dogs, and wild boars.

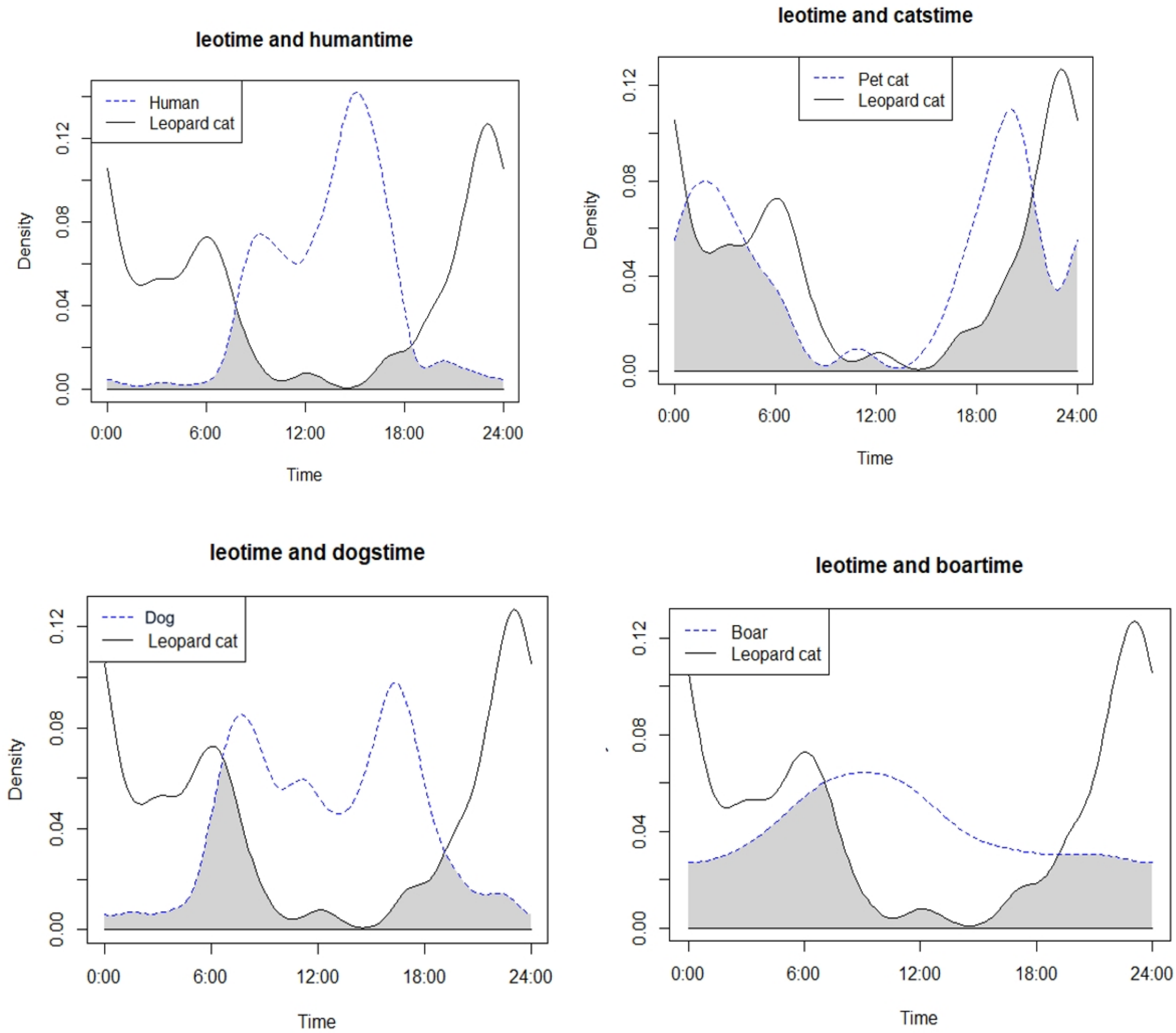


Figure 4.7: Temporal overlap between leopard cat and (a) human beings; (b) stray cats; (c) stray dogs; and (d) wild boars. Curves of change in activity levels for each species over a 24-hour period were plotted. The activity curves for humans, cats, dogs, and pigs were overlaid with those for leopard cats.

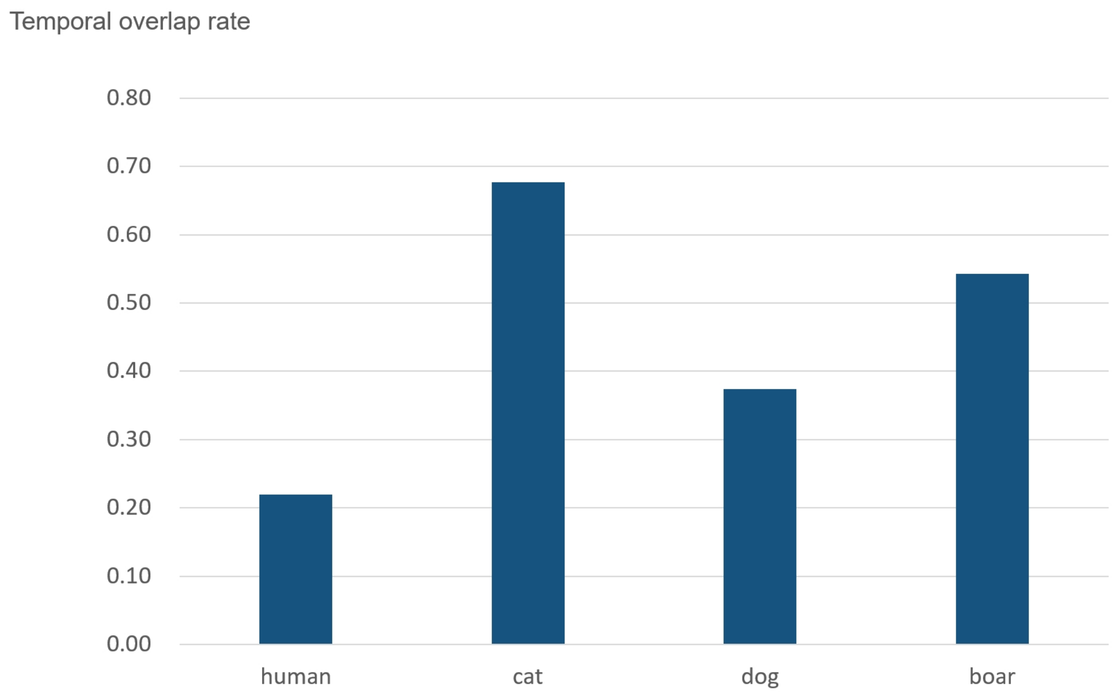


Figure 4.8: Temporal overlap rate between leopard cat and four species: human beings, stray cats, stray dogs, and wild boars. Temporal overlap coefficients between two species were estimated using the ”overlaps()” function in the ”overlap” package.

4.5 Discussion

The findings of this chapter suggest that the shrinking size and increasing fragmentation of urban habitats increases the spatial and temporal overlap of species with those of similar ecological niches (Haddad, 2015; Zanni et al., 2021). Similar findings were reported in a study from West Bengal, India: as anthropogenic disturbance (human settlements) increased, so did the overlap in the ecological niches of the Indian grey wolf (*Wolf indica*) and striped hyena (*Hyaena hyena*), and such disturbance was one of the most important predictors of ecological niche overlap for both species. Together with the results of this thesis and those of the broader fragmentation literature (Jacobson, 2019; Haddad, 2015; Digiulio, 2009), these findings suggest that an often overlooked impact of urbanisation is the increased spatial and temporal overlap of the native species that are able to persist in cities, which potentially puts species at even greater risk of extirpation. This overlap is particularly damaging to native wildlife when domesticated or exotic species, which are often better adapted to urban conditions, are involved (Banks, 2015; Doherty, 2017).

The spatial overlap between human activities and the distributions of stray cats, stray dogs and wild boars, as well as leopard cats, was found to be over 60%. This result is similar to the results of studies in Utah: in less urbanized areas, the overlap rate between coyotes and raccoons was 60%, while in more urbanized areas, the overlap rate between the two species reached 70% (Lourraine, 2002; Green, 2022). The results presented here, in combination with these other studies, indicate that habitat fragmentation due to urbanization compresses species' distributions, increases the spatial overlap of species distributions, and potentially intensifies the interactions and competition between species. Almost all areas where leopard cats are present in Shenzhen overlap with human activities.

It was also found that both leopard cats and stray cats have peak activity during nocturnal and dusk periods (George, 2006). However, there is a time difference between their peak activity in the city. Stray cats tended to be active 2-3 hours earlier than leopard cats. Therefore, I suggest that habitat loss and fragmentation due to urbanization have intensified competition between stray cats and leopard cats, causing them to stagger their activities to avoid competition for food and space. Humans and stray dogs have a limited impact on leopard cats because their daily habits differ from leopard cats (Grinder, 2001; Grassman, 2000). Unlike most wild boar activity pattern studies conducted in protected areas, I also found that wild boar had increased nocturnal activity in urban habitat patches (Young, 2019). This may be due to frequent human recreational hiking activities, which made wild boar less active during the day. In particular, wild boar activity decreased during the afternoon from 12:00-18:00, when human hiking activities are widespread, compared to the morning period.

The results show that wild boar and leopard cats are more widespread in the S1 patch. The probable reason for this is that the Shenzhen government established the Shenzhen Special Economic Zone (SEZ) border ”Second Line Pass” in the central part of Shenzhen from 1985 to 2010. S1 and S4 patches coincide with the location of this SEZ border. During these 25 years of rapid urbanization in Shenzhen, the S1 and S4 patches, which were the boundary of the Special Administrative Region, were off-limits, which may have been the basis for wildlife recovery. Thus, native habitat closure is an indispensable tool for city biodiversity restoration.

4.5.1 Conservation suggestions

The results of the study also confirm the negative impacts of exotic carnivores (e.g., stray cats and stray dogs) on wild carnivores (Doherty et al., 2017; Zapata-Ríos and Branch, 2016). Therefore, controlling the number of stray animals in urban habitats is recommended, mainly through sterilization and laws, to reduce the abandonment of stray animals (Bonacic et al., 2019; Contreras-Abarca and Simonetti, 2023). These measures reduce competition with wild predators and directly reduce predation by stray cats on small vertebrates such as birds, which is conducive to conserving urban biodiversity as a whole (Trouwborst et al., 2020; Loss et al., 2013). However, these policies are not without difficulties, including public opposition (e.g., on animal welfare grounds) and financial costs (Contreras-Abarca and Simonetti, 2023).

In addition, this study revealed that the rate of spatial overlap between humans and carnivores has increased due to urbanization; therefore, controlling access for recreational users is also necessary. It is important to urge people to stay on designated pathways, leash pets, and refrain from harassing wildlife. It should be forbidden to use off-road motorized vehicles inside the area.

4.5.2 Limitations

This study has the following two main limitations: 1) to prevent the camera traps from being stolen by others, I set the camera 1-2 m away from the main hiking trails and hid them behind tree trunks to avoid people's sight. This could have resulted in the actual number and amount of human activities captured being much smaller than the true values, and 2) simple kernel density estimates do not consider the role of environmental variables in driving species distributions. However, as shown in the results section of Chapter 3, my SDM analyses, including standard environmental predictors, resulted in low AUC values and thus, this method is not suitable for this species in this study system, potentially due to the strong effect of human actions.

4.5.3 Future study

Future studies could use GPS collars to accurately investigate the range of leopard cats and the overlap with the distributions of other species, such as stray cats. Faecal surveys should be conducted to determine the dietary habits of leopard cats and carnivores with similar ecological niches and to assess their competitive relationships further. A long-term investigation of the spatial and temporal overlap rate of the species should be carried out to determine the changes in the overlap rate with the advancement of urbanization. Finally, it should be judged whether the change in the overlap rate has a positive or negative impact on the population size of the leopard cat and to what extent.

4.6 Summary and synthesis

In conclusion, this study investigated the overlap of carnivores, human activities and feral pigs in urban habitat patches in Shenzhen. It was found that there was more overlap between leopard cats and stray cats and that leopard cats would avoid the space where stray cats were active and the busiest time in a limited space. Leopard cats would avoid areas of human activity. Compared with wild boars in protected areas, wild boars in urban patches have increased behaviour at night and decreased frequency of activity in the afternoon. It is recommended that people control the number of stray cats and stray dogs in urban patches and take measures such as de-sexing and adoption. In addition to hiking and playing, human activities that are destructive to leopard cats, such as poaching, are prohibited. Connect mutually independent habitat segments to expand the activities or hiding range of the species. Be aware of the presence of wild boars during morning walks to avoid unnecessary danger.

Chapter Five

Connecting habitat patches: potential leopard cat ecological corridors and their costs in Shenzhen

5.1 Abstract

Urbanization leads to habitat isolation, impedes gene flow, increases roadkill events, and threatens carnivores. Ecological corridor studies are needed to restore connectivity between habitat patches. However, most previous corridor studies have focused primarily on structural connectivity and consider little about functional connectivity, i.e., the range of wildlife movement. This chapter aims to assess the location of potential ecological corridors in Shenzhen. I simulated a potential leopard cat ecological corridor in the city (and conducted a cost analysis) using a minimum cumulative resistance (MCR) model by combining the distribution density of leopard cats and environmental factors such as topography and human and social factors. Ultimately, I proposed 118 potential ecological corridors in Shenzhen to aid the migration of leopard cats. From this pool, the 23 most important ecological corridors

were identified. The average width of the best corridors was 727 m, with a total corridor area of 97.7 km^2 and an estimated total cost of RMB 55.74 billion. The significance of this study is to provide a theoretical basis for guiding the government's decision-making. Ecological resilience can be achieved in densely populated places by delineating suitable areas for ecological corridors, which can serve as geographical guidance for their eventual implementation. This chapter examines the techniques for locating prospective biological corridors and calculating the costs of redevelopment, laying the groundwork for implementing useful land use changes that take ecological appropriateness and economic rationalization into account. Finally, it is important to understand the hotspots where wildlife is most active in cities, as well as the crucial connecting routes that these animals use via the urban matrix. Create fences that point animals in the direction of buildings. When rebuilding corridor habitat, plant folio species and pay attention to the vegetation quality.

5.2 Introduction

With increasing urbanization, ecological land is being encroached upon by expanding urban construction (Fahrig and McGill, 2018). The resulting habitat fragmentation poses a significant threat to biodiversity in cities by creating barriers to gene flow and individual exchange between different habitat patches (Cheptou et al., 2017; Wu, 2010; Wang and Dun, 2015; Liang et al., 2019). However, some studies have confirmed that establishing ecological corridors between urban habitat patches can help enhance population viability and biodiversity (Fischer and Lindenmayer, 2002; Gilbert-Norton et al., 2010). Ecological corridors increase the habitat area for urban wildlife and enable connected meta-populations that result in an increased rate of gene exchange and, thus, the probability of inbreeding decreases. Movement, energy flow and information exchange between patches all accelerate when corridors are present in the landscape (Albert et al., 2017). Trees or shrubs in ecological corridors facilitate the movement of wildlife, particularly in the presence of anthropogenic disturbances, by providing cover. Because of the small size and number of habitat patches in highly urbanized areas, ecological networks must be planned to restore connectivity (Gavrilidis et al., 2019). To achieve this goal, it is imperative to identify the location and spatial extent of potential ecological corridors in cities (Keeley et al., 2018).

One thing to consider when restoring connectivity between habitat patches is whether the connectivity is structural or functional. Structural connectivity means habitat corridors are constructed by considering only environmental factors, including topography and land use type, meaning whether a corridor can be constructed on the site. (Vogt, 2009) The likelihood that wildlife will use the corridor is not considered. Functional connectivity, on the other hand, considers the spatial distribution of wildlife and their trajectories when designing a corridor, and the probability of wildlife using the corridor at that location is added as a weighting factor in the model construction (Vogt, 2009; Mimet, 2013; Stewart, 2019). In this study, functional connectivity will be realized by introducing the spatial distribution of leopard cat occupancy probability into the connectivity study in conjunction with the camera trapping data in Chapter Three.

Research conducted in England has revealed that the establishment of connections between forest fragment clusters in northern England and those in southern Scotland has resulted in notable genetic intermingling of squirrel populations from Scotland and Cumbria, even at distances of up to 100km from the New Forest site. The findings of this study provide valuable insights into the conservation of plant and animal species in fragmented environments, such as the United Kingdom (Ogden, 2005). Research conducted in Australia has revealed that the utilisation of corridors located distant from forests by various species is comparatively lower than the utilisation of corridors and forest patches in close proximity to forests. Various species exhibit distinct patterns of corridor utilisation. The study findings indicate that there was a higher number of males observed in corridors compared to forests.

Additionally, it was observed that individuals of both sexes had lower body weights in corridors as compared to those in forests. This study posits that corridors have the potential to serve as valuable habitats for mammal groups, while they may not comprehensively mitigate landscape fragmentation (Downes, 2002).

Nevertheless, a significant number of corridors are deemed to lack practical utility. In accordance with national forest regulations, developers operating in the Brazilian Amazon are mandated to preserve forest strips measuring 60 metres in width along the sides of rivers. Nevertheless, a comprehensive examination of avian and mammalian species within these corridors revealed that any width below 400 metres did not provide advantages to riparian species populations and did not effectively facilitate gene transfer (Gregory, 2021). Research conducted in Queensland, Australia, determined that the creation of corridors had limited efficacy in facilitating the recovery of biodiversity. The implementation of wildlife corridors and the subsequent enhancement of landscape connectivity would lead to heightened levels of gene flow among species residing in remaining regions. Nevertheless, the genetic findings of this study do not align with this hypothesis, as both genetic markers indicate that populations residing in residual patches connected by corridors exhibit dissimilarities, unlike populations within patches that are entirely isolated by an inhospitable matrix. The observed substantial divergence across populations of *Mycobacterium cervicornis* (Trouwborst, 2020) indicates that there is likely minimal gene flow occurring among these populations inside residual patches, regardless of whether corridors connect them or not (as stated). According to a study conducted in Malaysia, corridors contributed to a more rapid decline in biodiversity due to their facilitation of poaching activities (Lacerda, 2009).

Many studies have explored the location of potential ecological corridors in mega-cities, but only a small number have scientifically analyzed the most effective corridor widths (P. Li et al., 2022). However, determining the width of corridors is an essential indicator for developing and implementing government ecological restoration policies. The effective width of a corridor depends on many factors, including the target species, the land available for construction, and the government budget. The width of the corridor plays a crucial role in the conservation of urban biodiversity; if it is too narrow, it will be overly negatively affected by human activities and not conducive to wildlife passage, but if it is too wide, the financial cost will increase significantly (P. Li et al., 2022). Therefore, ecological corridors need to be of appropriate width. Zhu proposed setting buffer zones around potential corridors and determining the final width of corridors by setting thresholds (Zhu, 2012). However, such a buffer zone is usually of fixed width, which does not correspond to the actual realities of urban ecological construction.

Land cover in cities is heterogeneous, so determining the location and width range of potential ecological corridors is a challenging task, and determining the effective width of the corridors is a top research and urban conservation priority. In this paper, I take the mega-city Shenzhen as the study site and the leopard cat as the focal species and use the MCR method to identify the best locations of potential corridors and their ideal widths, given both ecological and financial considerations. The objectives of this study are: 1) to select suitable ecological core areas in Shenzhen; 2) to identify the location and number of potential ecological corridors in Shenzhen; 3) to prioritize the construction of different ecological corridors; 4) to establish a systematic method for determining the width range of ecological corridors; 5) to calculate the cost of ecological corridor construction and to provide theoretical support for the construction of urban ecological corridors. By achieving these objectives, this study will provide a functional perspective on the connectivity question in urban areas and focus on a key research gap in the connectivity literature: the question of effective corridor width.

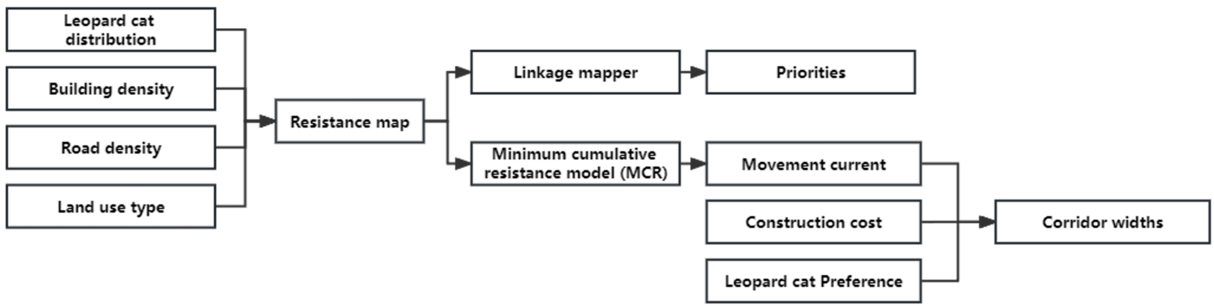


Figure 5.1: Methods flow chart for Chapter 5. The main methods used in this chapter and the sequence of method application.

5.3 Materials and methods

5.3.1 Method structure

The research methodology of this paper can be divided into four parts: 1) Combining ecological quality data and leopard cat density data to calculate the movement resistance surface layer of the Shenzhen leopard cat; 2) Identifying potential corridor paths using the Linkage Mapper tool; 3) Evaluating the prioritization of ecological corridor construction using the Linkage Mapping tool; and 4) Evaluating the trade-offs between the corridor widths and the economic costs to demarcate the ecological corridor areas suitable for the migration of the Shenzhen leopard cat (Figure 5.1)

5.3.2 Core areas

To study the location as well as the width of the optimal leopard cat ecological corridor in Shenzhen, I selected the six habitat patches in Chapter 3, as well as other patches within

30 km of the surrounding area with an area larger than 1 km^2, and continuous habitats in northern and eastern Shenzhen. Large core patches are irreplaceable for restoring leopard cat populations, so I prioritised large habitats for connectivity.

5.3.3 Building resistance surface

To find the potential ecological corridor construction area, it is necessary to calculate the minimum resistance area for leopard cats to pass through the surface. The resistance raster reflects the possibility or willingness of animals to pass through a particular area. The lower the resistance, the stronger the willingness of leopard cats to choose the place to pass through, and the more suitable the place is for ecological corridor construction.

This study built the resistance surface by considering three main factors: leopard cat density, habitat unsuitability and NDVI. Leopard cats are the focal species for my research because they are one of Shenzhen's only remaining top predators. They can act as an "umbrella species" to protect other small species in urban habitats. Therefore, the selection of ecological core patches, the setting of resistance values, and the determination of optimum corridor width are all related to the characteristics of leopard cats. First, I used the probability of leopard cat distribution obtained by the kernel density estimation method in Chapter 4. The denser the distribution of leopard cats, the higher the probability of use and potential passage rate of leopard cats to that part and the surrounding habitat, and the greater the need to build ecological corridors in that area. The natural breakpoint method (Jenks, 1967) was used to classify the distribution probability of leopard cats into ten levels. The smaller the level, the greater the resistance. The following environment variables have the same resolution of 250m x 250m and were analysed using the same grouping method.

Traffic accidents have been ranked as the leading cause of carnivore mortality (Bateman, 2012). Therefore, roads were selected as the greatest threat to the movement of leopard cats in this study. The length of the road within each $1km^2$ was calculated in ArcGIS using the method of line density estimation. The higher the road density, the higher the surface resistance. The shapefile data of building density in Shenzhen was downloaded from Beijing and was used for kernel density estimation. The higher the building density, the higher the surface resistance. Overlapping road density and building density according to the same weight constitute the 10-level resistance value of habitat suitability. The smaller the corresponding resistance value, the higher the biodiversity and the higher the ecological suitability (Zhang and Song, 2020).

In addition, animal migration depends heavily on vegetation density (Tremblay and St. Clair, 2011). Therefore, the Normalized Difference Vegetation Index (NDVI) was also used to assess the resistance for leopard cat passage. NDVI is an indicator of vegetation cover and growth status, with scores ranging from -1 to 1. Negative values indicate that the surface is covered by water, clouds or snow, 0 indicates bare soil or rock, and positive values indicate vegetation coverage (Saura and Pascual-Hortal, 2007). NDVI was classified into ten levels using the natural breakpoint method. The higher the level, the higher the resistance value. In the above process, I obtained three resistance variables of habitat quality, leopard cat density and NDVI, each resistance variable ranging from 1-10. Then, I used the analytic hierarchy process (AHP) to determine the weight of each resistance variable. Finally, I used the ArcGIS tool "Weighted overlap" to weigh the above three variables when overlapped to obtain the final resistance raster data.

5.3.4 Identify the least-cost path through the resistance surface

The minimum cumulative resistance model (MCR) was used to simulate the process of leopard cats crossing different landscapes to find potential corridor locations. The calculation equation is as follows:

$$MCR = f\min \sum_{j=n}^{i=M} D_{ij} \times R_i \tag{5.1}$$

Where MCR is the value of minimum cumulative resistance; f is the positive correlation between minimum cumulative resistance and ecological processes; D_{ij} is the spatial distance of species from core patch j to landscape unit i; R_i is the resistance coefficient of landscape unit i to species movement. The area with the least resistance has good ecological quality, so I designated the least-cost path as a potential ecological corridor. I used ArcGIS's supplemental toolbox, ”Linkage Mapper”. The network adjacency method of ”Cost-Weighted” was selected when creating the “Network of Core Area” , limiting each patch to a maximum of two other nearest patches. ”Cost -Weighted” was also used to identify the nearest Neighbor.

5.3.5 Priority of corridors

Since there are many potential corridors and the time and money available for their construction is limited, the most important corridors for overall connectivity need to be selected and prioritized for construction. The “Centrality Mapper” tool in “Linkage Mapper” can show the importance of each linkage in maintaining the overall network connectivity. Thus, it can be used to indicate the importance of different ecological corridors. Here, I classified the different links into high, medium and low importance, which can be used to help the government prioritize the construction of ecological corridors.

5.3.6 Corridor width

Width is the most controversial indicator when building ecological corridors. The wider the width of an ecological corridor, the better it is for wildlife migration through different habitat patches. However, the wider the ecological corridor, the larger the built-up area contained in it and the greater the economic cost of removing the built-up area and constructing the ecological corridor. Therefore, the economic cost of constructing ecological corridors should be carefully considered when determining corridor widths.

Narrow corridors are more vulnerable to human activities (Beier, 2019). Many studies have shown that leopard cats' most suitable ecological corridor width is around 1 km. Therefore, in this study, I mapped buffer areas on both sides of the identified corridor paths with buffer widths of 250 m, 750 m, and 1250 m, and the adaptation levels are 10, 8, and 5, respectively.

According to circuit theory (Dickson, 2019), high current density areas can be identified by simulating the current intensity as it passes through the resistance surface. These areas, known as ecological ”pinch points”, are relatively important in the corridor. To determine the location of the ecological ”pinch points”, it is first necessary to ”ground” one ecological core area, input the same ”current” to other ecological core areas, and calculate the cumulative current value of each pixel. Because of their importance to the current flow, the degradation or loss of "pinch points" is very likely to cut off the connection between ecological core areas. These areas should thus be prioritized for conservation actions (Kong et al., 2021). "Circuitscape" and "Pinch Point Mapper" were used in this study to determine the simulated current intensity distribution and areas of high current density in the corridor. Because it has been found that the width of the corridor does not affect the location of ecological pinch points (Peng et al., 2018b), I set the width of the ecological corridor to 10,000 m. I calculated the current for each pixel using the pairwise model. The current values were graded according to the natural breakpoint method into adaptation levels from 1 to 10.

To calculate the construction cost of an ecological corridor, one first needs to analyze the area and cost of the different land use types within the corridor. Of these, buildings and roads are the most costly types of use to restore, and because many urban non-volant mammalian wildlife cannot cross water, water is the greatest impediment. Buildings represent strong anthropogenic disturbance; therefore, most urbanized areas have a high associated cost. In this study, the cost of constructing ecological corridors was reallocated to different land-use types using the reclassification tool of ArcGIS. Level 10 was assigned to leopard cat habitats (forests, grasslands, shrubs, and wetlands), 8 to agricultural land, 4 to urban land, and 0 to water bodies; the higher the adaptation level, the higher the suitability and (roughly) the lower the cost of construction.

The traditional method of width determination is to build buffer areas of different widths around potential corridors, but this has the disadvantage that the corridors are of equal width everywhere. This approach is certainly not conducive to restoring maximum connectivity at minimal cost. Therefore, I overlapped width, current, and land-use types adaptation levels in my analysis. The higher the level, the narrower the width, and the lower the cost.

5.3.7 Financial cost

In this study, the cost of ecological corridor reconstruction was estimated quantitatively. Firstly, it was necessary to analyze the land-use type change within the potential ecological corridor area, which forms the basis for analyzing the spatial location of land-use adjustment and calculating the transformation cost. Depending on the land-use type classification, certain habitats (including woodlands, grasslands, and shrublands) have high ecological suitability and can be maintained as they are without restoration. The types of land that should be restored for ecological use include agricultural and built-up land. The built-up land can be subdivided into urban, rural, residential, and industrial land. Through the above analysis of potential ecological corridor suitable areas, I identified the corridor locations with non-constant widths. Land-use information of each potential corridor was obtained by overlaying the potential ecological corridors of different widths with the land-use data created in 2021.

Corridor construction costs will vary depending on the composition of the land-use types within the identified corridor location (P. Li et al., 2022). As such, the area of land-use types within the different corridor widths identified through my analysis was calculated. The cost of constructing ecological corridors includes the reduction of benefits from non-ecological land and expenditure associated with adjusting land-use types. The cost of undeveloped land adjustment was calculated as 166.65 Yuan/m^2 according to the document ”Compensation Standard for Land Acquisition in Shenzhen” (Shenzhen municipal government, 2018), and the cost of resettlement and relocation of rural settlements was calculated as 2230 Yuan/m^2. Urban land adjustment cost was calculated as 3210 Yuan/m^2 using the auction price of urban land in Shenzhen in 2021. The auction price of industrial land is 1500 Yuan/m^2 (Shenzhen Natural Resources Assets Market, 2022, https://pnr.sz.gov.cn/d-sznram/szmarket/#/home). The inclusive cost is calculated based on the area of different land-use types.

5.4 Results

5.4.1 Core areas

Ninety-two ecological core areas of 1292.9 km^2 were selected, with a spatial distribution centred in the north and southeast of the region. The southern and western CBD areas are densely populated and built, with a high degree of landscape fragmentation and a scattered, small number of ecological core areas. Although there are several municipal parks, human interference is substantial, and using them for ecological corridor construction is challenging for my target species. The northern and southwestern parts of the region are less densely populated and have more extensive areas of forests and other habitats. There are eighteen core areas larger than 10 km^2, accounting for 88.3% of the total area of the landscape. There are 67 core areas smaller than 3 km^2, accounting for 8.2% of the total ecological landscape area, which could be used as ecological stepping stones. The average distance between habitat patches in Shenzhen is 308.4 m. The minimum is 25 m, and the maximum is 1796.4 m.

5.4.2 Minimum cumulative resistance (MCR)

The total resistance value (Figure 5.2 a) was affected by leopard cat density, land-use type, road density, building density, and Normalized Difference Vegetation Index (NDVI). The maximum resistance value is ten and was mainly concentrated in overcrowded urban land. Parks and forests, as expected, have significantly lower resistance values than the surrounding areas. The resistance distribution map shows that the northern habitat patches have less resistance between them and the surrounding contiguous habitats, and there is a greater possibility of connecting these areas. Some agricultural land and urban parks exist between patches, which can be used as stepping stones to assist the movement of leopard cats between patches.

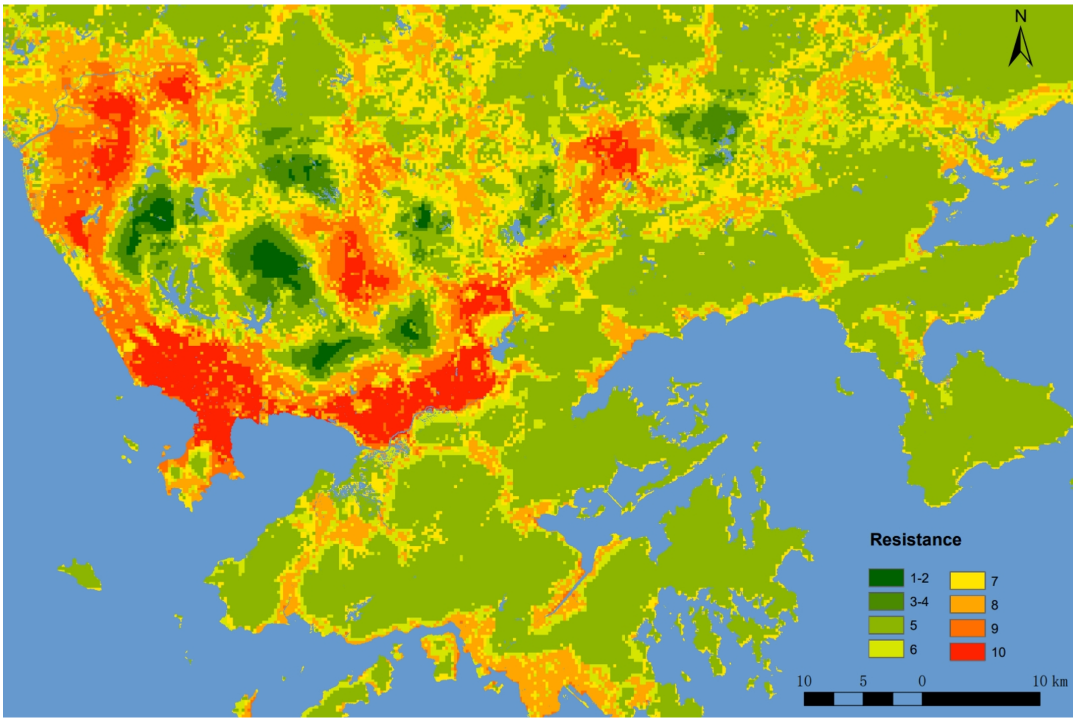


Figure 5.3: Shenzhen ecological resistance surface. The weights of each resistance variable were determined using hierarchical analysis (AHP). Finally, the ArcGIS tool "Weighted Overlap" was used to weigh the above three variables when they overlapped to obtain the final resistance raster data.

Building density (Figure 5.2 b) is high in the southern CBD area only, while road density (Figure 5.2 a) is high in the south, central and northwestern parts of the city. The distribution range of leopard cats (Figure 5.2 d) is concentrated in the central and southern patches, and the density of roads and buildings around these patches is high. Thus, the movement, feeding, mating, and survival of leopard cats are seriously threatened here.

Ecological corridor extraction in urban areas is essential to restore the effective migration and dispersal of leopard cats and other species. Based on the MCR model, I identified a

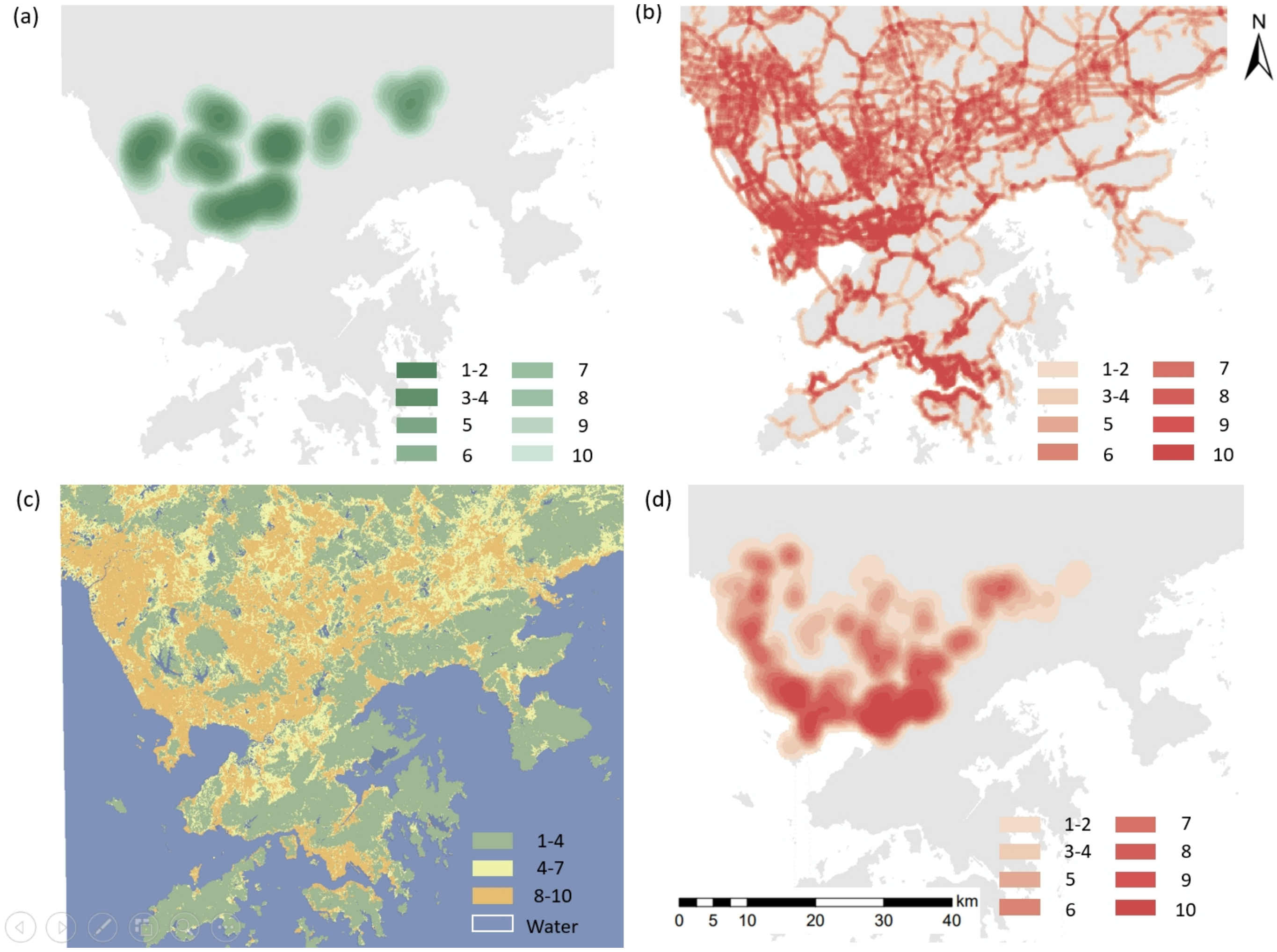


Figure 5.2: Shenzhen ecological resistance factors: (a) leopard cat density resistance surface; (b) road density resistance surface; (c) land-use type; (c) land-use type resistance surface; (d) building density resistance surface. The probability of leopard cat distribution was obtained using the kernel density estimation method in Chapter IV. The road density, building density and land-use type data of the city were downloaded, and the values were classified into ten levels using the natural breakpoint method. The higher the level, the higher the resistance value.

total of 186 potential ecological corridors in Shenzhen (Figure 5.4). The average length was 1.9 km, and the longest was 11.1km. There are 89 patches connected, with an average of 2.11 corridors per patch. Patch 14 has the most corridors, with five corridors connected. There are 81 corridors less than 1.2 km and only 20 ecological corridors greater than 2km, indicating that the ecological core areas in the southern part of Shenzhen are closely adjacent. These corridors are mainly located in the northern part of Shenzhen, as many contiguous ecological core areas are here. The southern part is overcrowded with urban land, and the highly urbanized areas have high migration resistance and few potential ecological corridors. The number of potential ecological corridors in the south is minimal (average 880.4 m, maximum 3603 m long, and minimum 250 m short). My results suggest that Shenzhen’s small ecological core areas are of high value as connection nodes.

5.4.3 Priority of corridors

A centrality analysis of ecological cores and corridors was conducted to prioritize the construction of corridors and restore connectivity between essential core areas. Twenty-three corridors were identified as primary, twenty were secondary, and seventy-five were tertiary (Figure 5.4). The primary corridors play a vital connectivity role in the migration and dispersal of leopard cats and should thus be the highest priority for construction. Although the overall current density of the tertiary corridors is relatively low, they still play an essential role in the migration of leopard cats and should not be neglected off-hand.

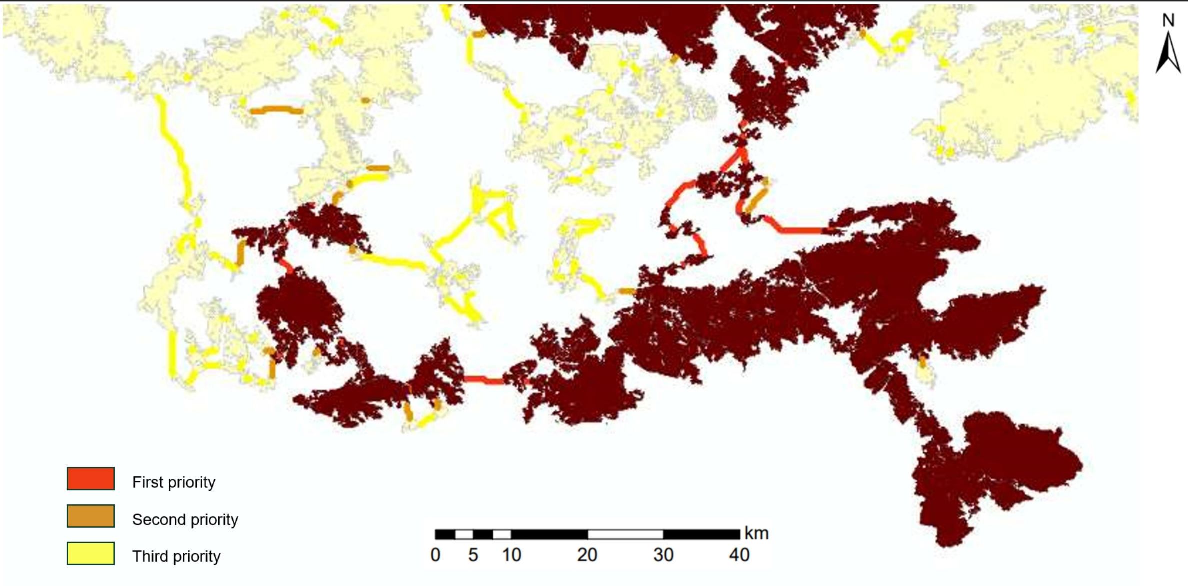


Figure 5.4: Centrality rankings of potential ecological corridors and core areas for overall network connectivity.

5.4.4 Corridor width

The current density distribution of the proposed urban ecological corridor network in Shenzhen is shown in Figure 5.5 b. I classified the parts of the current density greater than level 7 as ecological pinch points. The corridors with the highest current density are located in the south and east of the city. They are the solitary corridors connecting the heavily fragmented areas in the west and the continuous forest areas in the east.

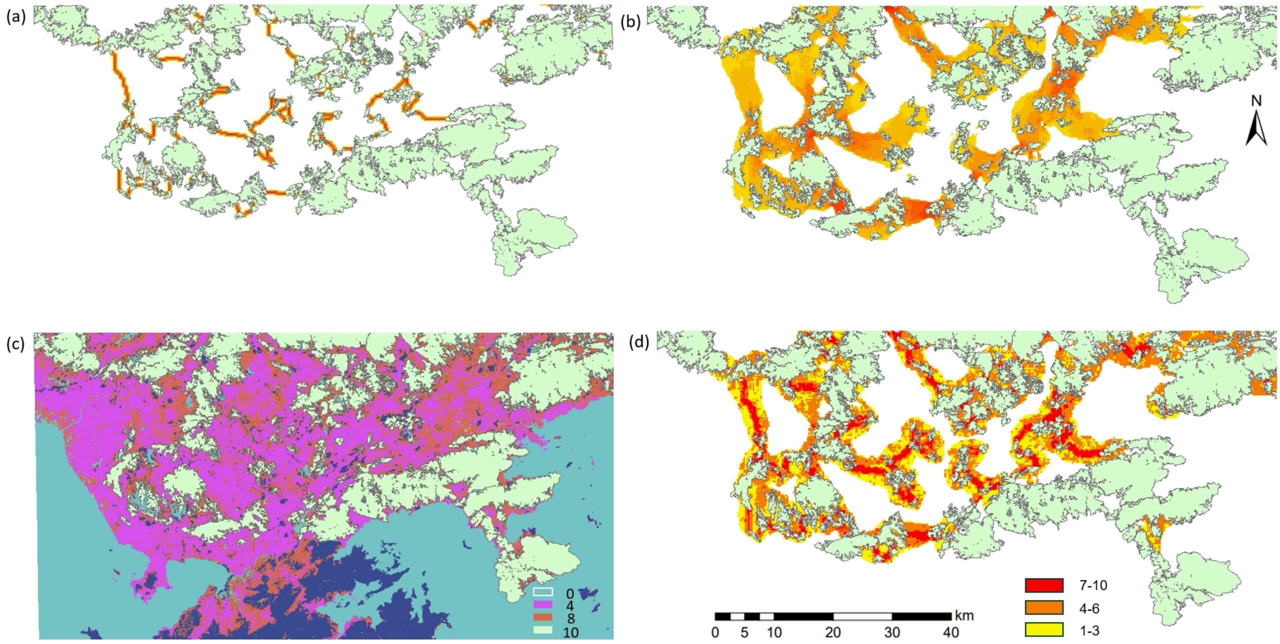


Figure 5.5: Adaptation level of ecological corridor construction (a) The suitable width for leopard cat migration; (b) Areas with high current density; (c) Areas with lowest restoration cost; (d) Shenzhen ecological corridor construction adaptation level map. With the lowest restoration cost; (d) Shenzhen ecological corridor construction adaptation level map. The width, current and land use types were categorised into 1 to 10 adaptation levels based on the natural breakpoint method. The higher the level, the narrower the width and the lower the cost. Finally, the three adaptation levels were overlapped.

The change in corridor width and area with increasing levels is shown in Figure C.1. From level 18 to level 14, the average width of the corridor increases from 571.51m to 1243.69m, and the area covered by the corridor increases from 59.25 km^2 to 128.94 km^2. The land-use types covered by corridors of different widths differ. For corridors of level 18, 16.1% is agricultural land, 36.2% is urban land, 0.74% is rural settlements, and 6.4% is industrial land. For corridors of level 14, 12.85% is agricultural land, 46.15% is urban land, 0.63% is rural settlements, and 6.11% is industrial land. The wider the corridor, the larger the proportion of the urban land area encompassed, as expected.

5.4.5 Financial cost

The change in corridor construction cost with the increase in corridor width is shown in Table 5.2. The construction cost of the corridors increases from 55.74 billion yuan (the narrowest) to 135.408 billion yuan (the widest). The proportion of urban land cost to total cost increased from 36.2% to 44.8%, significantly increasing the total costs. These increases are due to more urban land use within wider urban corridors. While wider corridors are better at maintaining ecological processes and restoring biodiversity, high construction costs associated with wider corridors make it more challenging to create these buffers. Urban land adjustment dominated the building costs for all corridor widths. These results suggest the need to control urban sprawl and restore municipal land to ecological land. The larger the area of non-ecological land (e.g., urban land and cropland), the higher the cost of land-use adjustment and the greater the expenditure required for ecological corridor construction.

The total cost of ecological corridor construction, with a score of 14, is 135.408 billion CNY yuan (November 24, 2023, 1 CNY Yuan was exchanged for 0.1405 USD, People's Bank of China, 2023), owing to the urban land expansion (Table 5.2). The highest cost is the municipal land adjustment of 89.25 billion yuan, accounting for about 46.15% of the total cost. The cost of acquiring other non-ecological land is much lower. The total cost of ecological corridor construction, with a score of 18, is 55.744 billion yuan (Table 5.2). The highest cost is the borough land adjustment of 32.156 billion yuan, accounting for 36.18% of the total cost. This indicates that the wider the corridor, the higher the cost of urban land adjustment.

Level	Crop		Urban		Rural		Industry		Total (km^2)
	Area (km^2)	percentage	Area (km^2)	percentage	Area (km^2)	percentage	Area (km^2)	percentage	
18	9.6	27.10%	21.4	60.80%	0.4	1.20%	3.8	10.80%	35.3
17	12.2	25.60%	30.3	63.70%	0.7	1.40%	4.4	9.20%	47.6
16	13.9	21.40%	44.1	68.10%	0.8	1.20%	6	9.30%	64.7
15	15.1	20.40%	51.1	69.10%	0.8	1.00%	7	9.50%	74
14	17.1	20.40%	57.8	68.90%	0.9	1.00%	8.1	9.70%	83.9

Table 5.1: Total land-use type area and structure of different ecological corridor widths (billion RMB yuan). By overlaying the potential ecological corridors of different widths with the land-use data in 2021, the percentage of land-use area of each potential corridor was obtained.

Level	Crop		Urban		Rural		Industry		Total (billion \$)
	Price (billion \$)	percentage	Price (billion \$)	percentage	Price (billion \$)	percentage	Price (billion \$)	percentage	
18	1.6	2.90%	32.2	57.70%	9.8	17.50%	12.2	22.00%	55.7
17	2	2.60%	45.5	59.10%	15.3	19.90%	14	18.30%	76.9
16	2.3	2.20%	66.1	63.30%	16.7	16.00%	19.3	18.40%	104.4
15	2.5	2.10%	76.7	64.80%	16.7	14.10%	22.5	19.00%	118.4
14	2.9	2.10%	86.6	64.10%	19.5	14.40%	26.1	19.30%	135.1

Table 5.2: Total investment and structure of different ecological corridor widths (billion RMB yuan). Comprehensive costs are calculated based on the area and construction costs of different land use types.

5.4.6 The optimum corridor

Based on the aforementioned results, I choose the optimal width level of 18, with an average width of 726.96 meters, the narrowest width of 250 meters, and the widest area of 1500 meters, with a total corridor area of 97.74 km^2 and a total cost of 55.74 billion yuan. Based on the results of my model, the construction of ecological corridors in Shenzhen will regenerate 59.25 km^2 of habitat, and the fragmented habitat in Shenzhen will be restored to 880.56 km^2 of continuous habitat. These ecological corridors will connect 91.37% of the original habitat area in Shenzhen. The percentages of area and construction costs of different land use types in corridors of different widths are listed in the table 5.2. The percentage of the area of ecological land (habitat and cropland) is 53.9%, which indicates that the ecological corridor designed in this study utilizes the existing ecological land as much as possible and reduces the construction costs. In contrast, non-ecological space (urban land, rural settlement and industrial land) accounts for 46.1% of the longitudinal extent of the corridor.

The width of most identified potential ecological corridors in western Shenzhen is relatively narrow (Figure 5.6), roughly 250 meters, due to the concentration of roads and buildings. Corridor B is the most important identified corridor, likely as it is the only corridor connecting the western urban centre in the contiguous eastern habitat (Figure 5.6). It is also the widest corridor, with a width of 1,311 m at its widest, and the area of urban built-up land contained within it is 2.75 km^2, with a percentage of 73.33%. The cost of construction associated with Corridor B is 4.13 billion yuan. Numerous short corridors are also scattered across the city. These corridors link adjacent patches divided by roads and, although short, are essential to improving the connectivity between Shenzhen's habitat patches.

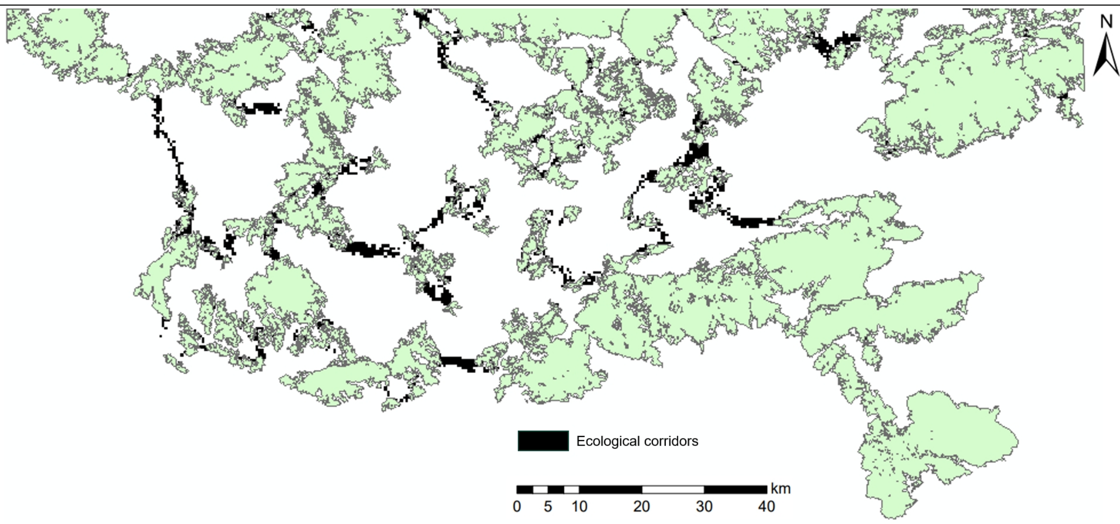


Figure 5.6: The best width of Shenzhen ecological corridors. Selection of the width among the different adaptive level thresholds allows for a connection with the lowest total financial cost.

5.5 Discussion

The results of the study indicate that the average width of the most suitable habitat corridors in Shenzhen is 727 meters, taking into account ecological and economic considerations. However, the width of the corridor in that Shenzhen corridor study was the same everywhere, which is inconsistent with the reality of urban corridor construction. In the Shanghai Minhang District ecological corridor study, the researchers used a similar width decision-making method as the study. They established a metric system and set four indicators: width suitable for animal migration, the maximum cost-weighted distance over which animals can migrate, areas with high mobility potential, and areas with maximum restoration opportunities (Yujie Wang et al., 2022). The trade-off between the width of urban ecological corridors and the cost of construction is a problem that every major city must face (Peng et al., 2017; Deng et al., 2021).

This dissertation also designed a multi-indicator metric system to determine the range of corridor widths at different locations in order to maximize connectivity between leopard cat habitat patches at minimal economic cost. Incorporating the leopard cat range (derived from its own field data) into the study of potential urban corridors allowed for a better measure of the ecological corridor ((Stewart et al., 2019; Salviano et al., 2021) functionality and connectivity for wildlife movement. The proposed corridor selection method averages three metrics: areas of biological fitness most likely to be utilized and corridor construction costs (i.e., buffer width, wildlife corridor water flow density, and land use type) to determine corridor widths based on fitness class. This avoids the limitation of equal widths for all corridors in traditional studies and reduces corridor construction costs while achieving the same connectivity goals. The information generated from this work deepens our understanding of urban carnivore corridor design in Asian cities and will help the local government manage biodiversity in Shenzhen in the future.

Rapid urbanization and dramatic land use change have led to landscape fragmentation and reduced landscape connectivity, which is a serious threat to biodiversity conservation (Fischer, 2007; Tarabon, 2019). Therefore, the location and width of ecological corridors must be identified, and the construction costs must be calculated. To this end, in this study, I selected the ecological core areas. I used the MCR method to identify the locations of potential ecological corridors in Shenzhen and ranked their priority of importance. Furthermore, the cost of restoring ecological connectivity was calculated according to different land use types. Overall, this study revealed that the length of potential corridors in urban Shenzhen was 103.7km.

A previous connectivity study in Shenzhen only focused on structurally connected ecological core areas, considering topography, vegetation, and other natural variables(Hong et al., 2017). In contrast to the structural connectivity between urban habitats, my study focused on the density of leopard cat distribution as a factor for corridor site selection. The higher the density of distribution of leopard cats in a particular area, the higher the weight associated with building an ecological corridor in that area. Such a corridor design strategy helps to maximize the restoration of leopard cats' dispersal among habitats. My study also considered urban road and building construction, as well as the distribution of leopard cats, which provides theoretical guidance for municipal ecological habitat connectivity and, ultimately, the restoration of ecological diversity. In regard to maximizing existing urban ecological space to reduce construction costs, I found many small habitat patches distributed between the larger patches, which might be utilized as "stepping stones" between the larger patches.

To determine the appropriate range of corridor widths to analyze, I focused on three indicators for the weighted averaging: the appropriate width for species migration, the current density of leopard cats through the resistance surface, and the type of land use(P. Li et al., 2022). First, the suitable width for leopard cat migration was selected based on previous studies (Jeong et al., 2018; Silmi et al., 2021). Secondly, potential corridor areas with a higher density of leopard cats are the most helpful for restoring the connectivity of the leopard cat, and these areas are also called "ecologically critical areas". They should be given priority in any connectivity strategy. Finally, it is essential to consider the benefits of building ecological corridors for wildlife more generally and the feasibility of implementing the project. Because the remaining available land in large cities is minimal (Ye et al., 2020) and the restoration cost varies by land use type, wider corridors are more beneficial for animal migration but may not be economically feasible.

While ecological corridors play a vital role in restoring habitat connectivity, the cost of constructing corridors can considerably strain local finances. Shenzhen's general public budget revenue in 2021 is RMB 488.96 billion, and the expenditure on ecological protection is only about RMB 12.51 billion (Shenzhen municipal government, 2021). Although the most suitable ecological corridor for leopard cat migration is about 1000 m (Juan et al., 1995; Peng et al., 2017), constructing a 1 km wide corridor in a densely populated city is prohibitively costly. The costs related to land use adjustments are much lower for narrow corridors because of the higher proportion of ecological land within the narrow corridor. The reconstruction cost of the 572 m wide corridor in Shenzhen is only 41.2% of the total cost reported to the total cost of the 1244 m wide corridor. Therefore, restoration of the 572-m wide ecological corridor is both ecologically appropriate and economically more straightforward. This conclusion is corroborated by the results of another urban corridor study in Nanjing (G. Li et al., 2022).

According to the Shenzhen Park City Construction Plan and Three-Year Action Plan (2022-2024), the Shenzhen government plans to connect biological habitats from west to east into a total of 330km of ecological corridors and to build 20 individual ecological corridors. In the three years from 2022 to 2024, seven corridors are planned to be restored, including one habitat restoration corridor, three upper-span animal corridors, one road corridor restoration, and two underpass animal corridors. In April 2020, the first wildlife protection ecological corridor in Shenzhen - "Paiyashan - Qiniang Mountain Node Ecological Corridor Bridge" in Dapeng District was built. The project covers an area of 132 hectares, with a span of 60 meters and a width of 50 meters. However, due to the construction density and other reasons, the role of restoring the connectivity of two isolated habitats has yet to be studied. However, this document does not spell out the corridor width, construction cost and specific construction timeline for the explicit construction of the ecological corridor. My study thus provides essential data, prioritization analysis, and cost analysis for the government's subsequent construction decisions.

5.5.1 Conservation suggestions

It is recommended that ecological corridors be constructed according to the plan presented in this chapter to restore connectivity between habitats and reduce the barrier and roadkill effects of roads on wild carnivores (Loro et al., 2015; Jackson and Fahrig, 2011). Ecological corridors must be constructed with consideration of their ability to facilitate wildlife migration, impacts on urban socioeconomic activities, and financial costs (**Soultan 2021**). More broadly, urban planners and managers must work with urban ecologists to better understand the hotspots where wildlife is most active in urban areas and the critical connectivity pathways these species use to move through the urban matrix (ideally through empirical research such as that conducted in this thesis).

To prevent animals from running into traffic and to steer them toward the crossing, buildings, fences, or other deterrents should be placed alongside highways and canals. Furthermore, fences ought to guide animals toward crossings rather than blocking their entrances (Bonacic, 2019). Animals are excluded from urban highways by fences, guardrails, and barriers that are at least two meters high. For example, a funnel-shaped fence boosted the number of species using culverts from 28 to 42 (Bonacic, 2019) and decreased road mortality by 93.5% in one study.

The quality of the new vegetation, particularly the choice of species to better preserve or re-establish the natural processes in the link, should also be considered while creating ecological corridors. The area's primary group-forming and dominant species of prospective natural vegetation, or folksonomia species, should be chosen. It is best to adopt planting techniques and configurations that resemble and imitate nature so that the native animals can identify it based on habitat, reproduction, and migration. The "landmark" plant communities in Shenzhen city must therefore be identified (Gregory, 2021; Olafsson, 2022).

5.5.2 Limitations

The limitations of this study are as follows: 1) Only a single evaluation criterion was used to select the core area, and only the area of the patch was used to judge whether the patch has ecological connectivity value. It is thus possible that my analysis missed habitat patches that are relatively small in size but in important locations in terms of spatial connectivity; 2) Only the distribution of a single species (leopard cat) was considered when analysing functional connections in the landscape. Although leopard cats can serve as an umbrella species for other species (Silmi et al., 2021), there may still be some species that are not well represented by the patterns of leopard cat distribution and dispersal; 3) Only three fundamental indicators were considered in determining the width of the corridors, and future studies should identify a more comprehensive system to analyze the mechanisms that determine the ideal width of ecological habitat corridors; 4) Finally, I did not fully consider whether these corridors aid the movement of leopard cats between habitat patches or whether they make such elusive animals more exposed to human influence, for example, by making poaching easier. 5) Lastly, the land use type in the study is only a simple vegetation-construction land classification. However, more detailed information on the vegetation, such as the type, quality, density, and material production index, was neglected. The quality of vegetation plays an essential role in selecting ecological corridors.

5.5.3 Future study

Future research will need to examine the feasibility of ecological corridors and specific construction plans, such as the efficacy of building an ecological bridge to reduce the impact of roadkill. In proximity to the corridor already constructed in Shenzhen, further study is needed on whether leopard cats, as well as other animals, make use of the corridor and whether the corridor can assist the genetic exchange of leopard cats, expand food sources, and contribute to the prosperity of the population.

Although there are no relevant reports in Shenzhen, there have been incidents of wild boars injuring people in urban suburbs in Tokyo and Nanjing cities. Therefore, with the construction of ecological corridors and the increasing number of urban residents in the country parks, avoiding possible human-animal conflicts with increased habitat connectivity is a problem that needs to be considered in advance.

5.6 Summary and synthesis

In summary, this paper uses the MCR method to identify potential ecological corridors in Shenzhen and determine their ideal width. First, core areas for restoring ecological connectivity in Shenzhen were identified. Then, potential corridor locations were identified and graded in importance using the Linkage Mapper tool. 1292.9 km^2 of urban ecological core areas were selected, and 103.7 km of ecological corridors were identified. Finally, this study provides a systematic approach to determining the appropriate width of ecological corridors. In conclusion, this study bridges the research gap regarding the restoration of functional connectivity and the most effective width of ecological corridors in Chinese mega-cities. It provides theoretical and applied guidance for future government policies on restoring ecological habitat networks.

Chapter Six

Synthesis and Future Perspectives

6.1 Main Results and Conclusions

In this thesis, the following studies were conducted. I investigated the survival and probability of the presence of carnivores in different Asian cities through a literature search, evaluating the effects of different urban environmental variables and species' physiological characteristics on the survival rate of carnivores in cities. I compared leopard cat occupancy in two cities, Shenzhen and Hong Kong, by using camera trap data and deploying additional camera traps at 1 km x 1 km in Shenzhen, and I assessed the pattern of occupancy in relation to different urban environmental factors. I also used kernel density estimation to study the distributional range and temporal activity patterns of leopard cats in Shenzhen. The spatial-temporal distribution patterns were overlaid with those of humans, domestic cats, and domestic dogs, with the interactions and ecological niche competition between leopard cats and the different species explored. Finally, I simulated potential leopard cat ecological corridors in Shenzhen (and performed a cost analysis) using a minimum cumulative resistance (MCR) model, combining the distribution density of leopard cats and environmental factors such as topography and human social factors.

The results show the following: 1) The proportion of built-up urban area was found to be the most critical factor influencing the survival of urban carnivores, with a highly significant coefficient of -0.5 ($p<0.05$) in my regression model. Canidae and Mustelidae were the two groups of carnivores with the highest survival rates in cities, with a greater than 70% survival rate. In terms of physiological characteristics, non-diurnal carnivores had a 28.4% higher urban survival rate than diurnal carnivores ($p<0.05$), and solitary carnivores had a 45.1% lower urban survival rate than social carnivores ($p<0.05$).

2) It was also found that the occupancy rate of leopard cats in Shenzhen was 18.8% lower than that in Hong Kong ($P<0.01$). In Shenzhen, the occupancy probability of leopard cats decreased from 0.55 at 0 m to 0.17 at 1750 m distance to the urban edge, and leopard cat occupancy probability was higher in agricultural (0.75) areas (i.e., areas with a larger proportion of agricultural land in the surrounding 2km buffer) than in urban land (0.35) in Shenzhen.

3) In terms of spatial overlap, I found that leopard cats overlap spatially with humans, domestic cats, domestic dogs, and wild boars by >55%. In terms of their temporal overlap, leopard cats had the highest temporal overlap with domesticated cats. They had peak activity times at 06:00 and 23:00, which were 3-4 hours later than the peak activity times of domesticated cats (02:00 and 20:00).

4) Combining the estimated spatial extent of the leopard cat's kernel density, I proposed 118 potential Shenzhen ecological corridors that assist the leopard cat's migration. The 23 most important ecological corridors were identified. The best corridors had an average width of 727 m, a total corridor area of 97.7 km^2, and a projected total cost of RMB 55.74 billion.

The main conclusions of my results are that the larger the forested area of a city, the higher the probability of survival of carnivores. Nocturnal and crepuscular, group-living animals are better adapted to urban life. Felines and mongooses are among the most sensitive species to urbanization and thus deserve priority protection. The occupancy rate of leopard cats in Hong Kong is greater than in Shenzhen because of the larger percentage of natural habitat area and more continuous habitats in the former. In addition to forests, agricultural areas are essential habitats for leopard cats within their urban distribution. The spatial and temporal distributions of leopard cats and domestic cats overlap significantly. However, they reduce mutual influence by setting different peak activity times. The location of critical ecological corridors, resistance points, and ecological nodes is essential in developing conservation policies.

6.2 Conservation recommendations

This study reveals a dearth of empirical evidence pertaining to urban wildlife, particularly in the urban centres of Asia that are still in the process of rapid development. Hence, it is imperative to build a comprehensive and enduring biodiversity monitoring network in Asian cities, with data available online and free-to-access repositories. Monitoring of population genetic data through time would also be useful. To this end, this study has provided fundamental data that can be utilized for prospective basic research endeavours and policy formulation pertaining to urban biodiversity.

This study also highlights the potential impact of urban built-up regions on biodiversity, underscoring the need for actions aimed at curbing excessive urban expansion and safeguarding the integrity of existing natural ecosystems. China has a strategy for habitat conservation in urban regions that involves the implementation of ”ecological red lines.” These red lines serve as boundaries that restrict urban development in order to safeguard habitats and preserve biodiversity (Bai, 2018). The efficacy of ecological corridors remains a subject of debate within scholarly circles (Rosenberg, 1997; McRae, 2012). Nonetheless, expanding the extent of natural urban habitats continues to be a viable strategy for achieving positive wildlife outcomes.

The survival of urban carnivores is also influenced by human activity and the presence of exotic species. The management of domesticated animal populations in urban environments primarily involves the implementation of sterilization programs and the enactment of legislation aimed at mitigating the issue of abandoned domesticated animals (Bonacic, 2019; Contreras-Abarca, 2023). These techniques effectively mitigate competition between domestic cats and wild predators, resulting in a direct reduction of predation by domestic cats on small vertebrates, particularly small rodents and birds, both food sources for leopard cats. As a result, these interventions play a significant role in the protection of urban carnivore biodiversity (Trouwborst, 2020; Loss et al., 2013).

6.3 Limitations

The main limitations of this study are threefold. First, insufficient sample size, both in the selection of Asian cities, the collection of urban carnivore presence data, and the infrared camera survey of leopard cats in Shenzhen. Larger sample sizes, as well as longer surveys, are needed in the future. Second, some of the data collection and modelling choices in this study may have led to biased conclusions, such as having to place cameras in out-of-the-way sites to reduce theft, which meant that high human activities sites were less sampled. In addition, individual identification of leopard cats was not possible, which in turn affected estimates of the population density of leopard cats. All of these issues may have resulted in the poor performance of certain models, for example, the GLMMs in Chapter Three. Third, due to COVID-19, the fieldwork of this study suffered from very significant limitations, including, but not limited to, my inability to (i) access the country parks during the lockdown period, and therefore, the infrared cameras were in the field for too short a period, and (ii) get permission to do wildlife cage trapping and work with GPS collars. These difficulties had substantial impacts on the direction of the project and the ability to analyse fine-scale leopard cat occupancy and movement. Future studies should explore these methodological approaches to understand leopard cat dynamics in Shenzhen.

6.4 Future study

The next step will be to carry out a dietary study on carnivores in urban habitat patches in Shenzhen to explore the proportion of human waste used and whether the overlap of diets of different carnivores affects the probability of the leopard cat's presence. As outlined in the previous section, using GPS tracking collars will enable the investigation of the behaviour and movement paths of leopard cats crossing roads and urban matrices, providing more detailed basic information for constructing ecological corridors. Metapopulation processes across different patches should be studied using genetics to explore whether factors such as the time of isolation and the composition of the landscape matrix hinder gene exchange (Ovaskainen, 2004; Wagner, 2013). Future research will also need to examine the effects of vegetation type and quality on urban carnivores more closely.

Fundamental studies on biodiversity in cities that combine theory with empirical data are sorely needed. For example, urban-rural gradient research (Faeth, 2011; McDonnell, 2008) can help to clarify the mechanisms through which human activities affect urban biodiversity. Research on how human impacts on urban biodiversity vary both spatially and temporally in complex urban environments is also significant from both theoretical and applied perspectives.

Appendix One

Appendices

City	Country	Region	Latitude	Longitude	Area	Population	Pop Density	GDP	Built-up	Coastal
Shanghai	China	EA	31.2	121.5	6340.5	24.9	3922.5	684.8	0.2	C
Beijing	China	EA	39.9	116.4	16411.0	21.9	1334.0	638.2	0.1	L
Shenzhen	China	EA	22.6	114.1	1997.5	17.6	8790.9	486.0	0.5	C
Guangzhou	China	EA	23.1	113.3	7434.4	18.7	2512.1	447.4	0.2	C
Chongqing	China	EA	29.6	106.6	16853.6	32.1	1901.9	442.1	0.0	L
Suzhou	China	EA	31.5	120.6	11920.0	12.7	1069.5	360.0	0.1	C
Chengdu	China	EA	30.7	104.1	8488.4	20.9	2466.5	315.6	0.1	L
Hangzhou	China	EA	30.3	120.2	14335.0	11.9	832.6	287.0	0.1	C
Nanjing	China	EA	32.0	118.8	6587.0	9.3	1414.1	259.2	0.1	L
Tokyo	Japan	EA	35.7	139.7	2188.7	13.7	6279.1	972.3	0.4	I
Singapore	Singapore	SEA	1.4	103.8	719.1	5.6	7843.1	361.0	0.5	C
Hong Kong	China	EA	22.3	114.2	2755.0	7.4	2683.1	341.4	0.2	C
Taipei	China	EA	25.1	121.5	271.8	2.6	9735.9	327.0	0.2	C
Kuala Lumpur	Malaysia	SEA	3.1	101.7	243.0	1.8	7377.0	54.5	0.9	C

Table A.1: City area data were taken from Demographic and Social Statistics of the United Nations Statistics Division (UNSD). Population data were taken from the Demographic Yearbook System 2020 of the United Nations Statistics Division (UNSD). Then, population density (PD) and gross domestic product per capita (GDPp) were calculated. Information on the percentage of built-up area (BU%, the ratio of land area of remote sensing land use type as built-up area to the area of the city's administrative division) for each city was obtained from the government website

Species	family	IUCN	habi	ER	BM	HR	LS	diur	social	diet	hiber
Arctictis binturong	Viverridae	VU	2	3000	9.88	6.2	18	non-diur	solitary	herbivore	0
Arctonyx collaris	Mustelidae	LC	4	1800	6.36	6.2	15.8	non-diur	solitary	omnivore	0
Ailurus fulgens	Ailuridea	EN	2	2600	4.9	1.83	14	non-diur	solitary	herbivore	0
Ailuropoda melanoleuca	Ursidae	VU	1	2900	108.4	9	12.5	non-diur	solitary	herbivore	0
Cuon alpinus	Canidae	EN	3	5300	14.17	9.88	16	diur	social	omnivore	0
Canis lupus	Canidae	LC	7	3000	32.18	6565	13	diur	social	carnivore	0
Catopuma temminckii	Felidae	NT	4	3738	11.5	4.15	20	diur	solitary	carnivore	0
Herpestes brachyurus	Herpestidae	NT	3	1500	1.85	1.825	8.9	diur	solitary	carnivore	0
Herpestes javanicus	Herpestidae	LC	6	1800	0.75	0.26	8	diur	solitary	carnivore	0
Lutra lutra	Mustelidae	NT	5	4120	6.75	4.025	22	non-diur	solitary	carnivore	0
Lynx lynx	Felidae	LC	5	5500	17.95	205	5	diur	solitary	carnivore	0
Meles anakuma	Mustelidae	LC	2	1963	5.1	0.785	10	non-diur	solitary	carnivore	1
Mustela erminea	Mustelidae	LC	6	4050	0.12	0.15	1.5	non-diur	solitary	carnivore	0
Martes flavigula	Mustelidae	LC	3	4510	1.84	6.75	16	diur	social	omnivore	0
Meles leucurus	Mustelidae	LC	4	3205	6.25	0.379	10	non-diur	solitary	omnivore	1
Martes melampus	Mustelidae	LC	3	2000	1	0.665	16	non-diur	solitary	omnivore	0
Meles meles	Mustelidae	LC	5	3300	11.7	0.75125	6	non-diur	social	omnivore	0
Melogale moschata	Mustelidae	LC	4	1500	0.81	0.2	13.5	non-diur	solitary	omnivore	0
Melogale personata	Mustelidae	LC	4	1505	1.7	0.65	10	non-diur	solitary	omnivore	0
Mustela sibirica	Mustelidae	LC	5	4875	0.41	4.08	2.1	non-diur	solitary	carnivore	0
Neofelis nebulosa	Felidae	VU	2	3000	19.68	35	11	diur	solitary	carnivore	0
Nyctereutes procyonoides	Canidae	LC	4	3000	4.04	1.13	7.5	diur	social	omnivore	0
Prionailurus bengalensis	Felidae	LC	5	3420	3.3	3.95	4	non-diur	solitary	carnivore	0
Paradoxurus hermaphroditus	Viverridae	LC	4	2500	3.16	12.5	18.5	non-diur	solitary	omnivore	0
Paguma larvata	Viverridae	LC	3	2680	4.3	2.7	20	non-diur	solitary	herbivore	0
Pardofelis marmorata	Felidae	NT	1	2500	2.85	5.3	12.25	diur	solitary	carnivore	0
Panthera pardus	Felidae	VU	6	5200	52.04	24	17	non-diur	solitary	carnivore	0
Prionailurus planiceps	Felidae	EN	2	700	6.75	12	14	non-diur	solitary	carnivore	0
Panthera tigris	Felidae	EN	3	4500	162.56	4658	26	diur	solitary	carnivore	0
Panthera uncia	Felidae	VU	4	5300	44.17	25.5	18	non-diur	solitary	carnivore	0
Ursus thibetanus	Ursidae	VU	5	4300	77.5	21.45	25	non-diur	solitary	omnivore	1
Viverricula indica	Viverridae	LC	5	2500	2.91	1.965	22	non-diur	solitary	carnivore	0
Viverra tangalunga	Viverridae	LC	3	2100	8.68	1.07	12	non-diur	solitary	carnivore	0
Vulpes vulpes	Canidae	LC	6	4500	3.51	8.5	12	non-diur	solitary	omnivore	0
Viverra zibetha	Viverridae	LC	3	3080	9.5	5.75	17.5	non-diur	solitary	carnivore	0

Table A.2: Information about Asian urban carnivores. Body mass (BM) data were obtained from the Late Quaternary Mammal Body Mass (MOM v3.3). From the IUCN Red List website, the endangeredness (ED), habitat type (how many habitats it can adapt to, HT), and elevation range (ER) were obtained for each target species. From the Animal Diversity Web (ADW), obtain information on home range size (HR), lifespan (LS), sociality (SC), activity patterns (AP), diet (Diet), and whether or not they hibernate (HB)

A.1 R code

Import packages

library(Hmisc) library(MASS) library(car) library(MuMIn) library(reshape2)

Set working directory and load data

spe <- read.csv(”species0317.csv”) city <- read.csv(”city0318.csv”)

Correlation matrix

Cormspe<- rcorr(as.matrix(spe[,8:12])) Cormspe

Cormcity<- rcorr(as.matrix(city[,8:12])) Cormcity

City Model

mcityf <- glm(ratio   scale(popdenb) + scale(gdpp) + BU +coast, data=city) mcityf summary(mcityf) vif(mcityf)

options(na.action = ”na.fail”) ddmcityf <- dredge(mcityf) ddmcityf submpp<- subset(ddmcityf,delta < 2) submpp

avgmcityf <- model.avg(ddmcityf, delta < 2) summary(avgmcityf) coef(avgmpp) confint(avgmpp, method=”boot”)

step<-stepAIC(mcityf,direction = ”both”)

mcity <- lm(ratio   BU, data=city) summary(mcity) coef(mcity) confint(mcity, method=”boot”)

#Plot city model

newdata <- with(city, data.frame(BU = rep(seq(from = 0, to = 1, length.out = 100))))

newdata2 <- cbind(newdata, predict(mcity, newdata = newdata, type = "link",se = TRUE)) newdata2 <- within(newdata2, PredictedProb <- plogis(fit) LL <- plogis(fit - (1.96 * se.fit)) UL <- plogis(fit + (1.96 * se.fit)))

ggplot(newdata2, aes(x = BU, y = PredictedProb))+ geom$_bar(stat = "identity", position = position_dodge(), color = "black", width = .4) + geom_errorbar(aes(ymin = LL, ymax = UL), position = position_dodge(.6), width = .2)theme_bw()$

ggplot()+ geom$_line(data = newdata2, aes(x = BU, y = PredictedProb), size = 1) + geom_point(data = city, aes(x = BU, y = ratio))$

p <- ggplot(city,aes(x=BU,y=ratio)) + geom$_point() + stat_smooth(method =' lm', formula = y x, colour =' royalblue')p$

l <- list(a = as.numeric(format(coef(mcity)[1], digits = 4)), b = as.numeric(format(coef(mcity)[2], digits = 4)), r2 = format(summary(mcity)$r.squared, digits = 4), p = format(summary(mcity)$coefficients[2,4], digits = 4))

eq <- substitute(italic(y) == b + a

p + geom$_text(aes(x = 4, y = 50, label = as.character(as.expression(eq))), parse = TRUE)$

Species Model

mspef <- glm(ratio scale(ER) + scale(BM) + scale(HR)+ scale(LS) + diur + social

+ diet + hiber + habi, data=spe) summary(mspef) vif(mspef)

options(na.action = ”na.fail”) ddmspef <- dredge(mspef) ddmspef submspef<-subset(ddmspef,delta < 2) submspef

avgmspef <- model.avg(ddmspef, delta < 2) summary(avgmspef) coef(avgmspef) confint(avgmspef, method=”boot”)

step<-stepAIC(mspef,direction = ”both”)

mspe <- glm(ratio scale(BM) + diur + social, data=spe) summary(mspe) confint(mspe, method=”boot”)

mspe1f <- glm(ratio scale(ER) + scale(BM) + scale(LS) + diur + social + diet + hiber + habi, data=spe1) summary(mspe1f) vif(mspe1f)

options(na.action = ”na.fail”) ddmspe1f <- dredge(mspe1f) ddmspe1f submspe1f<-subset(ddmspe1f,delta < 2) submspe1f

avgmspe1f <- model.avg(ddmspe1f, delta < 2) summary(avgmspe1f) coef(avgmspe1f) confint(avgmspe1f, method=”boot”)

step<-stepAIC(mspe1f,direction = ”both”)

mspe1 <- glm(ratio scale(ER) + scale(LS) + diet + habi, data=spe1) summary(mspe1)

mspe11 <- glm(ratio scale(LS) + diet, data=spe1) summary(mspe11)

coef(mspe1) confint(mspe1, method=”boot”)

#Plot species model

mspe <- glm(ratio scale(BM) + diur + social, data=spe) summary(mspe) newdata <- with(spe, data.frame(diur = factor(rep(c("diur","non-diur"), each = 10)), social = factor(rep(c("social","solitary"), each = 20)), BM = mean(BM))) newdata

newdataa <- with(spe, data.frame(diur = factor(rep(c("diur","non-diur"), each = 10)), social = factor(rep(c("social","solitary"), each = 20)), BM = mean(BM))) newdataa

newdatab <- with(spe, data.frame(diur = factor(rep(c("diur","non-diur"), each = 10)), social = factor(rep(c("social","solitary"), each = 20)), BM = mean(BM))) newdatab

newdata2 <- cbind(newdata, predict(mspe, newdata = newdata, type = "link",se = TRUE)) newdata2 <- within(newdata2, PredictedProb <- plogis(fit) LL <- plogis(fit - (1.96 * se.fit)) UL <- plogis(fit + (1.96 * se.fit)))

ggplot(newdata2, aes(x = social, y = PredictedProb, fill=diur))+ geom$_bar(stat = "identity", position = position_dodge(), color = "black", width =$ .5) + $geom_errorbar(aes(ymin = LL, ymax = UL), position = position_dodge(.5), width =$ .2)$theme_bw()$

newdataa <- with(spe, data.frame(diur = factor(rep(c("diur","non-diur"), each = 100)), social = factor(rep(c("social"), each = 200)), BM = rep(seq(from = 0.1, to = 170, length.out = 100),2))) newdataa

newdataa2 <- cbind(newdataa, predict(mspe, newdata = newdataa, type = "link",se = TRUE)) newdataa2 <- within(newdataa2, PredictedProb <- plogis(fit) LL <- plogis(fit - (1.96 * se.fit)) UL <- plogis(fit + (1.96 * se.fit))) newdataa2

ggplot(newdataa2, aes(x = BM, y = PredictedProb, fill=diur))+ geom$_ribbon(aes(ymin = LL, ymax = UL, fill = diur), alpha =$ 0.2) + $geom_line(aes(colour = diur), size = 1)$

newdatab <- with(spe, data.frame(diur = factor(rep(c("non-diur"), each = 200)), social = factor(rep(c(”social”,”solitary”), each = 100)), BM = rep(seq(from = 0.1, to = 170, length.out = 100),2))) newdatab

newdatab2 <- cbind(newdatab, predict(mspe, newdata = newdatab, type = ”link”,se = TRUE)) newdatab2 <- within(newdatab2, PredictedProb <- plogis(fit) LL <- plogis(fit - (1.96 * se.fit)) UL <- plogis(fit + (1.96 * se.fit))) newdatab2

ggplot(newdatab2, aes(x = BM, y = PredictedProb, fill=social))+ geom$_r ibbon(aes(ymin = LL, ymax = UL, fill = social), alpha = 0.2) + geom_line(aes(colour = social), size = 1)$

Appendix Two

Appendices

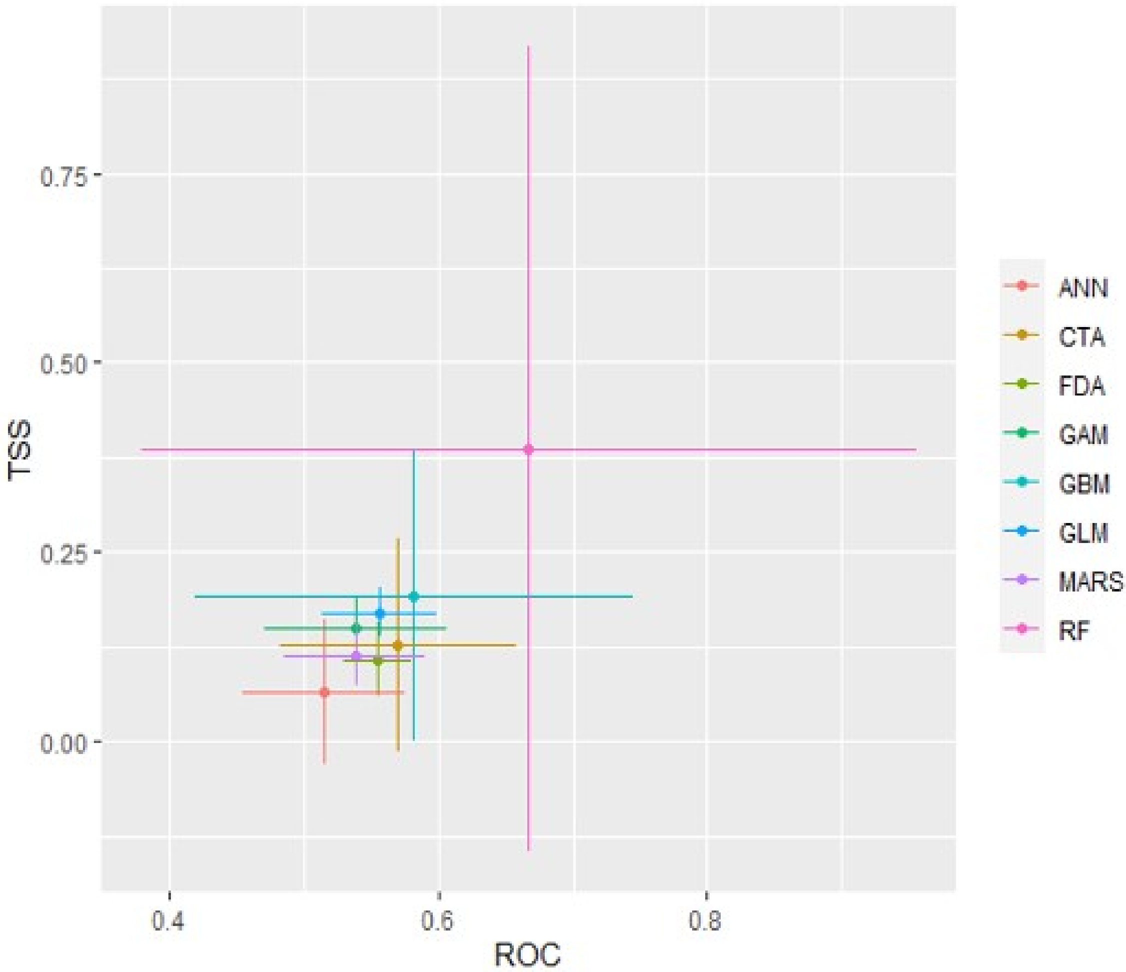


Figure B.1: Using the getevaluation() function in the R package biomod2, the Artificial Neural Network (ANN), classification tree analysis (CTA), flexible discriminant analysis (FDA), generalized additive models (GDA), generalized additive models (GAM), generalized linear models (GLMs), and multivariate adaptive regression splines (MARS) and the generalized additive model (GAM) algorithms were analyzed for Area Under Curve (AUC) and True Skill Statistic (TSS) to get the best algorithm.

B.1 R code

Import packages

library(lme4) library(sdm) library(MuMIn) library(Hmisc) library(car) library(vegan) library(tidyr) library(ggplot2) library(MASS) library(biomod2)

Set working directory and load data

SZHK <- read.csv(”SZHK220224.csv”)

names(env) p1 <- predict(mHK8,env,filename=”SZHK.img”) plot(getVarImp(mSZ8)) en <- ensemble(mSZ8,env,filename=”SZHK.img”,setting = list(method = ”weighted”,stat = ”tss”, opt = 2)) mSZ8 plot(du) mapview(du)

Correlation matrix

pairs(SZHK[,3:9]) CorSZHK<- rcorr(as.matrix(SZHK[,3:9])) CorSZHK write.csv(as.data.frame(CorSZHK),”SZHKcorr.csv”) summary(SZHK)

mo.dists <- as.matrix(dist(cbind(SZHK$X, SZHK$Y))) mo.dists.inv <- 1/mo.dists diag(mo.dists.inv) <- 0

Moran.I(SZHK$leo, mo.dists.inv, na.rm = TRUE)Moran.I(SZHK$du, mo.dists.inv, na.rm = TRUE) Moran.I(SZHK$dr, mo.dists.inv, na.rm = TRUE)Moran.I(SZHK$dw, mo.dists.inv, na.rm = TRUE) Moran.I(SZHK$ndvi, mo.dists.inv, na.rm = TRUE)Moran.I(SZHK$rug, mo.dists.inv, na.rm = TRUE)

VIF calculation (use 2 as cutoff threshold)

mszhkf <- glm(leo 1 + scale(du) + scale(dr) + scale(dw) + scale(ndvi) + scale(rug) + scale(du):scale(au) + scale(du):scale(pop)+ city + lu, data=SZHK, family=binomial(logit),na.action = na.fail) vif(mszhkf) dredge(mszhkf) summary(mszhkf) Anova(mszhkf) step<-stepAIC(mszhkf, $.^2, direction = "both")step < -stepAIC(mszhkf, direction = "both")$)

Determining the best random structure

mszhk1 <- glmer(leo 1 + scale(du) + scale(dr) + scale(dw) + scale(ndvi) + scale(rug)+ scale(du):scale(au) + scale(du):scale(pop)+ city + lu + (1|patch), data=SZHK, family=binomial(logit)) performance::icc(mszhk1) summary(mszhk1) isSingular(mszhk1)

mszhk2 <- glmer(leo 1 + scale(du) + scale(dr) + scale(dw) + scale(ndvi) + scale(rug)+ scale(du):scale(au) + scale(du):scale(pop) + city + lu + (1+scale(du)|patch), data=SZHK, family=binomial(logit)) performance::icc(mszhk2) summary(mszhk2) isSingular(mszhk2)

mszhk3 <- glmer(leo 1 + scale(du) + scale(dr) + scale(dw) + scale(ndvi) + scale(rug)+ scale(du):scale(au) + scale(du):scale(pop) + city + lu + (1+scale(dr)|patch), data=SZHK, family=binomial(logit)) performance::icc(mszhk3) summary(mszhk3) isSingular(mszhk3)

mszhk4 <- glmer(leo 1 + scale(du) + scale(dr) + scale(dw) + scale(ndvi) + scale(rug)+ scale(du):scale(au) + scale(du):scale(pop) + lu + city + (1+scale(du):scale(pop)|patch), data=SZHK, family=binomial(logit)) performance::icc(mszhk4) summary(mszhk4) isSingular(mszhk4)

mszhk5 <- glmer(leo 1 + scale(du) + scale(dr) + scale(dw) + scale(ndvi) + scale(rug)+ scale(du):scale(au) + scale(du):scale(pop) + lu + city + (1+scale(du):scale(au)|patch), data=SZHK, family=binomial(logit)) performance::icc(mszhk5) summary(mszhk5) isSingular(mszhk5)

Looking at the AIC values and select the best random structure (lowest AIC value)

ms <- model.sel(mszhkf,mszhk1,mszhk2,mszhk3,mszhk4,mszhk5,rank="AIC") write.csv(ms,”ms.csv”) ms

Best model mszhk0: with random effect of (1|city/patch)

summary(mszhkf) anova(mszhkf,mszhk1) coef(mszhkf) confint(mszhkf) r.squaredLR(mszhkf) plot(mszhkf)

Selecting the ”Optimal Model” with in the best random structure

options(na.action = ”na.fail”) ddmszhkf <- dredge(mszhkf) ddmszhkf write.csv(submszhkf,”ddszhk.csv”) # Select models with delta AICc < 2

submszhkf <- subset(ddmszhkf, delta < 2) submszhkf

Average them

avgmszhkf <- model.avg(ddmszhkf, delta<2) summary(avgmszhkf) anova(avgmszhkf) coef(avgmszhkf) confint(avgmszhkf, method=”boot”) r.squaredLR(mszhk) null.fit(avgmszhkf, evaluate = FALSE, RE.keep = FALSE, envir = NULL) plot(avgmszhkf) rep(seq(from = 0, to = 5400, length.out = 100),9)

mszhk <- glm(leo scale(dw) + scale(rug) + city + lu, data=SZHK, family=binomial(logit),na.action = na.fail) mszhkaa <- glm(leo scale(dw) + city + lu, data=SZHK, family=binomial(logit),na.action = na.fail)

varImp(mszhk) varImp(avgmszhkf)

newdata <- with(SZHK, data.frame(dw = mean(dw), rug = mean(rug), du = mean(du), city = factor(rep(c(”SZ”,”HK”), each = 1)), lu = factor(rep(c(”F”), each = 2))))

newdata2 <- cbind(newdata, predict(mszhk, newdata = newdata, type = ”link”,se = TRUE)) newdata2 <- within(newdata2, PredictedProb <- plogis(fit) LL <- plogis(fit - (1.96

* se.fit)) UL <- plogis(fit + (1.96 * se.fit)))

ggplot(newdata2, aes(x = du, y = PredictedProb)) + geomline(aes(colour = patch),size = 1)

SDM for SZ

dSZ <- BIOMODFormatingData(resp.var = SZ["leo"], expl.var = SZ[,c("du","dr","dw","ndvi","rug","lu","patch")], resp.xy = SZ[,c("Y","X")], resp.name = "leo") dHK <- BIOMODFormatingData(resp.var = HK["leo"], expl.var = HK[,c("du","dr","dw","ndvi","rug","lu","patch")], resp.xy = HK[,c("Y","X")], resp.name = "leo")

mSZf <- BIOMODModeling(data = dSZ, models = c('GLM','FDA',"MARS",'GAM','CTA','RF'), models.eval.meth = c("TSS", "ROC"), NbRunEval =500, DataSplit = 80) mHKf <- BIOMODModeling(data = dHK, models = c('GLM','FDA',"MARS",'GAM','CTA','RF'), models.eval.meth = c("TSS", "ROC"), NbRunEval =500, DataSplit = 80)

modelsscoresgraph(mSZf, by = "models", metrics = c("ROC","TSS")) mod-elsscoresgraph(mHKf, by = "models", metrics = c("ROC","TSS"))

dSZ <- sdmData(leo ., train = SZ) mSZf <- sdm(leo du + dr + dw + ndvi + rug + boar + cat + dog + human + lu + patch, data = dSZ, methods = c("glm","gam","mars", "mda","fda", "svm","tree","cart","brt","rf","gbm", "maxent","md","domain","bioclim"), replication = "boot", n = 100) mSZf plot(getVarImp(mSZf,id = c(1:100)))

mSZ <- sdm(leo patch + lu + rug + ndvi, data = dSZ, methods = c("glm"), replication = "boot", n = 100) mSZ rcurve(mSZ)

SDM for HK

dHK <- sdmData(leo du + dr + dw + ndvi + rug + lu, train = HK) mHKf <- sdm(leo du + dr + dw + ndvi + rug + lu, data = dHK, methods = c(”glm”,”gam”,”mars”, ”mda”,”fda”,”svm”,”cart”,”brt”,”rf”,”domain”,”bioclim”), replication = ”boot”, n = 100) mHKf plot(getVarImp(mHKf,id = c(1:100)))

gui(mHKf) mHK <- sdm(leo du + dr + dw + ndvi + rug + lu, data = dHK, methods = c(”glm”), replication = ”boot”, n = 100) mHK rcurve(mHK0) plot(getVarImp(mHK))

Appendix Three

Appendices

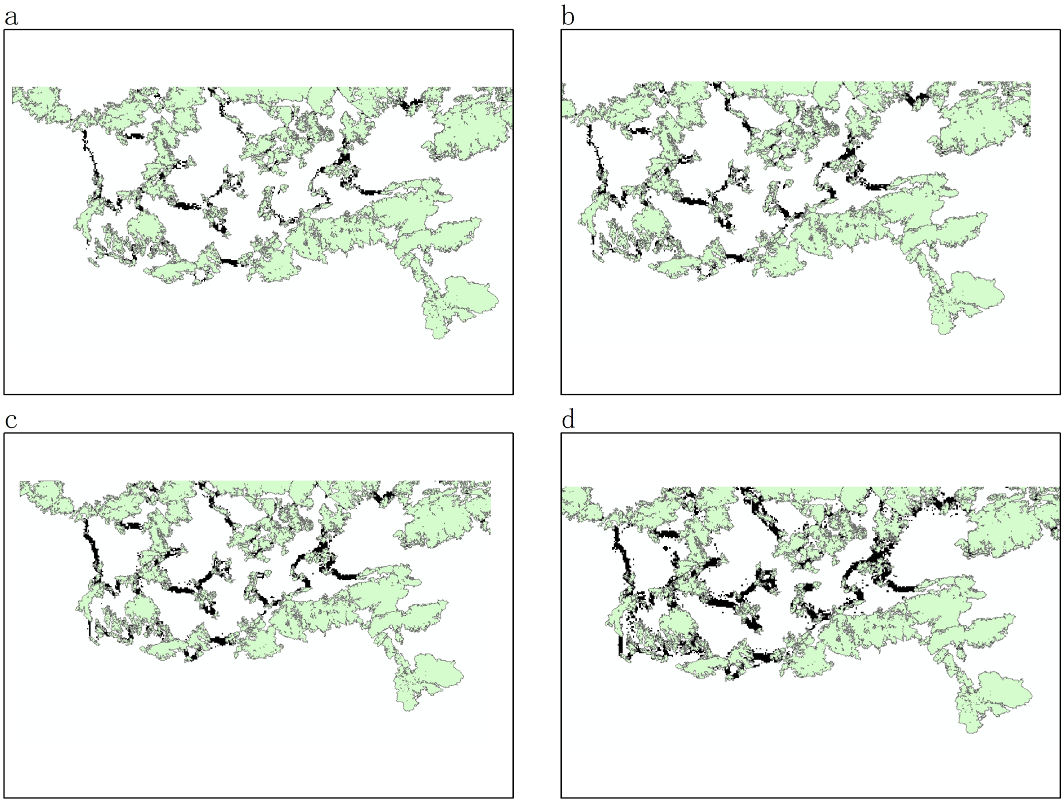


Figure C.1: The different widths of Shenzhen ecological corridors. Different corridor widths and their distribution ranges were selected based on different adaptive numerical thresholds.

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