Feeding Ecology and Conservation Biology of the Black Snub-nosed Monkey (*Rhinopithecus strykeri*)

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Painting by Xu Ling

Statement of Originality

The work presented in this thesis is, to the best of my knowledge and belief, original and my own work except where otherwise acknowledged or where reference is made in the text. This dissertation has not been submitted either in whole or in part, for a degree at this or any other university. This dissertation is structured as a series of connected papers that have been published (Chapter 5, 6, 7, 8 and 9) or is in preparation (Chapter 3) for publication at the time of thesis submission. As each chapter is an individual manuscript, some repetition occurs, particularly in the introductions, and there are inconsistencies in style and formatting.

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Table of Contents

Abstract

The black or Myanmar snub-nosed monkey (*Rhinopithecus strykeri*) was discovered in the Gaoligong Mountains of northeastern Kachin state, Myanmar in 2010, and was subsequently found in the mountains of northwestern Yunnan, China in 2011. Across these regions, there were an estimated 14-15 sub-populations with approximately 950 individuals in total (10 sub-populations with 490-620 individuals in China, and 4-5 sub-populations with 260-330 individuals in Myanmar). However, teams of people conducting field surveys and camera trap studies, of which I was part, only confirmed five sub-populations with 400 individuals on the Sino-Myanmar border from data collected 2012-2017. Based on approximately two years field searching, I and my colleagues discovered one sub-population (Luoma population) in the Gaoligong Mountains and conducted another 203 days of field observation to collect dietary data. I also conducted cafeteria feeding trials with 600 wild plant species on two captive individuals housed at Yaojiaping Wildlife Rescue Centre in the Gaoligong Mountains National Nature Reserve. I found that *R. strykeri* can potentially consume 593 items from more than 170 food plants of trees, bushes, and herbs representing 76 genera and 41 plant families, as well as 15 species of lichen. Among these food items and species, 14 plant species and four lichen species also are consumed by the wild monkeys as well. The food plants mainly distribute in intact sub-tropical evergreen broadleaf forests and hemlock-broadleaf mixed forests at an altitude of 2200-3000 m. Based on interview surveys, camera trap records, and habitat distribution modelling, I confirm this is the main elevational range used by *R. strykeri*. Nutritional studies and comparisons of 100 leaf items the monkeys selectively consumed (n = 70 plant species) with the nutrient content of 54 leaf items (n = 48 plant species) the monkeys' avoided in spring and autumn reveal that *R. strykeri* preferentially select leaves high in moisture (77.7%), crude protein (21.2%), total nonstructural carbohydrates (34.9%) and phosphorus (0.37%) while tending to avoid foods with a neutral detergent fibre content close to 35%. Foods selected in autumn were characterized by a higher amount of metabolisable energy than those rejected (1350 kJ/100g vs. 1268 kJ/100g). Random Forests modeling, an ensemble learning method, indicated that foods consumed during the two seasons were selected primarily based on their proportion of moisture, crude protein, neutral detergent fibre, metabolisable energy, phosphorus and total nonstructural carbohydrates. This nutritional profile is similar to other snub-nosed monkeys. Using interview-based survey data and MAXENT modelling of *R. strykeri* along the Sino-Myanmar border, I found that *R. strykeri* may inhabit a range from E98°20′–98°50′ to N25°40′–26°50′. Within this range, high-quality habitat at 1420 km^2, medium-quality habitat at 750 km^2, and low-quality habitats at 1410 km^2. Only 13.9% of the highly suitable habitat (medium + core habitat) for *R. strykeri* falls within protected areas in China. Approximately 2.6% of the entire habitat has been lost in the past 15 years, 96% of which has been in Myanmar. Two national parks (Imawbum National Park in Myanmar and Nujiang Grand Canyon National Park in China) are therefore proposed for saving this species. Lastly, for structuring a systematic transboundary conservation network in the highly-biodiverse but poorly-studied Gaoligong Mountains region, I used interview-based survey results (on animal distribution data) of three taxa (Primates, Pheasants and Mishmi Takin) and identified five flagship species (*R. strykeri*, *Hoolock tianxing*, *Trachypithecus shortridgei*, *Lophophorus sclateri*, *Budorcas taxicolor*) as surrogates of community biodiversity in the Gaoligong Mountains. After confirming the reliability of species distribution data via selective field surveys, I applied multicriteria decision analysis techniques along with data on habitat suitability (MAXENT Models) to highlight areas for transboundary conservation efforts. My results indicate that approximately 83.4% (10,398.7 km^2) of remaining habitat with high conservation value for each of the five flagship species is unprotected. This includes six large

zones separated by rivers and human settlements that should be designated as transboundary World Nature Heritage, National Parks, or Wildlife sanctuaries along the northern Sino-Myanmar border. Accordingly, I propose related conservation actions and policies for transboundary conservation in the Gaoligong Mountains along the northern Sino-Myanmar border.

List of Figures

List of Tables

Abbreviations

ADF	Acid Detergent Fibre
ADL	Acid Detergent Lignin
AMU	all male units
ANU	Australian National University
AOAC	Association of Official Analytical Chemists
AUC	area under curve
AW	absorb water
AZE	Alliance for Zero Extinction
BANCA	Biodiversity and Nature Conservation Association (Myanmar)
CA	crude ash
Ca	Calcium
CF	crude fibre
CL	crude lipids
cm	Centimetre
CP	crude protein
CR	Critically Endangered
EIA	Environmental Investigation Agency (UK)
EN	Endangered
FFI	Fauna and Flora International (Myanmar Program)
FW	free water
GLGMNNR	Gaoligong Mountains National Nature Reserve
GLGMs	Gaoligong Mountains
ha	Hectare
HI-LIFE	Landscape Initiative for Far-Eastern Himalayas
ICIMOD	International Centre for Integrated Mountain Development
IEHBR	Institute of Eastern-Himalaya Biodiversity Research
IUCN	International Union for Conservation of Nature
kg	Kilograms

kj	Kilojoules
km	Kilometres
km^2	square kilometre
KUN	Herbarium of Kunming Institute of Botany, the Chinese Academy of Sciences
l	Litre
LEK	local ecological knowledge
m	Metres
MAXENT	maximum entropy
MCDA	multi-criteria decision analysis
ME	metabolisable energy
mm	Millimetre
mo	Month
MoNREC	Ministry of Natural Resources and Environmental Conservation (Myanmar)
NDF	Neutral Detergent Fiber
NFE	Nitrogen Free Extract
NFGA	National Forestry and Grassland Administration (China)
OMU	one-male multi-female units
p	p-value of statistical test
P	Phosphorus
PA	protected area
RL	regression line
ROC	receiver operating characteristic curve
SAC	Standardization Administration of the People’s Republic of China
SE	standard error
UNEP	United Nations Environment Programme
UNESCO	United Nations Educational, Scientific and Cultural Organisation
WCS	Wildlife Conservation Society
WNH	world natural heritage

Chapter 1 Introduction

1 General Introduction

1.1 Background to the Current Research

Snub-nosed monkeys (*Rhinopithecus* genus) are a group of relatively large and phenotypically unique primates. They include five globally threatened species: golden snub-nosed monkey (*R. roxellana*, Endangered, Long & Richardson, 2008), black-and-white snub-nosed monkey (*R. bieti*, Endangered, Bleisch & Richardson, 2008), gray snub-nosed monkey (*R. brelichi*, Endangered, Bleisch et al., 2008), Tonkin snub-nosed monkey (*R. avunculus*, Critically Endangered, Le et al., 2008) and black or Myanmar snub-nosed monkey (*R. strykeri*, Critically Endangered, Geissmann et al., 2012). These species inhabit an array of environments in South and Southeast Asia, from limestone tropical rainforests in far northern Vietnam to alpine fir forests at the edge of the Tibetan Plateau in China. These rare and shy animals inhabit large home ranges in remote areas with precipitous and rugged terrain and severe climatic conditions and thus are difficult to habituate. Therefore, it is not surprising that their evolutionary history, behaviour, ecology and social organisation, have only recently been studied and partially understood by the scientific community (see Qi et al., 2014; Yu et al., 2016; Guo et al., 2018).

The black or Myanmar snub-nosed monkey (*Rhinopithecus strykeri*) is a newly described species of *Rhinopithecus* that was discovered in 2010, which is approximately 100 years after the discovery of the closely-related *R. avunculus* in 1912 (Geissmann et al., 2011). *R. strykeri* is Critically Endangered and restricted to small areas of the Gaoligong Mountains (GLGMs) in north Sino-Myanmar border areas (Meyer et al., 2017). To date, there have been few ecological or conservation studies of *R. strykeri* and before the works of this dissertation, the literature on this species comprised primarily information on species' evolutionary history, population distribution (based on interview survey), social organisation, home ranges (based on camera trapping data) and basic conservation needs (Geissmann et al., 2011; Liedigk et al., 2012; Ma et al., 2014; Li et al., 2014; Chen et al., 2015). Although these initial studies have offered insights into the behaviour and ecology of *R. strykeri*, studies on the feeding ecology and conservation biology are greatly needed for advancing our knowledge about the natural history of this iconic species and thus adopting pertinence measures to protect this Critically Endangered species. Before this study, there was only one confirmed *R. strykeri* population in Pianma at the western slopes of GLGMs, Yunnan Province of China (Long et al., 2012). Under difficult environmental conditions in GLGMs, my colleagues and I searched and located a second *R. strykeri* population at the eastern slopes of GLGMs in China.

The aim of this study was to give the first account of detailed feeding and nutrition of *R. strykeri* in the context of food resource distribution along elevational gradients and the consumption of food by both wild populations and two captive individuals. This dissertation also gives the first comprehensive evaluation of habitat quality, distribution and conservation status of this little-understood species and suggests a conservation framework for protecting *R. strykeri* in both China and Myanmar. Finally, this dissertation presents advanced interview-based surveys and selected flagship species to prioritise transboundary conservation networks in the data-deficient but highly biodiverse area of the GLGMs region. This same method can be applied to assess conservation priorities in other mountainous regions in the world. The significance of this study to primatology thus includes both increasing our ecological knowledge of *R. strykeri* and the genus *Rhinopithecus*, but more importantly, increases the possibility of the long-term survival of this species through the protection of local biodiversity by establishing critical areas for conservation priority.

1.2 Research Limitations

While the current study presents the first insights into the *R. strykeri*'s feeding-ecology and provides a conservation framework for this Critically Endangered primate, there were some limitations that affected the scope of this study. The study population was only discovered in September 2015 by myself and colleagues, thus they needed to be habituated. However, due to a long rainy season, dense forests and difficult logistics, our habituation process was impeded, resulting in relatively small and fragmented datasets and an inability to accurately assess ranging behaviour. Therefore, we decided to also use cafeteria feeding trials on two captive monkeys as an auxiliary mean to understand the feeding characteristics of the black snub-nosed monkey. In addition, although I tried to collect ground vegetation type data for modelling and mapping different vegetation distributions (such as mid-montane moist evergreen broad-leaved forest and mixed hemlock and evergreen broad-leaved forest), the current algorithm and model could not identify all vegetation types with satisfactory accuracy. Therefore, I used closed forest, open forest, scrubland, and grassland as a proxy for the specific forest types. All these limitations require further addressing in future research.

1.3 Outline of Dissertation

Following this Introduction, Chapter 2 presents a systematic review of the colobine monkeys including their evolutionary history, feeding ecology, ranging behaviour and social organisation patterns. Chapter 3 specifically discusses how food availability and feeding habits relate to snub-nosed monkeys' home range and daily ranging pattern, especially for temperate snub-nosed monkeys (*R. bieti* and *R. roxellana*). This chapter then uses this information to conduct a meta-analysis to make hypotheses for the current study. Chapter 4 gives an introduction to *R. strykeri* including specific information (climate, geographic information, and vegetation) regarding the study site, research subjects, population and conservation status, and how sub-populations of *R. strykeri* are found in GLGMs. Chapter 5 is a short chapter that shows the process of how we located the Luoma population. Chapter 6 presents the feeding ecology of *R. strykeri* through direct field observation and cafeteria-style feeding trials and the core elevational distribution range of most of their food plants. Chapter 7 provides a nutritional analysis of consumed and non-consumed plant parts of *R. strykeri* to determine what nutritional factors may be guiding food selection. In Chapter 8, the habitat distribution and qualification of *R. strykeri* are predicted and evaluated using information from interview-based surveys and MAXENT modelling. Accordingly, this chapter also uses habitat dynamics recorded over the past 15 years through remote sensing. Chapter 9 lays the foundation for a systematic conservation plan of the GLGMs region that both confirms the biodiversity of the area and provides a practical and rapid conservation prioritisation procedure that can be replicated for conservation planning in other remote mountainous areas. Finally, Chapter 9 summarises the findings from all the previous chapters to propose transboundary conservation recommendations and policies for *R. strykeri* and other important fauna species in the GLGMs region.

In this dissertation, I follow the updated primate taxonomy of Colin Groves (2001) and IUCN 2019-3. Therefore, some species' scientific names in this dissertation differ from how species were described in the cited literature. In addition, if there is no special statement unless otherwise noted, the photos and maps used in this dissertation were taken or illustrated by the dissertation author, Yin YANG.

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Chapter 2 Literature Review

2 Introduction to Colobines

Colobines are a group of Old World monkeys from Southeast Asia and Africa that include more than 60 species (Zinner et al., 2013). They are best known as leaf-eating monkeys as they have a large multi-chambered stomach that helps them deal with foods with high amounts of fibre and secondary plant compounds (Nijboer & Clauss, 2006). Many colobines are also renowned for their shy disposition and the difficult habitats in which they live (e.g. swamp forests, limestone rainforests, and subalpine conifer forests), making them particularly challenging study subjects (Fashing, 2006; Matsuda et al., 2010; Dong, 2012; Grueter et al., 2012). Nevertheless, research on colobines over the past 30 years has resulted in some exciting discoveries and controversial results (e.g. folivore paradox and ecological constraints model, Snaith & Chapman, 2007). In this chapter, I review literature of the distribution, evolutionary history, feeding, ranging behaviour, and social organisation of colobines, particular the Asian tribes and its unique genus, *Rhinopithecus*.

2.1 Taxonomy and Distribution of Colobines in China

The Colobinae is a subfamily of the primate family Cercopithecidae, consisting of two subtribes, the African Colobina (23 species in 3 genera: *Colobus*, *Piliocolobus*, and *Procolobus*) and the Asian Presbytina (55 species in 7 genera: *Nasalis*, *Presbytis*, *Pygathrix, Rhinopithecus, Semnopithecus, Simias,* and *Trachypithecus*) (Anandam et al. 2013; Zinner et al. 2013). Asian colobines are further classified into the odd-nosed monkey group (*Nasalis, Pygathrix, Rhinopithecus, Simias*) and the langur group (*Presbytis, Semnopithecus, Trachypithecus*) (Groves, 2005; Roos et al. 2011; Zinner et al. 2013; Roos et al., 2014).

China is home to 11 Asian Presbytina species, which belong to three genera: *Trachypithecus*, *Semnopithecus* and *Rhinopithecus* (Roos et al. 2014). The 11 confirmed Chinese colobine species and their distribution are listed in Table 2.1.

Table 2.1 Name list and distribution ranges of Asian Colobines in China.

Name	Distribution	Reference
Shortridge's langur *Trachypithecus shortridgei* (Wroughton, 1915)	Dulong River Basin, Nujiang Autonomous Prefecture of Yunnan Province	Cui et al., 2016
Capped langur *Trachypithecus pileatus* (Blyth, 1843)	Cuona County, Tibet	Hu et al., 2017
Phayre's langur *Trachypithecus phayrei* (Blyth, 1847)	Central, southern and southwestern Yunnan, with the Salween River as its west boundary	Groves, 2001
Indochinese grey langur *Trachypithecus crepusculus* (Elliot, 1909)	Biluo Snow Mountains, Wuliang Mountian, Yongde Great Snow Mountian, with Salween River as its east boundary and Southern of Yunnan Province	Xiao et al., 2013; Roos et al., 2013; Ma et al., 2015
Fran çois' langur *Trachypithecus francoisi* (Pousarges, 1098)	Chongqing, Guangxi and Guizhou provinces	Groves, 2001
White-headed langur *Trachypithecus leucocephalus* (Tan, 1957)	Longzhou, Ningming, Chongzuo and Fusui County, with the Mingjiang and Zuojiang River as its south, west and north boundaries	Huang et al., 2002
Nepal gray langur *Semnopithecus schistaceus* (Hodgson, 1940)	Bo Qu, Ji Long Zang Bu and Chumbi Valleys in Tibet;	Roos et al., 2013
Golden snub-nosed monkey *Rhinopithecus roxellana* (Milne Edwards, 1970)	Southern of Gansu, southern of Shaanxi, western and northwestern of Sichuan and western of Hubei Province	Roos et al., 2013

Black-and-white snub-nosed monkey *Rhinopithecus bieti* Milne (Edwards, 1897)	Yunling Mountains in Yunnan and Tibet, with the Yangtze River Mekong River as its west and east boundaries	Long & Kirkpatrick, 1994
Gray snub-nosed monkey *Rhinopithecus brelichi* (Thomas, 1903)	Fanjing Mountain in Guizhou Province	Yang et al., 2002
Black snub-nosed monkey *Rhinopithecus strykeri* (Geissmann et al., 2011)	Gaoligong Mountains in Yunnan Province	Meyer et al., 2017

Beyond these 11 species, it is also likely that the Tonkin snub-nosed monkey (*Rhinopithecus avunculus* Dollmann 1912) is distributed in China due to its occurrence in the border area of China and Vietnam.

2.2 Evolutionary History of Asian Colobines

The colobines (Colobinae) originated in Africa in the early Miocene, about 16-18 million years ago (Perelman et al., 2011; Roos et al., 2011). It is thought that the Asian colobines were separated from African colobines approximately 10-12 million years ago and subsequently moved into Eurasia and spread through middle Asia using an emerging land bridge that connected Africa and the Arabian Peninsula (Delson et al., 1994; Stewart & Disotell, 1998; Sterner et al., 2006; Perelman et al., 2011; Roos et al., 2011; Liedigk et al. 2012). After this ancestor of the Asian colobines entered the Hengduan Mountains it likely became geographically isolated by the development of the Himalayan Mountains and the reorganisation of large rivers in the eastern-Himalayan region (Yarlu Tsangpo, Irrawadi, Salween, Mekong, Yangtze and Red River). This is thought to have facilitated the split of Asian langurs into three lineages: *Semnopithecus*, *Trachypithecus*/*Presbytis*, and the odd-nosed monkeys (*Rhinopithecus*, *Pygathrix*, *Nasalis*, and *Simias*) by approximately 8-10 million years ago (see Fig. 2.1; Peng et al., 1993; Stewart & Disotell, 1998; An et al., 2001; Clark et al., 2004; Liang et al., 2008; Roos et al., 2011; Roos et al 2017). *Semnopithecus* then colonised the Indian subcontinent, while the ancestor of *Trachypithecus*/*Presbytis* invaded the Southeast Asian continent and the ancestor of the odd-nosed monkeys spread to the region that is currently China (Roos et al., 2011). *Trachypithecus* then re-contacted and hybridised with *Semnopithecus* again after diverging from *Trachypithecus*/*Presbytis*, and *Presbytis* first entered the Sundaland via Malaysian Peninsular (Ting et al., 2008; Osterholz et al., 2008; Roos et al., 2011). At this same time the ancestor of the odd-nosed monkey group spread south from the Inland Peninsula to Malaysian Peninsular and to Sundaland in the late Miocene. *Rhinopithecus* then separated from the ancestor of the odd-nosed group seven million years ago, followed by *Pygathrix* six million years ago, with the divergence of *Nasalis* and *Simias* as the end point at approximately one to two million years ago (Miller et al., 2005; Liedigk et al. 2012; Roos et al., 2011; Roos et al., 2017).

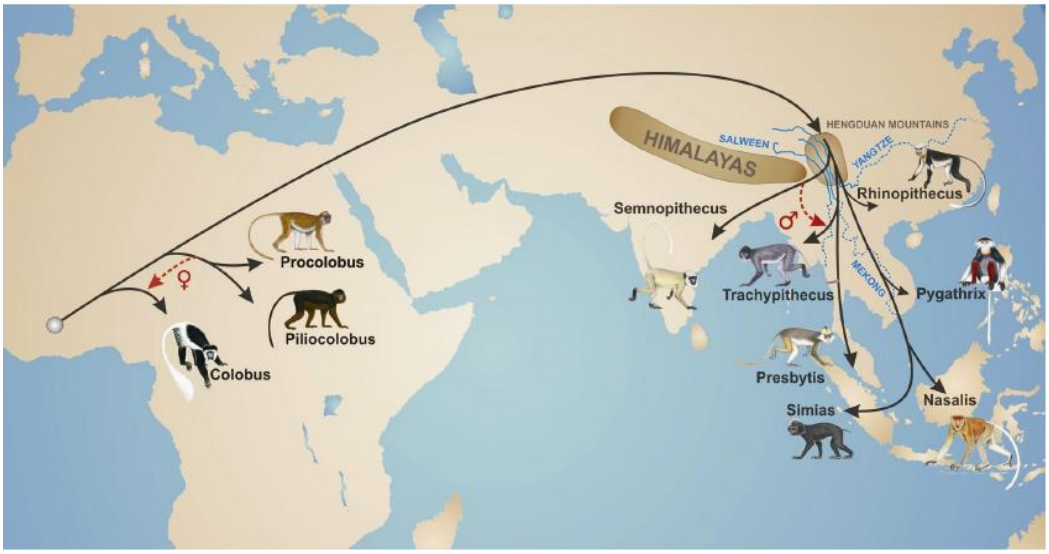

Figure 2.1 Dispersal routes of colobine monkeys (sourced from Roos et al., 2011).

This relatively rapid radiation and spread of Asian colobines enabled these primates to inhabit diverse environments in South China and Southeast Asia, ranging from tropical forests (e.g. douc langur *Pygathrix nemaeus*; nilgiri langurs *Semnopithecus johnii*), to swamp forests (e.g. maroon langur *Presbytis rubicunda*; proboscis monkeys *Nasalis larvatus*), to karst forests (e.g. white-headed langur *Trachypithecus leucocephalus*; Cat Ba langurs *T. poliocephalus*), to temperate forests (e.g. golden langur *T. geei*; Phayre's langurs *T. phayrei*), to along coastlines (e.g. silvered langurs *T. cristatus*), and in alpine mountains above 4,000 m in elevation (e.g. Kashmir gray langur *S. ajax*; black-and-white snub-nosed monkey, *Rhinopithecus bieti*) (Nijman, 2000; Riaz et al., 2010; Mittermeier et al., 2013; Ma et al., 2017).

2.3 Behavioural Ecology

2.3.1 Feeding ecology of colobines

Morphological adaptations for folivory

The diets of primates are informed by many internal and external factors including metabolism, digestive system, physical and chemical characteristics of foods and their spatiotemporal availability. Species of the subfamily Colobinae display many anatomical features that are adaptations to their highly-specialised diet that includes the ingestion of primarily fibrous vegetation. One such adaptation is their more robust mandible (compared to *cercopithecoids*) (Pan et al., 2008) and some dentition features – including a narrow incisor row, long and sharp molar crests, and high cusps – to better assimilate nutrients from highly-fibrous plants (Kay & Hylander, 1978; Wright & Willis, 2012). In addition, colobines possess anatomical specialisations of the stomach and foregut, including a complex, multi-chambered stomach, where symbiotic gastrointestinal microflora can break down and ferment structural polysaccharides (such as hemicellulose and chitin) from fibrous foods to yield energy-rich short chain fatty acids or simple sugars that can be more readily utilised by the monkey (Chivers & Hladik, 1980; Hume, 1989; Chivers, 1994; Kay & Davies, 1994; Cork, 1996; Lambert, 1998; Zhang et al., 2002; Yu et al., 2010; Karasov & Douglas, 2013; Zhou et al., 2014). Most colobines (e.g. *Semnopithecus* spp., *Trachypithecus* spp., and *Presbytis* spp.) have a three-chambered ('tripartite') forestomach, although odd-nosed monkeys have four-chambered ('quadripartite') forestomachs (Matsuda et al., 2019). This may be an adaptation to a dietary niche with a particularly high reliance on leaves/lichen as fallback foods. Moreover, the duplicated functional genes RNASE1 in colobines help them to produce more RNase for enhancing digestive efficiency in the small intestine (Zhang et al., 2002; Zhang et al., 2006; Zhou et al., 2014). This likely occurs through a process where the bacteria encompassing the highest RNA-nitrogen to total-nitrogen ratio of all cells are rapidly grown during fermentation, resulting in higher amounts of RNase being needed by the host to rapidly degrade RNA so that nitrogen can be efficiently recycled (Beintema, 1990; Zhang et al., 2002; Zhou et al., 2014).

Digestion inhibitors (which include phenolic compounds, such as tannins), and toxins (i.e. alkaloids) also can be neutralised more efficiently and tolerated better by colobines than by most monogastric primates, which increases the types of folivorous and non-folivorous foods that they can ingest (Oates et al., 1980; Davies et al., 1988; Ganzhorn, 1992; Chivers, 1998; Lambert, 1998, Karasov & Douglas, 2013, Bissell, 2014; Zhou et al., 2014; Li et al., 2015). Colobines have large salivary glands that produce salivary proteins that bind to tannins in ingested food before it even enters the digestive system, which enables these animals to immediately begin to enhance the nutritional value of a diet high in digestion inhibitors (Kay et al., 1976; Mau et al., 2011).

Characteristic diets of folivorous primates

While colobines are often referred to as leaf-monkeys, due to the high amount of leafy material they ingest, it has more recently been discovered that this group of primates often ingest a diet of diverse food items other than foliage. For instance, the pale-thighed langur (*Presbytis siamensis*) and *P. rubicunda* in Malaysia spend equal amounts of time eating leaves as they do eating fruits and seeds (Davies et al., 1988). In Western Ghats (India), *S. johnii*'s diet is dominated by fruits, seeds and flowers (71.43%) (Sivaperuman & Kumar, 2012) rather than by leaves. An extreme example comes from Sabangau (Borneo), where the diet of the *P. rubicunda* is made up of only 7.7% leaves with the rest made up of the consistently-available fruit supply (Ehlers Smith, 2013). Similar trends are seen with respect to food species where colobines are known to be selective and ingest a diverse number of food items and plant species. *T. francoisi* in Nonggang (China) consumed more plant species and more diverse food items than sympatric Assamese macaque (*Macaca assamensis*) (Zhou et al., 2018) (see Table 2.2 for more relevant information). Moreover, other than plant foods, some colobines also opportunistically consume animal foods, such as insects, birds and their eggs, and vertebrate flesh, including rodents, birds and their hatchling (Goodman, 1989; Srivastava, 1991; Koenig & Borries, 2001; Yang & Zhao, 2001; Li et al., 2003; Sayers, 2008; Zhao et al., 2008b; Ren et al., 2010; Yang et al., 2016).

As alluded to above, although colobines developed a great ability to forage on diverse foods that contain antifeedants like tannins and toxins, they are actually highly-selective feeders (Yeager, 1989; Kool, 1993; Boonratana, 2003; Dela, 2007, Guo et al., 2016). Waterman & Kool (1994) point out that primates must avoid possible toxins from eating only one food source, and thus select species from numerous dietary categories. Colobines may need to balance the choice between unripe fruits and high sugar-containing ripe/pulpy fruits (Collins

& Roberts, 1978; Ayres, 1989) because mature fruits with more sugars may incur augmented fermentation and bloat (Davies et al., 1983; Davies et al., 1988) but unripe fruits may contain more secondary plant compounds (Mehansho, 1987). This has been especially supported by Ungar (1995), who found that three monogastric primates (Lar gibbon *Hylobates lar*, crab-eating macaque *Macaca fascicularis*, Bornean orangutan *Pongo pygmaeus*) in Gunung Lcuser National Park (Indonesia) preferred ripe fleshy fruits with high sugar content, but sympatric Thomas's langur (*Presbytis thomas*) mainly consumed unripe fruits or large dry seeds with extremely low pH values. This is explained by Parra (1978) and Cork’s (1996) finding that foregut fermenters such as colobines seem to be better at detoxification but inefficient at energy intake compared with hindgut fermenters; hence, a balance of food selection between energy efficiency and the capability of expelling antifeedants may have to be considered by colobines (Freeland & Janzen, 1974; Milton, 1979; Parra, 1978; Glander, 1982; Kool, 1993; Cork, 1996; Schülke et al., 2006; Sayers et al., 2010).

Despite the role of digestion inhibitors, it is thought that food selection both within and between colobine species is primarily driven by protein availability (e.g. Milton, 1979; Waterman et al., 1988; Chapman & Chapman, 2002; Matsuda et al., 2013, Hou et al., 2018; Zhao et al., 2020). In fact, many colobine studies suggest that protein and fibre content play a decisive role in leaf selection, with animals primarily selecting food items with high protein-to-fibre ratios, and this ratio is correlated with colobine biomass at many sites (Davies et al., 1988; Oates et al., 1990; Wasserman & Chapman, 2003; Chapman et al., 2004; Chapman et al., 2012). Across three sites with different habitat qualities (intact forest, fragmented forest and degraded forest) at Kibale National Park (Uganda), colobine biomass of four populations of western red colobus (*Piliocolobus badius*) and four populations of mantled guereza (*Colobus guereza*) were significantly and positively related to the protein-to-fibre ratio of common food sources, rather than to any other measure of food energy (Wasserman &

Chapman, 2003). Similarly, *N. larvatus* in secondary riverine forests of Sabah (Borneo) and the *Presbytis rubicunda* in primary forest of the same area, forage more consistently on leaves with high protein-to-fibre ratios (Matsuda et al., 2013; Matsuda et al., 2017). Ganzhorn et al. (2017), on the other hand, found that based on a meta-analysis of 24 studies of folivorous primates' food selection that it is protein – rather than fibre content – that is the major determinant of leaf consumption, especially when the protein levels in the environment are lower than average. Moreover, changes to the biomass of eastern red colobus (*Piliocolobus rufomitratus*) over the past three decades cannot be positively associated with changes in the protein-to-fibre ratio of mature leaves in the forest, suggesting the protein-to-fibre model does not hold true for all species or in all habitats (Chapman et al., 2010; Gogarten et al., 2012).

In addition to plant chemistry, food selection both within and between species is also largely driven by food availability and habitat types. Davis et al. (1988) found that both the mitred langur (*Presbytis melalophos*) in West Malaysia and the maroon langur in east Malaysia spend about half of their feeding time eating leaves with high protein concentration and low fibre content; however, compared to *P. melalophos* in Leguminosae forest, they feed on leaves with lower levels of plant secondary compounds from some common tree species. Thus, *P. rubicunda* in the Dipterocarpaceae forest have to find leaves with relatively high levels of secondary compounds from rare trees and lianas. Within species, the annual diet of purple-faced langur (*Semnopithecus vetulus*) in the dry evergreen forest in Polonnaruwa (Sri Lanka) is dominated by the mature leaves (63%) of a few plant species with less seasonal variation (Vandercone et al., 2012); however, the diets of two populations of the same species in tropical wet evergreen rain forests in Panadura and Piliyandala (Sri Lanka) are largely dominated by fruits (52.3% and 53.9%) and include a diverse array of food plants (Dela, 2007). Moreover, the *S. vetulus* populations living in diverse environments exploit more plant species, with a higher proportion of seasonal foods across the year, than the popuations living in relatively-impoverished environments. In addition, compared with those populations in a continuous habitat who eat around 100 plant species and spend half their feeding time on leaves, a *T. francoisi* population in a highly-fragmented habitat eats fewer plant species (40 species) and devotes about 87% of their feeding time to leaves (Li et al., 2009).

Dietary Seasonality

Colobines are also known to shift the foods they eat throughout the year in response to what is seasonally available (Bennett, 1983; Gautier-Hion, 1993; Dasilva, 1994; Fashing, 2001b; Koenig & Borries, 2001; Marshall & Wrangham, 2007; Sayers & Norconk, 2008; Hanya & Bernard, 2015; Zhou et al., 2018) (see Table 2.2). This behavioural adaption enables primates to survive in diverse habitats and to adapt to seasonal fluctuations of available food items in those habitats (Kirkpatrick, 2007). Matsuda et al. (2009b), for example, found that seasonal fruit availability significantly impacts dietary diversity and activity budgets for *N. larvatus* in Malaysia and is negatively associated with the monkeys' daily travel ranges (Matsuda et al., 2009a). Therefore, the spatiotemporal distribution of preferred foods can impact colobines' habitat uses within forest strata. Bennett (1986) and Snaith & Chapman (2007) indicate that the availability of preferred foods are associated with daily range lengths, while length of time for a preferred food resource occurring, distribution and mass are correlated with the size of home ranges. Although many colobine species show this typical evolutionary dietary strategy, there is enormous plasticity expressed depending on environmental conditions (Tsuji et al., 2013; Grueter et al., 2009a).

Table 2.2 Information about food diversity and food selection of colobine species in different study sites.

Species	Study Site		Observation time	Dietary diversity	Top dietary species	Food preference related to availability (month or year)	Reference
Black-and white colobus *Colobus guereza*	Kalinzu Forest, Uganda		November 2013- April 2016 (4,308 h)	31 species in 24 families	Top 3 species (73.9%, 58% *Celtis durandii* leaves)	Young leaves (all year round)	Matsuda et al., 2020
Western red colobus *Piliocolobus badius*	Salonga National Park, Central DR Congo		September 1990 - September 1991 (13 mo)	84 species	Caesalpinioideae (65% in total, 61% legumes leaves, 33% seeds)	Leaves (December - July); seeds & fruits (August - November)	Maisels et al., 1994
Angola colobus *Colobus angolensis*			February - September 1991 (8 mo)	46 species	Caesalpinioideae (39% in total, 27% legumes leaves, 50% seeds)	Leaves (year-round); fruits (February - August); seeds (July - September)	
François' langur *Trachypithecus francoisi*	Nonggang Nature Reserve, China		August 2003 - July 2004 (739 hr)	90 species in 43 families	Top 10 species (62.2%)	Young leaves & flowers (April - September); seeds, petioles, & stems (October - March)	Zhou et al., 2006
Javan langur *Trachypithecus auratus sondaicus*	Pangandaran Nature Reserve, Indonesia	Cagar Alam	November 1984 - March 1985 (5 mo)	49 species	Top 15 species (73.3%)	Fruits (November - December); young leaves & leaf buds (January - March); flowers (March)	Kool, 1993
		Taman Wisata	July 1984 - February 1985 (8 mo)	88 species	Top 15 species (70.6%)	Fruits (July - September); young leaves & leaf buds (October - December); flowers (July - November, February)	
White-headed langur *Trachypithecus*	Fusui Rare and Precious Animal Reserve, China		September 1997 - September 1998 (13 mo)	50 species in 28 families	Top 10 species (61.8%)	Young leaves (February - May, August - October); mature leaves (December - January); fruits and flowers (beginning	Li et al., 2003; Li & Rogers, 2005

leucocephalus						of the rainy season)	
Nepal gray langur ***Semnopithecus schistaceus***	Langtang National Park, Nepal		January 2003 - February 2004 (1,067 hr)	43 species in ≥ 30 families	34 species (74.9%)	Leaf buds, ripe fruits, & evergreen mature leaves (December - March); deciduous young leaves (April - May); flowers, unripe fruits, & deciduous mature leaves (June - October)	Sayers & Norconk, 2008
Purple-faced langur ***Trachypithecus vetulus***	Sri Lanka	Panadura	August 1985 - February 1987 (19 mo)	22 species in 15 families	Top 5 species (54.4%)	Fruits (year around); seeds (around August); young leaves (year-round)	Dela, 2007
		Piliyandala	February 1986 - February 1987 (13 mo)	14 species in12 families	Top 5 species (82.1%)	Fruits (November - July); leaf petioles (August - October); seeds (around August); flowers (February - March)	
Proboscis monkey ***Nasalis larvatus***	Sabah, Malaysia		May 2005 - May 2006 (1,968 hr)	188 species in 55 families	Not available	Fruits (July - August); leaves (year-round)	Matsuda et al., 2009b
Maroon langur ***Presbytis rubicunda***	Sabangau, Indonesia		January - December 2011 (924.6 hr)	65 species in 32 families	25 species (87.7%)	Leaves (10%, late in the dry season & early of the wet season); seeds (76.4%, year-round)	Ehlers Smith et al., 2013
Indochinese gray langur ***Trachypithecus crepusculus***	Wuliangshan, China		1738 hours during 2010-2013	148 species (including two lichens)	Top 10 species (43.6%)	Young leaves (February - May); fruit/Seeds (August - November); mature leaves (June - July)	Fan et al., 2015
Phayre's langur ***Trachypithecus phayrei***	Mt. Gaoligong, China		August 2012–July 2013	50 species	17 species (75%); Fagaceae (20.5%); Lauraceae (17%); Rosaceae (16%)	Fruits (> 50%, August - September); seeds (87.1%, October); buds/young leaves (> 90% April - July)	Ma et al., 2017

Fallback foods

Falback foods refer to foods that are relied on when preferred foods are unavailable (Marshall et al., 2009). Fallback foods play an important role in shaping the feeding, ranging, socioecology, and population abundance of colobines (Grueter et al., 2009b; Hanya & Bernard, 2015, 2016). For example, on Cat Ba Island, northeastern Vietnam, *T. poliocephalus* fall back on less nutritious mature leaves and thus devote more time to foraging and less time to socialising in the dry season than in the wet season (Hendershott et al., 2016, 2017). In the lowland dipterocarp forest of Danum Valley, Borneo, *Presbytis rubicunda* feed on young leaves of a liana (*Spatholobus macropterus*) (with high available protein and high stem density compared to other young leaves) as their fallback strategy, which results in them having a relatively long daily range length for a species with a small home range (Hanya & Bernard, 2012, 2015, 2016). When fallback food quality is high enough to satisfy nutritional requirements and quantity is superabundant, animals are, in principle, not limited by food.

2.3.2 Ranging behaviours of colobines

As with diet, ranging patterns of colobines vary between and within species. Compared to sympatric frugivores species, such as macaques (*Macaca* spp.) and apes (i.e. chimpanzee *Pan troglodytes*), folivores (colobines) tend to have smaller home ranges (smaller than 1 km^2) (Clutton-Brock & Harvey, 1977; Chapman & Chapman, 2000; Singh et al., 2000). Harris & Chapman (2007), for example, found that overlapping home ranges of eight multimale–multifemale groups of eastern black-and-white colobus (*Colobus guereza*) range from 0.067 to 0.328 km^2 in Uganda's

Kibale National Park. In Western Ghats of south India, the home range of northern plains gray langur (*Semnopithecus entellus*) (0.12 km^2) is far smaller than sympatric lion-tailed macaque (*Macaca silenus*) (1.5 km^2) and bonnet macaque (*M. radiata*) (0.8 km^2) (Singh et al., 2011). See Table 2.3 for more relevant information.

Regarding daily ranging patterns, daily range length differs from 491 m for *T. leucocephalus* (Zhou et al., 2011a) to 1,734 m for *N. larvatus* (Matsuda et al., 2009a). The daily range lengths of *T. francoisi* in Fusui Nature Reserve (China) vary from 341 to 577 m, which correlates with changes of food resources between the rainy and dry seasons (Zhou et al., 2007). In *S. ajax* units in Machiara National Park (Pakistan), the daily range lengths of an all-male unit (1,840 m) are larger than two one-male multi-female units (small units: 1,230 km; large units: 1,750 km) because bachelor males are more actively searching for takeover and mating opportunities (Minhas et al., 2013).

Table 2.3 Relationship of range sizes and diet of colobines.

Species	Study Site	Habitat	Group Name/Size		Home Range Size (ha)	Daily range length (m)	Fruit/Seeds in Diet (%)	Leaves in Diet (%)	Reference
Mantled guereza *Colobus guereza*	Kakamega, Kenya	Tropical Rainforest	T	11	20.25	434	NA	NA	Fashing, 2001a
			O	7	16	614			
			GC	10	12	552			
			ML	15	12.75	551			
			BS	21	17.25	708			
Black colobus *Colobus satanas*	Lopé Reserve, Gabon	Tropical Rainforest	G1	18	573	852	52	43	Fleury & Gautier-Hion, 1999
			G2	11	224	608	NA	NA	
Western red colobus *Piliocolobus badius*	Kibale National Park, Uganda	Moist Evergreen Rainforest	L	48	36.9	577	NA	NA	Gillespie & Chapman, 2001
			S	24	5.04	257			
White-headed langur *Trachypithecus leucocephalus*	Fu Shui, China	Limestone Seasonal Rainforest	G1	16	23.8	491	12.4	83.4	Zhou et al., 2011a
			G2	11	33.8	512	14.5	91.0	
François' langur *Trachypithecus francoisi*	Nonggang, China	Limestone Seasonal Rainforest	12		65.4	541	31.4	52.8	Zhou et al., 2011b
Maroon langur *Presbytis rubicunda*	Sabangau, Indonesia	Tropical Peat-Swamp Forest	7		108	1,645	83.7	10.2	Smith et al., 2013

Species	Study Site	Habitat	Group Name/Size	Home Range Size (ha)	Daily Length Range (m)	Fruit/Seeds in Diet (%)	Leaves in Diet (%)	Reference
Proboscis monkey *Nasalis larvatus*	Sabah, Malaysia	Riverine Forest	17	138.3	799	69.5	25.9	Matsuda et al., 2009a, b
Northern plains gray langur *Semnopithecus entellus*	Langtang National Park, Nepal	Coniferous Forest and Scrub	30	NA	1,500	29.7	57.1	Sayers & Norconk, 2008
Kashmir gray langur *Semnopithecus ajax*	Machiara National Park, Pakistan	Mixed Moist Temperate Forest	SBG 74 LBG 177 AMG 25	3 2 14	1,230 1,750 1,840	17	63.39	Minhas et al., 2010; 2013
Gray snub-nosed monkey *Rhinopithecus brelichi*	Mt. Fangjinshan, China	Temperate Broadleaf Forest	385	3500	1,290	15	71	Bleisch et al., 1993
Golden snub-nosed monkey *R. roxellana*	Zhouzhi, China	Mixed Deciduous Broadleaf & Conifer Forest	112	1830	2,100	29.4	24 (lichen 29)	Guo et al., 2007; Tan et al., 2007
Black-and-white snub-nosed monkey *R. bieti*	Samage, China	Mixed Fir & Broadleaf Forest	410	2125	1,514	13.9/8.8	16.9/16.1 (lichen 66/67.7)	Grueter et al., 2008; 2009b

There is also variation in the degree of territoriality among colobines. For example, limited overlap and strong territorial defence has been reported in six *Presbytis* langurs across 12 different sites of Southeast Asia (van Schaik et al., 1992), while the home ranges between two sympatric species of colobine monkeys (Mentawai langur *Presbytis potenziani* and pig-tailed langur *Simias concolor*) in Mentawai Islands (Sumatra) completely overlaps with little evidence of territoriality (Hadi et al., 2012). For example, colobines living in multimale–multifemale groups or modular societies (e.g. *Semnopithecus* spp., *N. larvatus* and *Rhinopithecus* spp.; See section 2.3.3) are observed to have a lack of intergroup hostility and territoriality, resulting in total or partial overlapping of neighbouring units or bands' home ranges (Boonratana, 2000; Borries, 2000; Tan et al., 2007; Matsuda et al., 2009a; Ren et al., 2016).

Many detailed studies show that the ranging patterns of colobines are impacted by the spatiotemporal availability of food resources (Bennett & Davies, 1994; Boonratana, 2000; Koenig, 2000; Kirkpatrick, 2007; Sayers & Norconk, 2008; Matsuda et al., 2009a). The rate of food patch depletion, for example, is positively correlated with group size, daily range length and home range for some colobine monkeys (i.e. *Piliocolobus badius* Gillespie & Chapman, 2001; Table 2.3; ursine colobus *Colobus vellerosus* Teichroeb & Sicotte, 2009). Large groups likely need to prolong or broaden their day ranges to reach enough food patches to adequately feed all group members, especially as female reproductive success is limited by access to such foods, increasing scramble competition (Isbell & Young, 2002; Borries et al., 2008). This is referred to as the ecological-constraints model and proposes that in order to obtain enough food, larger groups will have longer daily range lengths than smaller groups in order to obtain sufficient food resources; thus, large groups face the negative consequences of increased energy costs associated with long travel distances and bigger home ranges (Milton, 1984; Janson, 1988; Wrangham, et al. 1993; Chapman & Chapman, 2000; Gillespie &

Chapman, 2001; Chapman & Pavelka, 2005; Snaith & Chapman, 2008; Teichroeb & Sicotte, 2009; see also Hanya & Bernard, 2016). In Kibale National Park (Western Uganda), for instance, a larger population (48 individuals) of *P. badius* rested for less time and visited feeding patches more frequently across a larger home range than did a smaller population (24 individuals) (Gillespie & Chapman, 2001). In Chongzuo National Nature Reserve (China), two larger one-male multi-female groups of *T. leucocephalus* (15/16 individuls) spent more time moving, less time resting, and had a longer mean daily range length than those in two small groups (6/5 individuls) (Zhang et al., 2019). Isbell (1991; 2012) suggests that daily range length is determined through a balance of energetic costs and benefits between selection of different paths and their travel distance. In addition, daily range length is longer in some colobine monkeys when the nutritional needs are centred on seasonal and restricted food sources instead of folivory (Struhsaker, 1980; Kirkpatrick, 2007; Sayers & Norconk, 2008).

Despite this, some studies do not support the ecological-constraints model, with no relationship between large group size and either daily path or home size and feeding competition (Struhsaker & Leland 1987; Yeager & Kirkpatrick, 1998, Yeager & Kool, 2000; Grueter, 2009; Isbell, 2012). In the Kakamega Forest (Kenya), there was no intragroup contest competition among five multimale–multifemale groups of *C. guereza* and these groups' mean daily range length had no correlation with their group sizes or local food availability (Fashing, 2001a). For six one-male multi-female groups of *Presbytis thomasi* in northern Sumatra (Indonesia), larger groups exploited larger home ranges, however, seasonal differences in diet had no effect on daily range length and time budget (Steenbeek & van Schaik, 2001). This variation may exist due to variation in dominance structure. Isbell (1991) suggests that where groups do not have strong, linear dominance hierarchies, then group size does not influence daily range length. This is because energetic costs associated with female-

female competition for food resources and reproductive success within groups may be weak or non-existent in species that have undifferentiated relationships between females (Isbell, 1991). Some Asian colobines form large bands with weak dominance hierarchies leading to large overlapping home ranges with other bands in the area (see section 2.3.3). For example, in intact habitats in Sabah (Bornean Malaysia), *N. larvatus* units living in multilevel societies with completely overlapping home ranges of neighbouring units are nonterritorial and their daily range lengths negatively correlate with fruit availability (Boonratana et al., 2000; Matsuda et al., 2009a). In addition, competition between females in this species seems to be more related to sleeping "blanch" than access to feeding sites (Yeager, 1990; Matsuda et al., 2012a). In Samage (China), two bands of *R.bieti* with more than 400 members used large and partially-overlapping home ranges but rarely shifted their home ranges (Grueter, et al., 2008; Ren et al., 2016), indicating that their habitat can support the dietary needs of large bands and that intragroup scramble competition may have little impact on band relationships. In the Wuliang Mountains (China), when *T. crepusculus* population size increased from 81 to over 90 individuals, there was no associated increase in daily range length or change in time budgets because of high dietary diversity (Fan et al., 2015). These results may reflect that in many of these species feeding resources are adequate for the populations that exist in local areas and the majority of the populations do not reach environmental carrying capacity, especially for those which are able to survive on mature leaves or other non-patchily distributed, abundant resources (Yeager & Kirkpatrick, 1998). Moreover, daily range length may simply be decided by food quality and food distribution. For example, daily range lengths of *S. entellus* show a significant discrepancy related to seasonal availability of certain foods: in winter they have longer daily-travel distances in search of soft underground vegetative organs, fruits, and deciduous mature leaves (compared to monsoon and spring seasons) (Sayers & Norconk, 2008). Similar trends occur among most colobine species in Himalayan areas (Bishop, 1979; Bennett, 1986; Stanford, 1991; Kirkpatrick, 2007).

2.3.3 Social organisation of colobines

Generally, the social organisation of colobines has three patterns (one male-multifemale groupings, multimale-multifemale groupings, and modular societies) with different dispersal patterns (Grueter & van Schaik, 2010). One male-multifemale groupings are a representative pattern for *Presbytis* spp., *Trachypithecus* spp. and *Semnopithecus* spp., and may represent the ancestral social organisation of Asian Colobines (Grueter & van Schaik, 2010). Less common than one male-multifemale groupings, monogamy appears in only two Asian colobines (*P. potenziani* and *Simias concolor*) in Mentawai Islands (Indonesia) (Watanabe, 1981; Bennett and Davies, 1994; Hadi et al., 2009). Such social organisation may be the result of extensive hunting with habitat fragmentation that reduced grouping sizes (Watanabe, 1981; Hadi et al., 2009; Erb et al., 2012). Multimale-multifemale groupings are often made up of groups of both males and females intermixed with one or several all male units (AMUs). These can be found in *Pygathrix* spp. and *Semnopithecus* spp. (Rajpurohit et al., 1991; Grueter & Zinner, 2004, Phiapalath et al., 2011). Extra males (such as mature sons or males whose groups have been taken-over) come together to form loosely-bonded AMUs and travel in a congregation accompanying one male-multifemale units (OMUs). Modular societies are found in *N. larvatus* (Yeager 1990) and *Rhinopithecus* spp. (Qi et al., 2014) and are usually formed by three structural levels: the core OMUs, the band, and the troop or herd. Several OMUs and AMUs form bands and the aggregation of bands become troops, which temporarily share a foraging area or sleeping place (Qi et al., 2014; Grueter et al., 2017). Dispersal in these social organisations of Asian colobines include female philopatry and male dispersal (e.g. capped langur *T. pileatus*, Stanford, 1991; Nepal gray langurs *S. schistaceus*, Borries et al., 2001, 2017) and both-sex dispersal (e.g. *P. thomasi,* Sterck et al. 2005; *S. johnii,* Kavana et al., 2014; *N. larvatus*, Matsuda et al., 2012b; *R. roxellana,* Qi et al., 2014; *T. leucocephalus,* Wang & Yao, 2017). Male philopatry and female dispersal is currently found in *T. phayrei* (Feder et al., 2019) and African colobines (e.g. *Piliocolobus badius,* Struhsaker, 1975; Ugandan red colobus *P. tephrosceles,* Struhsaker, 2010).

2.4 Chinese Colobines

China is one of the top conservation priorities for the world's non-human primates (Li et al., 2018). China has at least 25 primate species, of which nine species are endemic and 20 have been listed as threatened by the IUCN Red List (Li et al., 2018). It is worth noting that around 70% of primate species in China have less than 3,000 individuals (Estrada et al., 2017). All 11 Chinese colobine species are listed as IUCN-Critically Endangered, Endangered or Vulnerable (Li et al., 2018). The population trends of *T. pileatus* and *S. schistaceus* in China are ‘status unknown’. Only *R. bieti* is showing signs of population increase due to great conservation efforts in the past two decades (Long Y. C., pers. comm., 2018). Continued hunting, wildlife trade, and transformation of old-growth forests in China for wood products, agricultural expansion, and urbanisation in the past thousand years have resulted in habitat reduction, altitudinal shifts and declines of distribution ranges, and population declines and extirpation of these Chinese colobines and other primates (Li et al., 2002; Brandt et al., 2012; Turvey et al., 2015; Zhao et al., 2018; Ni et al., 2018). For example, China's primary forests are estimated to have lost between 1.9 and 2.7 million km^2 in the past two thousand years (Ahrends et al., 2017). Based on an analysis of distribution records of ancient county annals and history books of China, Li et al. (2002) found that snub-nosed monkeys were once distributed from south of the yellow river basin in central China, east to Zhejiang province, west to Yunnan province, and all the way south to the sea border 400 years ago. However, now they are either Endangered or Critically Endangered and distributed in very limited areas of China, north Vietnam and northwest Myanmar (see details in next section) due to rapid human population increases, deforestation, hunting and wars in the first half of the 20th century (Li et al., 2002). Therefore, understanding the drivers of continued threat factors, identifying protection priorities and developing conservation action plans will help to effectively protect these colobines in China.

2.5 General Introduction to Snub-Nosed Monkeys

The *Rhinopithecus* genus is more commonly known as the snub-nosed monkey group and consists of five allopatric species (Tonkin snub-nosed monkey *R. avunculus*, gray snub-sosed monkey *R. brelichi*, golden snub-nosed monkey *R. roxellana*, black-and-white snub-nosed monkey *R. bieti* and black snub-nosed monkey *R. strykeri*). The name ‘snub-nosed monkey’ is derived from their unusual nose morphology including the short stump of a nose with forward nostrils on their round and colourful faces (Fig. 2.2). Compared with other colobine species, snub-nosed monkeys have a relatively large body, and distinct sexual dimorphism in terms of canine dimension (Jablonski & Pan, 1995) and body size (female 7.8-9.4 kg, male 17-39 kg, Kirkpatrick, 1998; Kirkpatrick & Grueter, 2010; Anandam et al., 2013; Grueter & van Schaik, 2009). They also have extraordinarily red lips or bulbous flaps that are thought to be used as badges in mating competitions (Grueter et al., 2015a; Grueter et al., 2015b). As mentioned in Chapter 1, extremely difficult-to-traverse habitats and the shy nature of the snub-nosed monkeys make the behaviour of these amazing animals relatively unknown before the 1990s. Over the past 30 years, a large number of new and exciting studies have been carried out on these rare species due to the joint efforts of both local- and international-primatologists. In this section, I give an overview of the current knowledge of the natural history, social organisation, and conservation status of the five species of snub-nosed monkeys.

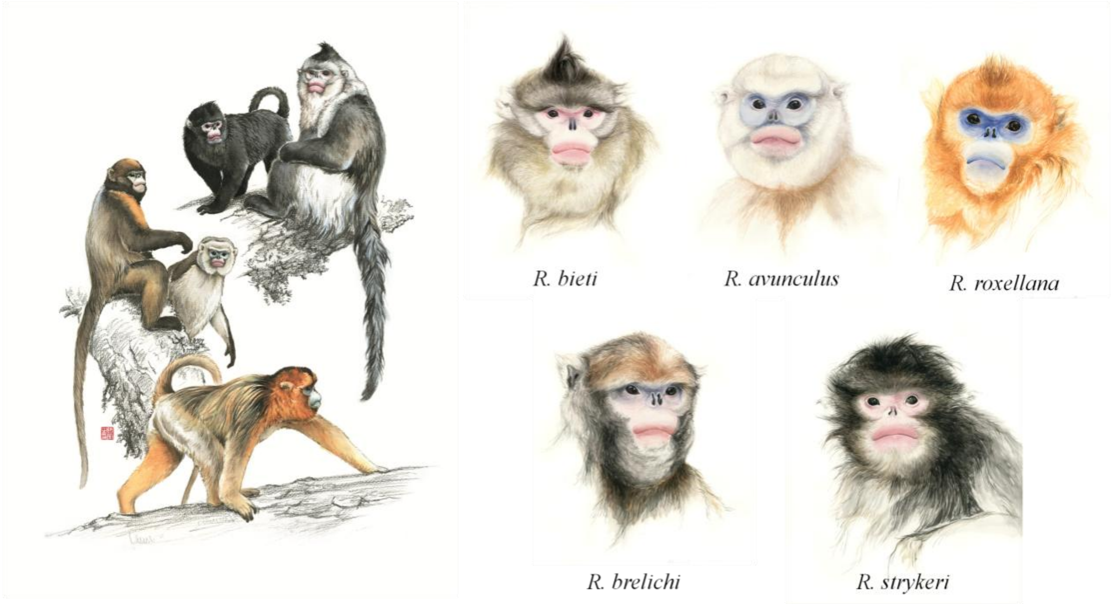

Figure 2.2 Portraitures of snub-nosed monkey species (*Rhinopithecus* spp.). Drawings by Zhang Yu & Li Qian, Chinese National Geography.

2.5.1 Distribution and conservation status of snub-nosed monkeys

The five snub-nosed monkey species are limited to China, northern Myanmar and northern Vietnam (Fig. 2.3) and all of them have been threaten by human impacts and are thus listed as either Endangered (EN) or Critically Endangered (CR) on the IUCN Red list (Roos et al., 2014; Li et al., 2018). Below, each species will be discussed individually.

Tonkin snub-nosed monkey: *R. avunculus* Dollman 1912 (CR)

Population: decreasing with approximately 200-250 individuals remaining in the wild (Le, 2014; Covert et al., 2017).

Distribution: restricted to small forest patches in Tuyen Quang and Ha Giang Provinces, far northeastern Vietnam.

Threats: hunting, development activities (including a large hydroelectric dam project) and deforestation.

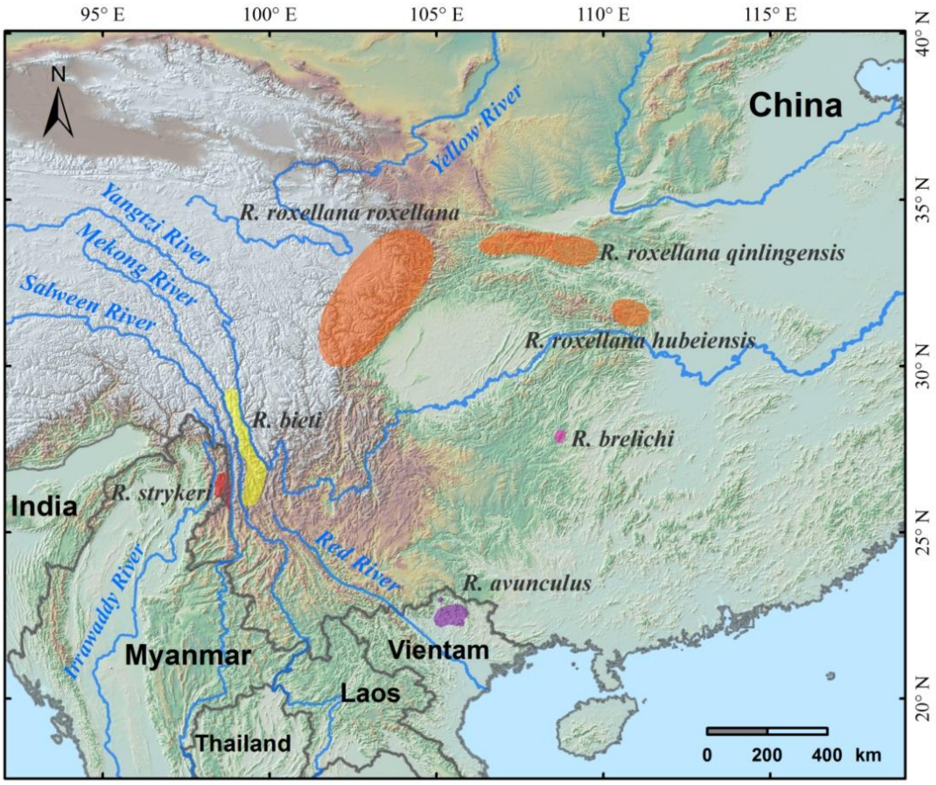

Figure 2.3 Distribution of snub-nosed monkey species (*Rhinopithecus* spp.)

Gray (Guizhou) snub-nosed monkey: *R. brelichi* Thomas 1903 (EN)

Population trends: almost stable with approximately 600-800 individuals remaining in the wild (Niu K. F., pers. comm., 2018).

Distribution: Fanjing Mountain (south of the Yangtze) in Guizhou Province of China.

Threats: non-targeted hunting, agriculture and infrastructure developing, and disturbances from unrestricted tourism and forest products collection.

Golden (Sichuan) snub-nosed monkey: *R. roxellana* (Milne-Edwards 1870) (EN)

Three subspecies: *R. r. roxellana* (Milne-Edwards 1870) (EN), *R. r. hubeiensis* Wang et al. 1998 (EN), *R. r. qinlingensis* Wang et al. 1998 (EN).

Population trends: decreasing with approximately 12,000 individuals of *R. r. roxellana*, 4,000 of *R. r. qinlingensis* and 1,800 of *R. r. hubeiensis* remaining in the wild (Li B. G., pers. comm., 2018).

Distribution: West and north-east Sichuan provinces, Qinling Mountains of South Shaanxi Province; South Gansu, and Shennongjia in West Hubei Province.

Threats: low-intensity hunting, forest loss and fragmentation due to agriculture, road and infrastructure developing, and unrestricted tourism.

Black-and-white (Yunnan) snub-nosed monkey: *R. bieti* Milne-Edwards 1897 (EN)

Population: increasing, with an estimated 3,000-3,600 individuals remaining (Long Y. C., pers. comm., 2018).

Distribution: Yunling Mountains in south-east Tibet and northwest Yunnan Province.

Threats: low-intensity hunting, forest loss and fragmentation due to agriculture, road and infrastructure developing, and frequent forest products collection.

Black (Myanmar) snub-nosed monkey: *R. strykeri* Geissmann et al. 2011 (CR)

Population: decreasing with an estimated 400 individuals remaining in the wild (Meyer et al., 2017).

Distribution: border region between Myanmar and China between N'mai Hka River and Salween River in Kachin State, North-East Myanmar and North-West Yunnan Province, China.

Threats: hunting, wildlife trading, forest loss and fragmentation due to agriculture, road and infrastructure developing and large scare logging, and frequent forest products collection.

2.5.2 Taxonomy and evolution of snub-nosed monkeys

The evolutionary history of snub-nosed monkeys is quite complex and controversial due to hybridisation among populations and species (Liedigk et al. 2012; Kuang et al., 2018). Based on mitochondrial sequencing, whole genome sequencing and molecular dating, *R. avunculus* diverged first from a common *Rhinopithecus* ancestor approximately 2.4 million years ago (Liedigk et al. 2012; Zhou et al., 2014). The common ancestor of the two Himalayan species (*R. strykeri* and *R. bieti*) diverged from the lineage leading to the two northern species (*R. roxellana* and *R. brelichi*) approximately 2 million years ago. These same data indicate that the latter two species (*R. roxellana* and *R. brelichi*) divided 1.8 million years ago. Afterward, *R. avunculus* had a second contact with the ancestor of the Himalayan species about 0.7 million year ago. The Himalayan species diverged into *R. bieti* haplogroup A and *R. bieti* haplogroup B + *R. strykeri* about 0.6 million year ago, while the latter two spilt most recently, some 0.24 million year ago. After that, both *R. bieti* haplogroups contacted and bred again (Liu et al. 2009).

In *R. roxellana* groups, the ancestral *R. roxellana* population once colonised south and central China (even Taiwan, see Chang et al., 2012) and then suffered a bottleneck 0.18-0.22 million years ago (Kuang et al., 2018). Around 18.9-24.5 thousand years ago, the ancestral *R. roxellana* population reduced and shrank into the *R. r. hubeiensis* (Shengnongjia) population and *R. r. roxellana* (Sichuan-Gansu) population (Kuang et al., 2018). A small founder population then spilt from *R. r. roxellana* population and moved to Qinglin Mountains becoming the *R. r. qinlingensis* population 13.5 thousand years ago (Kuang et al., 2018). The *R. r. qinlingensis* population later outspread about 9.3 thousand years ago, whilst the *R. r. hubeiensis* population experienced a harsh decrease approximately 1.3 thousand years ago (Kuang et al., 2018).

2.5.3 Social organisation of snub-nosed monkeys

All five snub-nosed monkey species are characterised by living in modular social organisations with two basic social units, OMUs (bisexual reproductive individuals, 2-16 individuals) and AMUs (unisexual pre- or post-reproductive units, solitary male; >20 individuals) (Bleisch et al, 1993; Boonratana & Le, 1998; Kirkpatrick et al., 1998; Yang et al., 2002; Kirkpatrick & Grueter, 2010; Dong, 2012; Li et al., 2014; Qi et al., 2014; Wada et al., 2015). A band can be composed of anywhere from 50 to several hundred individuals in anywhere from four to two dozen OMUs with at least one AMU (Kirkpatrick & Grueter, 2010). In one *R. bieti* (in Tianchi, Wang et al., 2019) and one *R. avunculus* (in Tat Ke sector, Dong, 2012) population, band sizes were less than 30 individuals, which is thought to be due to by severe hunting and habitat fragmentation. Recent studies of social connections within and between OMUs reveal that both male and females largely devote their time to social grooming that may promote OMU integrity (Grueter et al., 2017). Meanwhile, females maintain kinship or develop friendships with other females within and across each OMU to help maintain the cohesion of OMUs within a band (Zhang et al., 2012; Wang et al., 2013; Guo et al., 2015; Wada et al., 2015). For example, allomaternal nursing (infants being nursed by females other than their mothers), was found in 87% of infant *R. roxellana* in Shengnongjia and likely arose in tolerant kin-based support networks and thus support reciprocity hypotheses (Xiang et al., 2019). Additionally, although infanticide has been documented in wild *R. roxellana* and wild *R. bieti* societies (Xiang & Grueter, 2007; Ren et al., 2011), it is relatively infrequent given the large band size of most snub-nosed monkeys (50 to 200 individuals) (Grueter, 2013). In the OMUs of *R. roxellana*, females have been reported to show extra-unit copulations (Li et al., 2005; Zhao et al., 2005; Guo et al., 2010; Qi et al., 2020), which may be used as a mechanism of paternity confusion to reduce potential infanticide and inbreeding avoidance (van Schaik, 2000; Guo et al., 2010; Qi et al., 2020). In addition, alliances also appear for the breeding males of OMUs as a means to defend their units from bachelor males' takeover attempts, while bachelor males live in large units to launch collective riots for more breeding opportunities (Xiang et al., 2014; Qi et al., 2017).

The two or more associated breeding bands – which have overlapping home ranges, coordinate travel, transfer OMUs between each other and participate in fission–fusion dynamics – create the herd or troop (Nie et al., 2009; Ren et al., 2012; Qi et al., 2014). Further, both sexes transfer between OMUs of the same band, whereas males dispersing between bands also occur in this multilevel society (Zhang et al., 2006; Zhao et al., 2008a; Qi et al., 2009; Yao et al., 2011; Chang et al., 2014). Such complex social organisation patterns of snub-nosed monkeys have been uncovered in recent years due to provisioning herds of *R. roxellana* in Qingling Mountains and advanced radio tracking and individual-marking technologies (see Qi et al., 2014 and Huang, 2015). Reports of social organisation variations between the five snub-nosed monkey species are few due to the rarity of these species and the difficulties inherent in following monkeys in the steep mountainous area across which they are distributed.

Bands practicing fission-fusion are seen in *R. avunculus* (Boonratana & Le, 1998; Dong, 2012), *R. bieti* (Ren et al., 2012; Grueter et al., 2017), *R. roxellana* (Kirkpatrick et al., 1999; Qi et al., 2014), and *R. brelichi* (Bleisch & Xie, 1998; Yang et al., 2002; Nie et al., 2009). Field observations show that some subgrouping events occur during the day and then individuals reunite when they return to their sleeping sites (Bleisch et al., 1993; Boonratana & Le, 1998; Kirkpatrick et al., 1999; Dong, 2012; Grueter et al., 2017; Niu Kefeng, pers. comm., 2018). In *R. brelichi*, mating opportunities and genetic exchange is thought to be a benefit of relatively long-term (3 days) bands forming large herds (>400 individuals) in autumn (Yang et al., 2002; Nie et al., 2009), which also have been recorded in *R. roxellana* (Ren et al., 1998; Qi et al., 2014). However, the fact that the large herds never form in winter is evidence that food resources may be the limitation to super herd formation (Bleisch & Xie, 1998; Nie et al., 2009). A band can also divide and subbands of these snub-nosed monkeys may remain apart for weeks to months (Kirkpatrick, 1998; Nie et al., 2009; Grueter et al., 2017). In the case of *R. bieti*, fission behaviours are recorded for a large AMU in a part of its home range (in Xiangguqing forest) where the density of their year-round food (lichens) is low and the high-quality food (bamboo shoots) dramatically declines seasonally (Ren et al., 2012). Grueter (2009) suggested that fissioning in the same species in the Samage forest may arise from valued winter fruits becoming rare and increasingly patchy. For the *R. bieti* bands in the species' northern-most distribution ranges (e.g. Wuyapiya), there is a heavy reliance on a year-round stable food, lichens, and fission - fusion is much rarer in these bands (Kirkpatrick et al., 1998). Therefore, fission may be a strategy used by snub-nosed monkeys to allow the flexibility to respond to changes of food availability and seasonal diet, and hence reduce costs of assembling in large bands.

From the literature reviewed in this chapter, we can see that research on colobines has been deeply and broadly developed in the area of evolutionary history, feeding ecology ranging behaviours, and social organisation in the past twenty years. Colobine monkeys provide us a window to understanding some key questions such as 'what are the ecological or genetic determinants that assisted colobines radiation and spread into forests in tropical area to subalpine mountains?' (Tran, 2014; Yu et al., 2016), 'What ecological determinants limit the abundance of species?' (Chapman et al., 2002; Teelen, 2007), and 'How are complex multilevel societies formed in primates?' (Yeager & Kirkpatrick, 1998; Qi et al., 2017). In the next chapter, I will specifically review feeding ecology and ranging behaviours of snub-nosed monkeys and use a meta-analysis to further examine these behaviours over the past two decades in an to attempt to answer the following three questions: 'Is lichenivory developed by snub-nosed monkeys as a determinant to radiation of this genus?' 'Do the temperate snub-nosed monkeys follow the ecological-constraints model?' and 'Do the temperate snub-nosed monkeys have a regular trend in seasonal daily range length and seasonal home range size?'.

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Chapter 3 Lichenivory and Ecological-Constraints in (Temperate) Snub-Nosed Monkeys

3 Introduction

Understanding the set of adaptations that primate species have evolved to exploit a range of habitats is key to understanding a species' biology, ecology, and conservation. Snub-nosed monkeys (*Rhinopithecus* spp.) represent an adaptive radiation of five species of Asian colobines that exhibit a suite of behavioural, anatomical, and physiological traits that include a shift in dietary emphasis from frugivory to folivory to lichenivory. They possess a more robust mandible for the mechanical breakdown of tough foods, somatic cells that are more resistant to UV irradiation, and enhanced capillary development in high-elevation species (black-and-white snub-nosed monkey *R. bieti*, black snub-nosed monkey *R. strykeri* and golden snub-nosed monkey *R. roxellana*) that enable them to successfully exploit a range of habitat types ranging from the limestone tropical rain forests of Vietnam to the mixed conifer-deciduous forest or subalpine conifer forest in the Hengduan mountains in China and Myanmar (Kirkpatrick, 1995; Yang et al., 2002; Pan et al., 2008; Kirkpatrick & Grueter, 2010; Dong, 2012; Yu et al., 2016; Yang et al., 2019). Compared to other Asian colobines, all species of snub-nosed monkeys form a large modular society that can consist of 62-450 individuals and occupy an annual home range of 8.2-32 km^2 (Table Appendix II). Unlike many other species of colobines, the peculiar feeding habit of lichenivory (with an average of 45% feeding time) shown by temperate snub-nosed monkey have allowed them to adapt to high altitude alpine environments. Given that the radiation of snub-nosed monkeys is closely associated with the uplifting of the Himalayas and the Tibetan Plateau, these primates provide an important comparative model offering insight into the adaptive solutions required to successfully exploit high altitude, cold, temperate forests.

I begin this chapter with a literature review of the feeding and ranging patterns of snub-nosed monkeys. Then I present data from the published literature and conduct a meta-analysis of information on the diet, daily range length, and home range area utilised by each of the five snub-nosed monkey species (*R. avunculus*, *R. bieti*, gray snub-nosed monkey *R. brelichi*, *R. roxellnana*, *R. strykeri*) to examine three questions. The first is the function of lichenivory in the adaptive radiation of snub-nosed monkeys exploiting mountainous, high altitude habitats. The second is whether temperate snub-nosed monkeys (*R. bieti* and *R. roxellana*) follow the ecological-constraints model, in which individuals in larger bands encounter increased intragroup feeding competition – measured through increased daily range length and increased home range – compared to individuals living in smaller bands (Gillespie & Chapman, 2001).

3.1 A Review of the Feeding Ecology of Snub-Nosed Monkeys

3.1.1 Dietary diversity, dietary selectivity and seasonal dietary patterns

Snub-nosed monkeys (*R. bieti*, *R. roxellana* and *R. brelichi*) exhibit larger mandibular dimensions (including width of the mandibular corpus and molars), bigger moment arm of the masseter, and stronger masticatory muscles than other Asian colobines (e.g. *Trachypithecus* spp., *Presbytis* spp.) or macaques (*Macaca* spp.) (Pan & Oxnard, 2003; Pan et al., 2008). This results in greater masticatory forces to chew and shred tough foods such as twigs, bark, and lichen. Similar to other colobine monkeys, snub-nosed monkeys have enlarged salivary glands and a relatively long gastrointestinal tract including a multi-chambered forestomach that maintains a pH of 5-7 (low acid stomach) and a capacious caecum and proximal colon for enhancing bacterial fermentation processes (Chivers, 1995;

Lambert, 1998). Asian colobines are characterised by duplication of specific genes such as colobine RNASE1 that function to enhance digestive efficiency at low pH, extract energy from fatty acids, and degrade xenobiotics from foliage (Zhou et al., 2014). This allows snub-nosed monkeys to more effectively hydrolise and absorb cellulose, hemicellulose, non-protein nitrogen and lichenin, and detoxify plant secondary compounds such as tannins and phenols (Kirkpatrick et al, 2001; Nijboer, 2006; Bissell, 2014).

Interspecific differences in the diet of snub-nosed monkeys relate to differences in the ecology of each species. For example, *R. avunculus* inhabit tropical evergreen broadleaf forests in limestone mountains at an elevation of 200 m, and principally consume fruits and young leaves. *R. bieti* inhabit temperate conifer forests in Tibetan alpine areas up to an elevation of 4,600 m, and principally consume lichens all year round (Dong, 2012; Kirkpatrick, 1996).

Diet of the Tonkin snub-nosed monkey

The habitat of the *R. avunculus* is dominated by tropical rainforest species, such as *Ficus* spp. and *Laurus* spp. (Nadler et al., 2007). Early observations at the Khau Ca area in northern Vietnam indicated that more than half of their feeding time is spent on fruits and seeds (52.7%), followed by leaves (33.31%) and other vegetative parts (11.11%) (Le, 2007). A more recent study at Khau Ca suggests that 53.1% of *R. avunculus*'s feeding time is dedicated to leaves (of which 46.2% are young leaves), followed by ripe fruits (13.8%), unripe fruits (11.2%), flowers (11.2%), and seeds (7.2%) (Dong, 2012). This study also examined seasonal shifts in feeding behaviour and found that the consumption of young leaves by *R. avunculus* is negatively correlated with the availability of fruits, flowers, and seeds in autumn and winter (Dong, 2012) (Table 3.2).

Diet of the gray snub-nosed monkey

Habitats presently exploited by *R. brelichi* are dominated by subtropical evergreen broadleaf forests or evergreen-deciduous broad leaf forests with common plant taxa that include *Cantanopsis* spp., *Lithocarpus* spp., *Fagus* spp., and *Schima* spp. (Xiang et al., 2009). These primates range from altitudes of 800-2,300 m (Xiang et al., 2009). There are currently roughly 700 *R. brelichi* individuals remaining in the wild, and therefore it is uncertain how closely their current habitat and distribution coincides with their historical distribution. Winter temperatures in the Fanjing Mountain can drop to below zero and snow falls sporadically (Bleisch et al., 1993; Wu et al., 2004). Xiang et al. (2012) report a diet of leaves (25.5% young and 21.8% mature) fruits/seeds (21.6%), buds (15.3%), and flowers (9.4%) at Fanjing Mountain. Other foods ingested included bamboo shoots, bark, and arthropods (Bleisch & Xie, 1998; Xiang et al., 2012; Xiang et al., 2013a). *R. brelichi* concentrates its diet on plant parts and plant species from only five or six genera in each season. For example, the monkeys feed on the bark and flowers but not the buds and leaves of *Rhododendron* (Ericaceae) (Xiang et al., 2012). In summer, the unripe fruits/seeds of *Dendrobenthamia angustata* account for 25% of feeding time. In autumn to early winter, the leaves (47%) and flower buds (28%) of *Magnolia* spp. were the most common items consumed (Bleisch & Xie, 1998). Similar to the *R. avunculus*, *R. brelichi* shifts its dietary emphasis from flowers (34.1%) in March and April to young leaves (71.4%) in May to mature leaves (44.6%) in November, to ripe fruits and seeds (65.2%) in September, and to buds (61.2%) in January (Xiang et al., 2012).

Table 3.1 Information about food plant part and their proportion consumed by *Rhinopithecus* spp. in different study sites

Species	**Study site**	**The proportion of diet composition (%)**						**Other food items**	**Band size**	**Altitude range (m)**	**Forest type**	**Reference**
		Young/Mature leaves	**Fruits/ Seeds**	**Flowers**	**Buds**	**Bark**	**Lichens**					
R. avunculus	Khau Ca	46.2/5.9	24/7.2	11.2	-	-	0	-	22-81	600-1,300	Tropical evergreen broad leaf forests	Dong, 2012
R. brelichi	Fanjing Mountain	25.5/21.8	21.6	9.4	15.3	Recorded in winter	0	Bamboo shoots & invertebrates	450	1,300-2,000	Subtropical evergreen and deciduous broad leaf forests	Xiang et al., 2012
R. roxellana	Shennongjia	13.5/8.9	9.5/20.8	1.3	5.8	0.2	38.4	Insects	236 ± 38	1,550-2,663	Temperate evergreen/deciduous broadleaf forest - mixed - Conifer forest	Liu et al., 2013b
	Shennongjia	28.71/3.51	14.57	1.13	5.36	1.36	43.28	Herbs	120-140	1,700-2,630	Temperate evergreen/deciduous broadleaf forest - mixed - Conifer forest	Li, 2006
	Zhouzhi East (N Qinglin)	24.0	29.0	–	4.2	11.1	24.0	Unidentified	112	1,400-2,900	Subtropical & temperate evergreen/deciduous broadleaf forest - mixed - conifer forest	Guo et al., 2007
	Zhouzhi West (N Qinglin)	37.2	22.5	2.4	16.2	13.5	1.3	Arthropods, mushroom, clay	135	1,400-2,600	Temperate deciduous broadleaf forest; Deciduous broadleaf & coniferous mixed forests	Hou et al., 2018
	Zhouzhi West (N Qinglin)	35.92	21.42	7.52	18.44	8.07	0.95	Invertebrates, fungi & clay	232	1,400-2,600	Temperate deciduous broadleaf forest; Deciduous broadleaf & coniferous mixed forests	Huang, 2015

	Qingmuchuan (S Qinglin)	25.0	25.94	–	21.3		0	–	100-120	800-2,054	Subtropical & temperate evergreen/deciduous broadleaf forest - mixed - conifer forest	Li & Jiang et al., 2010
	Guanyinshan (S Qinglin)	9/11	9/16	–	11	15	22	Petioles, mushroom	70	1,150-2,574	Temperate deciduous broadleaf forest - mixed - Conifer forest	Zhao et al., 2015
R. bieti	Xiaochangdu	12.1	1.1	1.1	– [a]	4.2	82.1	Invertebrates, resin, & herbs	210	3,500-4,250	Temperate evergreen/deciduous broadleaf forest - mixed - Conifer forest	Xiang et al., 2007
	Wuyapiya	6.0	0.1	Occasionally recorded	-	Occasionally recorded	85.0	-	175	3,300-4,600	Temperate evergreen/deciduous broadleaf forest - mixed - Conifer forest	Kirkpatrick, 1996
	Tacheng	8.4/16.3	10.5	1.9	3.0	0.8	50.6	Bamboo shoots (7.9%), petiole, stem, fungi, bird and bird eggs & insects	480	2,600-4,100	Bamboos, temperate evergreen/deciduous broadleaf forest - mixed - conifer forest	Li et al., 2011a
	Samage	12.4/4.1	11.4	0.2	3.6	0.5	67	Petiole, Pith, bamboo shoots, mushrooms, tubers, bird eggs, &flying squirrel	410	2,500-4,000	Subtropical & temperate evergreen/deciduous broadleaf forest -mixed - conifer forest	Grueter et al., 2009a, b
	Fuhe [b]	79.7	4.8	2.3	–	–	13.2	Bamboo leaves (23.9%) & Bamboo shoots	80	2,700–3,600	Bamboos, Subtropical & temperate evergreen/deciduous broadleaf	Liu et al., 2004

											forest - mixed - conifer forest	
	Lasha Mt.	9.1 (with bud)	7.3	1.5	–	-	80.2	Insects (1.9%); Bamboo shot	> 100	2,800-3,600	Deciduous broadleaved forest-deciduous-conifer forest-conifer forest	Huang et al., 2017

Note: a. Being included with leaves. b. Using micro-histological analysis of feces, not recorded by feeding time.

Table 3.2 Information about food selection and food seasonality of *Rhinopithecus* spp. in different study sites.

Species	**Study Site**		**Observation time**	**Dietary diversity**	**Predominant dietary (proportion of preferred foods or frequently selected foods in feeding records)**	**Food selection varied according to seasonality and local availability**	**Reference**
Rhinopithecus avunculus	Vietnam	Na Hang NR	October 2004 - August 2005 (11 mo)	50 species in 25 families	6 species (71%)	Young leaves (year-round); Flowers (February - May); Fruits & seeds (October - January)	Dong, 2012
		Khau Ca HSCA	September 2005 - September 2006 (13 mo)				
Rhinopithecus brelichi	Fanjingshan National Nature Reserve, China		January 2006 - July 2008 (> 12 mo)	107 species in 28 families	Not available	Buds (September - January); Young leaves (February - May); Mature leaves (May - November); Flowers (March - April); Fruits & seeds (June - October)	Xiang et al., 2012
Rhinopithecus roxellana	Shennongjia Nature Reserve, China		July 2003 - September 2004 (14 mo)	51 species in 24 families	4 species (35.1%)	Lichens (November - April); Leaves & lichens (May - July); Fruits, seeds & lichens (August - October)	Li, 2001; Li, 2006
	Shennongjia Nature Reserve, China		August 2006 - July 2008 (20 mo)	53 plant species (6 lichen species)	Fruticose lichens (38.4%)	Flowers (March - April); Young leaves (April - July); Mature leaves (May -September), Fruits (June - October); Seeds (September - March); Buds (December - April); Fruticose lichens (year-round)	Liu et al., 2013b
	Qingmuchuan Nature Reserve, China, Southern slopes of Qinglin		November 2006 - July 2007 (9 mo)	40 species	Winter: 24 species; Summer: 20 species.	Frugivorous (summer) - Folivorous (winter)	Li and Jiang et al., 2010
	GuanyinShan, Southern slopes of Qinglin		March 2013 - February 2014	53 species in 37 families	21 species (36.9%); lichens (22%); leaves (20%); fruits/seeds (25%)	Young leaves & buds (spring); mature leaves (summer); fruits/seeds (autumn); lichens (winter)	Zhao et al., 2015

	Zhouzhi West, Northern slopes of Qinling	November 2001 - October 2003 (> 10 mo)	84 species in ≥ 29 families	Lichens (62.3%, *Ramalina sinensis & Parmelia spp*.).	Lichen & seeds (winter); Lichen, barks, buds & young leaves (spring); Seeds and mature leaves (summer); Lichen and seeds (winter).	Guo et al., 2007
	Zhouzhi East, Northern slopes of Qinglin	July 2012-June 2013 (12 mo)	82 plant species	25 plant species (88.7%, five preferred species 23.1%)	Leaves, buds & Twigs/Flowers (spring); Fruits/seeds & Leaves (summer-autumn); Ripe fruits/seeds, bark & buds (winter)	Hou et al., 2018
	Zhouzhi East, Northern slopes of Qinglin	September 2013-Janerary 2015 (19 mo)	129 species in 43 families	Plants in 15 families (88.7%)	Leaves & flowers (spring); Leaves & Fruits/seeds (summer); Fruits/seeds (autumn); Fruits/seeds, bark& buds (winter)	Huang, 2015
	Baihe Nature Reserve, China	Since June 1996 (12 mo)	> 38 species	Lichens (Usneaceae), mature leaves (Rosaceae)	Leaves (summer); Lichens, buds & bark (winter)	Kirkpatrick et al., 1999; Kirkpatrick & Grueter, 2010
Rhinopithecus bieti	Xiaochangdu, Tibet of China	June 2003 - March 2005 (13 mo)	25 species in 13 families	Lichens (82.1%); Buds & leaves (15.9%)	Lichens (May - February); Buds & leaves (March - August); Flowers (spring); Unripe & ripe fruits (summer)	Xiang et al., 2007
	Wuyapiya, Yunnan of China	May 1992-June 1994 (> 12 mo)	22 species in ≥ 12 families	Lichens (86%, 75% on arboreal lichens)	Lichens (year-round); Buds & leaves (spring - summer); Ripe fruits (summer)	Kirkpatrick, 1996
	Samage, Yunnan of China	September 2005-July 2007 (20 mo)	94 species in 38 families	Lichens (67%, 90% *Usnea spp*.). Top 10 tree species (> 90% in plant diet).	Bamboo shoot (June - July); Lichen and leaves (year-round); Fruits (summer, fall, & winter); Buds (winter).	Grueter et al., 2009a, b
	Tacheng, Yunnan of China	March - December 1999 (9 mo)	59 plant species in 28 families	Lichens (60%); bamboo leaves/shoots (43% in fecal)	Lichens (winter - spring); buds (spring); Dicot & bamboo leaves, shoots (summer - early autumn)	Ding & Zhao, 2004
	Tacheng, Yunnan of China	June 2008 - May 2009 (12 mo)	105 species in 42 families	Lichens (50.6%); Leaves (24.7%, 17.6% on Rosaceae)	Young leaves (spring); Bamboo shoot & mature leaves (summer); Fruits & seeds (fall); Lichens & buds (winter)	Li, 2010, Li et al., 2011a

	Jingsichang, Yunnan of China	December 1997- October 1998 (6 mo)	Not available	Bamboo leaves (59%); Dicot leaves (28%)	Bamboo leaves & lichens (year-round); Dicot leaves (summer); Seeds (autumn)	Yang & Zhao, 2001
	Fuhe, Yunnan of China	December 2000 - January 2002 (> 12 mo)	Not available	Broad leaves (50.1%)	Broad leaves & lichens (year-round); Bamboo leaves & shoots (winter - spring); fruits (Summer- Autumn)	Liu et al., 2004
	Mt. Lasha, Yunnan of China	January 2008 to September 2010 (33 mo)	36 species	Lichens (80.2%)	Lichens (year-round)	Huang et al., 2017

Diet of the golden snub-nosed monkey

R. roxellana exploits subtropical forests at an elevation of 1,500 m to high elevation temperate forests at 4,100 m in the Qing-Bashan Mountains of southwestern China (Kirkpatrick & Grueter, 2010). They are distributed in three isolated populations with a total of approximately 17,800 individuals in the wild (Li B. G., pers. comm., 2018). Different from the tropical and subtropical species, some *R. roxellana* found in deciduous forests, mixed deciduous broadleaf, and coniferous forests include lichen as an important dietary component, varying from 1-43% of total feeding time (Li, 2001; Guo et al., 2007, Huang, 2015). For example, fruit/seeds (29.4%), lichen (29.0%), leaves (24.0%), bark (11.1%), buds (4.2%), and twigs (1.3%) represent the primary dietary components in a population on the northern slopes of the Qingling Mountains (Guo et al., 2007). In Qingmuchuan, *R. roxellana* feed on 20 plant species in summer, with 72.2 % of their foraging activities concentrated on the fruits of a single tree species, the giant dogwood (*Cornus controversa*) (Li and Jiang et al., 2010). Also, of 23 plant species fed on by *R. roxellana* in Shennongjia, six broadleaf deciduous species accounted for 42.6% of feeding time, and the lichens of nine tree species accounted for 43.3% of feeding time (Li, 2006). In the Mingshan-Qinling-Bashan Mountains, foods varied markedly across months (Kirkpatrick et al., 1999; Li, 2006; Guo, 2007, Li et al., 2010; Hou et al., 2018). At Guanyingshan, mature leaves and fruits/seeds are fed on principally in summer (29%) and autumn (64%), while lichen (41%) and bark (15%) are fed on in winter. Under adverse weather conditions in winter – including freezing temperatures and blizzards (Zhang et al., 2011) – *R. roxellana* at Shennongjia increase their consumption of lichen (from 35.9% to 45.3%), buds (from 0% to 32.1%), and bark (from 0% to 1.2%) compared to summer (Liu et al., 2013). When lichen is less available at Zhouzhi West (Yuhuangmiao), the winter diet of this northern population includes a marked increase in the consumption of bark (from 2.6% to 49.5%) (Huang, 2015).

Diet of the black-and-white snub-nosed monkey

R. bieti is currently restricted in its distribution to the Yunling Mountains, which border the Himalayas, and range from 2,600 m to 4,600 m in elevation (Kirkpatrick, 1996; Li et al., 2008; Xiang et al., 2013). Habitats exploited by *R. bieti* are characterised by temperate deciduous broadleaf forest to mixed deciduous broadleaf and coniferous forest. There are estimated to be less than 3,600 *R. bieti* individuals remaining in the wild (Long Y. C., pers. comm., 2018). Similar to *R. roxellana*, the diet of *R. bieti* in mixed deciduous broadleaf and coniferous forest habitat consists of lichens (50.6%), leaves (24.7%), fruits/seeds (10.5%), flowers (1.9%), buds (3%), and bark (0.8%) (Li et al., 2011a). Other foods such as mushrooms, corms, resin, insects, birds and their eggs, and small mammals are occasionally eaten (Xiang et al., 2007; Ren et al., 2008; Grueter et al., 2009a; Zhao et al., 2008; Zhao et al., 2009). Zhao et al. (2009) found that the Tacheng's *R. bieti* population is accustomed to villagers who frequently appear in their habitats, and therefore range at lower elevations (from 3000 m to 2700 m). When they descend to these lower elevations they broaden the number of plant species in their diet (from 59 to 113 species) (Zhao et al., 2009). *R. bieti* at Samage consume food from 43 out of a total of 89 available tree species, however, just 10 tree taxa accounted for 91.6% of yearly plant feeding time (Grueter et al., 2009a).

Changes in plant species diversity and distribution moving from south to north across the Yunling Mountains – which are characterised by gradually decreased plant biodiversity and forest productivity – are associated with population changes in *R. bieti* feeding behaviour. For example, in their central distribution in the Baima Snow Mountains, *R. bieti* devote more feeding time to bamboo leaves (59%, based on per feceal sample levels) than in their southern-most distribution in the Fuhe and Longma Mountains (43%) (Yang & Zhao, 2001; Ding & Zhao, 2004; Liu et al., 2004; Grueter et al., 2009a). The smallest number of plant species consumed by *R. bieti* occurs in their northern-most distribution (Xiaochangdu: 25 species over 13 families, Xiang et al., 2007; and Wuyapiya: 22 species over 12 families, Kirkpatrick, 1996). At lower elevations and in more southern habitats there is an increase in the number of species consumed (Samage: 94 species in 20 families, Grueter et al., 2009a; Longma Mountain: 97 species of 27 families, Huo, 2005) (Table 3.3). In contrast, the monkeys' annual feeding time on lichen shows a decreasing gradient from over 80% at Wuyapiya (Kirkpatrick, 1996) and Xiaochangdu (Xiang et al., 2007) in the northern-most distribution of their range, to 67% at Samage (Grueter et al., 2009b) in their central distribution, and to 63% in the Fuhe Mountain (Liu et al., 2004) which lies at their southern-most distribution. Snub-nosed monkeys increase lichenivory (from 63% to 86%) while decreasing folivory (from 24.7% to 12.1%) from low elevation to high elevation (Table 3.1, 3.3).

Table 3.3 Food diversity and feeding effort of lichens of *Rhinopithecus bieti*

Band	Median Latitude	Median Longitude	Median Elevation	Food species diversity		Feeding time on lichens (%)	Reference
				Species	Family		
Wuyapuya	28°30'	99°12'	3,950	22	12	86	Kirkpatrick, 1996
Xiaochangdu	29°15'	98°37'	3,875	25	13	82.1	Xiang et al., 2007
Tachen	27°36'	99°18'	3,150	105	42	50.6	Li et al., 2011
Samage	27°34'	99°17'	3,300	94	20	67	Grueter et al., 2009a
Fuhe	26°25'	99°20'	3,100	97	27	63	Liu et al., 2004; Huo, 2005
Lasha[a]	26°20'	99°15'	3,200	36	NA	80.2	Huang et al., 2017

a. Food diversity constrained by degraded forest fragment

Seasonal variation in the diet of *R. bieti* has been reported in the Yunling Mountains, with young leaves/lichen (30.8%/49.1%), mature leaves/bamboo shoots/lichen (25.1%/26.1%/32.3%), fruits+seeds/lichen (27.9%/53.9%), and lichen/buds/leaves (75.7%/10.3%/10%) representing the major components of the diet during spring, summer, autumn and winter, respectively (Li et al., 2011a).

Exceptions to this pattern occur at high altitudes and in the northern edge of the species range where their dietary diversity is more limited. In this region, lichens account for approximately 82-86% of total annual feeding time (Kirkpatrick, 1996; Xiang et al., 2007). It appears that *R. bieti* obtains a nutritionally-adequate diet by switching patterns of food intake across seasons. Van Schaik et al. (1993) has hypothesised that primates regulate their patterns of ranging and altitudinal movements to ensure a nutritionally-balanced diet. In the case of *R. bieti*, this is accomplished by them moving along an ascending altitudinal gradient for newly-emerging foliage during spring-summer and only visiting coniferous forest at higher elevation for lichens in winter (Li et al., 2008).

Diet of the black snub-nosed monkey

R. strykeri ranges across semi-moist evergreen broad-leaved forests and temperate hemlock coniferous forests at elevations of 1,700-3,600 m (Geissmann et al., 2011; Ma et al., 2014). There are approximately 400 *R. strykeri* individuals remaining in the wild (Meyer et al., 2017). Prior to this dissertation, the sole dietary information for wild *R. strykeri* was from a video recording that included the animals ingesting food from three plants: *Dodecadenia grandiflora*, *Schefflera* sp. and *Eurya* sp. (Yunnan TV, 2013). According to their geographic distribution and forest type, *R. strykeri*'s dietary pattern may be most similar to *R. bieti*.

3.1.2 Nutritional chemistry

During food selection, primates balance nutrient intake from a diverse set of nutritionally-distinct plant species and plant tissues (Felton et al., 2009; Guo et al. 2018). In addition, factors such as digestive efficiency – which is affected by the species gut microbiome and gut physiology – and limiting the intake of plant secondary metabolites, directly affect food choice (Milton, 1979, 1998; Chapman et al., 2012). In the case of snub-nosed monkeys, there is evidence that young leaves, flowers, and fruits/seeds (which are nutritionally characterised by higher protein, lower fibre, or increased water soluble carbohydrate content) are consumed more frequently than mature leaves (which are generally characterised by lower protein, reduced water content, and higher fibre compared to young leaves) when available (Bleisch et al., 1998; Hou et al., 2018). In *R. roxellana* and *R. bieti* inhabiting temperate and coniferous forests, the seasonal availability of young leaves, flowers, fruits, seeds and bamboo shoots is negatively correlated with the proportion of lichen in the diet (Li, 2006; Guo et al., 2007; Grueter et al., 2009b). Folivorous primates generally consume leaves higher in protein but lower in fibre compared to leaves present but not consumed (Chapman et al., 2012; Ganzhorn et al., 2017). However, the precise factors affecting food choice in this primate genus remain unclear. Studies of nutritional chemistry for *R. avunculus* (Lan Anh et al., 2014), *R. brelichi* (Bleisch et al., 1998), *R. roxellana* (Hou et al., 2018) and *R. bieti* (Li & Yang, 2009; Zhang et al., 2013; Li et al., 2019) demonstrate that the consumption of leaves from seasonal plants is positively correlated with the protein content. However, the leaves consumed by *R. bieti* in the Longma Mountain (Huang et al., 2010) and *R. roxellana* at Shengnongjia (Liu et al., 2013) did not differ in crude protein from leaves that were not consumed. Studies by Bleisch et al., (1998), Huang et al., (2010) and Hou et al. (2018) indicate that some snub-nosed monkey populations (for example, *R. brelichi*), select leaves with a higher protein-to-fibre ratio (0.344) than leaves not consumed (0.260). However, there are no significant differences in fibre content between consumed and non-consumed leaves by *R. brelichi* in Fanjing Mountain (neutral detergent fibre, acid detergent fibre and lignin) (Bleisch et al., 1998), *R. roxellana* in Shennongjia (crude fibre) (Liu et al., 2013) and in Qinling (neutral detergent fibre and acid detergent fibre) (Hou, 2018), *R. bieti* in Longma Mountains (neutral detergent fibre) (Huang et al., 2010), and *R. avunculus* in Khau Ca (acid detergent fibre) (Lan Anh et al., 2014). This means the higher protein-to-fibre ratio of consumed leaves is mainly a matter of protein content. In addition, several studies report that ingested leaves are richer in ash (minerals) (Bleisch et al., 1998; Huang et al., 2010; Zhang et al., 2013; Hou et al., 2018) and lower in antifeedants (e.g. tannin and phenols) than leaves that are not consumed (Li & Yang, 2009; Zhang et al., 2013; Guo et al., 2016). Other seasonally-preferred foods, such as fruits and seeds, are generally rich in water-soluble carbohydrates, fat and metabolisable energy (Liu et al., 2013b; Hou et al., 2018). For example, the fruits/seeds consumed by *R. roxellana* at Shennongjia contain higher fat (18.91-57.72%) than other plant parts and lichens (<5.10%) (Liu et al., 2013b). Per unit consumed, fats provide two times more energy than proteins or carbohydrates (El Bacha et al., 2010). For captive *R. bieti*, despite lichens being lower in protein (4.7-6.1% in lichens vs. 15.2-27.8% in leaves), they are higher in digestibility [mean digestibility of lichen *Usnea longissima* vs. neutral detergent fibre (NDF) is 81.3% vs. 64%] and contain more non-structural carbohydrates compared to leaves (14.3% in lichens vs. 1.9-4.9% in leaves) (Kirkpatrick et al., 2001; Bissell, 2014). Given that lichens are consumed more commonly in the winter (75.7%) than in spring (49.1%), summer (32.3%) and autumn (53.9%) they may provide the additional energy required for thermoregulation during an extended and cold winter (Kirkpatrick et al., 2001; Li et al., 2011a; Bisell, 2014; Zhao et al., 2020).

3.2 A Review of Ranging Behaviours of Snub-Nosed Monkeys

More than 100-400 snub-nosed monkeys can form a single herd. This occurs when two or more bands temporarily join each other (Qi et al. 2014). Band size is commonly 20-100 individuals in tropical and subtropical forest-dwelling snub-nosed monkeys (*R. avunculus*: 20-80 individuals; *R. brelichi*: 30-100 individuals), which is smaller than those reported for temperate snub-nosed monkeys, who live in bands of 60-360 individuals (*R. roxellana*: 60-280 individuals; *R. bieti*: 80-360 individuals) (Ren et al., 2000; Yang et al., 2002; Nadler et al., 2007; Nie et al., 2009; Qi et al., 2014; Table Appendix II). Ren et al. (2016) report that the super-large *R. bieti* group (480 individuals) at Samage is actually composed of two monkey bands with some overlapping home ranges. Also, the home ranges of temperate snub-nosed monkeys (*R. bieti*: 8.2-25.3 km^2; *R. roxellana*: 12.4-22.1 km^2) appear to be larger than the tropical subtropical species (*R. avunculus*: 3.7-7 km^2; data not available for *R. brelichi*) (Appendix II, Niu et al., 2010; Thach & Covert, 2012; James, 2016). In contrast, daily range length does not appear to differ between the temperate snub-nosed monkey species and the tropical-subtropical species (*R. bieti*: 765-

1,514 m; *R. roxellana*: 1,100-2,100 m; *R. brelichi*: 523-1,672 m; *R. avunculus*: 550-1,470 m) (ibid).

According to the ecological constraints model, band size is expected to be positively correlated with daily range length and home range area (Gillespie & Chapman, 2001). This relationship has been reported in some Asian and African colobines (e.g. Thomas's langur *Presbytis thomasi*, Steenbeek & van Schaik, 2001; western red colobus *Procolobus badius*, Gillespie & Chapman, 2001; Chapman & Pavelka, 2005; black-and-white colobus *Colobus vellerosus*, Teichroeb & Sicotte, 2009; mantled guereza *C. guereza*, Harris et al., 2010). A pattern of increased home range area and increased band size has also been found in temperate snub-nosed monkeys. For example, at Shenglongjia, the large *R. roxellana* band (205 individuals) had a larger summer home range (4.3 vs. 2.8 km^2) and larger winter home range (2.9 vs. 1.84 km^2) than the small band (104-114 individuals) (Li, 2004). A follow up study conducted seven years later also found that the larger band (236 individuals) maintained a larger annual home range (22.4 vs. 12.4 km^2) than the smaller band (63 individuals) (Fan et al., 2019). At Samage, *R. bieti* gradually extended their home range from 7.67 km^2 to 17.14 km^2 over a nine-year period (1998-2007) as band size increased from 160 to 450 individuals (Li et al., 2011b).

Contrary to this pattern, in other populations home range area was not positively correlated with band size. At Yuhuangmiao, the home ranges of two neighbouring *R. roxellana* bands overlapped and were stable over a period of 12 years with no changes in home range associated with changing band sizes (Su et al. 1998; Li et al., 2000; Tan et al., 2007). At this site, East Ridge band (112 individuals) had a smaller home range (18.3 km^2) (Li et al., 2000) than the West Ridge band, which contained 90 individuals (22.5 km^2) (Tan et al., 2007). Similarly, although *R. bieti* at Samage are best described as a large herd (> 450 individuals in two bands), its annual mean home range over 10 years was estimated to be 14.8 km^2, which was almost half the size of the Wuyapiya band’s range (25.25 km^2) that contained 175 individuals (Kirkpatrick et al., 1998; Grueter et al., 2008; Li et al., 2011b; Ren et al., 2016). This difference may be explained by other environmental and social factors that affect ranging, including provisioning, proximity to human settlements, habitat fragmentation, resource distribution and availability, predation risk, accessible travel routes in mountainous terrain, and the frequency that bachelor males attempt to take over the breeding position of an OMU.

When exploring daily range length, little evidence can be found of a positive correlation with band size. In *R. roxellana*, the large Qingmuchuan band (120 individuals) inhabiting mixed evergreen and deciduous broadleaf forests had a shorter mean daily range length (837 m vs. 2,100 m) than the East Ridge band (112 individuals) inhabiting mixed deciduous and conifer forest in Zhouzhi (Tan et al., 2007; Li & Jiang et al., 2010). Similarly, in Shengnongjia, when the band size increased from 113 to 129 individuals, the daily range length (1,082 ± 434 m vs. 1,113 ± 401 m) did not increase (Li et al., 2005). In mixed deciduous broadleaf and conifer forest in Tibet, the daily range length of a larger *R. bieiti* band (210 individuals, 765 m) at Xiaochangdu do not have longer path lengths than the band at Wuyapiya (175 individuals, 1,131 m) (Kirkpatrick et al., 1998; Xiang et al., 2013).

Although the current data are very limited, there does not appear to be a strong relationship between band size, daily range length, and home range in temperate snub-nosed monkeys, and thus the ecological constraints model for snub-nosed monkey is questioned. Moreover, given their relatively large group size compared to other Asian colobines such as the Maroon langur (*Presbytis rubicunda*, group size = 7 individuals, daily range length = 1,645 m, home range = 1.08 km^2; Smith et al., 2013); proboscis monkey (*Nasalis larvatus*, group size = 17 individuals, daily range length = 799 m, home range = 1.38 km^2; Matsuda et al., 2009), Kashmir gray langur (*Semnopithecus ajax*, group size = 74/177 individuals, daily range length = 1,230/1,750 m, home range = 0.03/0.02 km^2; Minhas et al., 2013), and northern plains gray langur (*S. entellus*, group size = 30 individuals, daily range length = 1,500 m, home range = not available, Sayers & Norconk, 2008), snub-nosed monkeys appear to exploit a larger sized home range than other Asian colobines (see Table 2.3 in Chapter 2).

The altitudinal ranging patterns of snub-nosed monkeys appear to be driven primarily by food abundance and distribution. Seasonal altitudinal ranges have been described for the West Ridge band at Yuhuangmiao: they forage at higher elevations in summer and autumn and move to lower elevations in winter and spring (Li et al., 2000). This pattern was not found for the neighbouring East Ridge band, whose activities are mainly concentrated at the elevation of 2,137 ± 171 m (Tan et al., 2007). The moving records show that they have the narrowest elevational ranges in autumn but widest in spring (Tan et al., 2007). Evidence of seasonal altitudinal migration for other snub-nosed monkeys is not supported by hunters' reports; *R. strykeri* spend most snow-free periods in mixed temperate forests and conifer forests at higher elevations but descend to lower altitudes during period of snowfall (Geissmann et al, 2011). The same phenomenon was also observed in *R. bieti* (Li et al., 2008) and *R. roxellana* (Li et al., 2010).

For the temperate snub-nosed monkeys (*R. bieti* and *R. roxellana*), daily range length is not strongly correlated with the seasonal variation in home range size. For example, *R. roxellana* at Yuhuangmaio had a larger home range in winter (7.1 km^2) and smaller home range in summer (5.0 km^2), yet their daily range lengths showed the reverse pattern; they were shorter in winter (1,600 ± 633 m) and longer in summer (2,600 ± 1,025 m). Both daily range length (1,900 ± 928) and home range area (2.9 km^2) were smallest in autumn (Tan et al., 2007). *R. bieti* at Samage had equally large home ranges in winter (18.2 km^2), spring (17.8 km^2), and summer (18.6 km^2) but a smaller home range in autumn (9.3 km^2). Their daily range lengths were longer in spring (1,721 km), summer (1,516 m) and autumn (1,877 m) and smallest in the winter (985 m) (Grueter et al., 2008, 2013).

Such seasonal changes of ranging pattern are strongly derived by seasonal changes in range use, food availability and food distribution. Several studies indicate that temperate snub-nosed monkeys have relatively larger daily range lengths in autumn and shorter daily range lengths in winter, while home ranges are smaller in autumn and larger in winter (Kirkpatrick et al., 1998; Li, 2002, Li et al., 2005; Tan et al., 2007; Grueter et al., 2008, 2013; Ren et al., 2009a). This is not what is expected unless food is distributed in large, superabundant feeding sites that are widely scattered through the entire home range in the winter. In this regard it is important to distinguish between daily range length – which is a measure of the distribution, availability, and renewal rate of resources on the time scale of days – and home range – which is a measure of the distribution, availability, and renewal rate of resources on the time scale of several weeks and months. The inconsistency between the daily range length and the home range size mentioned above is probably due to the fact that in late summer and autumn, the flushed and clumped fruits enable the temperate snub-nosed monkeys to maintain a smaller home range. However, the fruits patches scattered in the forest demand the monkeys increase travel in order to visit more patches and gain higher returns. For example, Grueter et al. (2008) found that daily range lengths of *R. bieti* at Samage exhibited a zigzag pattern in which the monkeys frequently returned to previously-visited areas during the month of September, a time when lichen accounted for 55.6%, and fruit 31.4%, of their diet. Hou (2018, Fig. 6 & Fig. 11) report that in *R. roxellana*, the study band visited more patches in autumn and winter (median level: 23/21 patches respectively) when fruits/seeds (> 25%), buds (32.7%), and bark (36.7%) accounted for the majority of feeding time compared to the summer (14 patches), and spring (17 patches) when leaves (>53%) accounted for the majority of feeding time.

Several hypotheses can be proposed to explain these results. It is possible that the shorter daily range lengths but larger home range in winter (Kirkpatrick et al., 1998; Xiang, 2010; Hou, 2018) may mean that lichen, buds, and tree back represent a spatially-concentrated and highly-available source of structural carbohydrates (lichens and buds, & tree bark) required to counter the added costs of remaining thermoneutral during cold periods. In other words, the spatially-concentrated high carbohydrate foods enables the temperate snub-nosed monkeys to take an energy-saving foraging strategy to confront the temperature constraints in winter, which may be reflected by their shorter winter daily range lengths (Kirkpatrick et al., 1998; Grueter et al., 2013). A second hypothesis is that larger daily range lengths in the summer and autumn (Kirkpatrick et al., 1998; Li, 2002; Tan et al., 2007; Li, Jiang, et al., 2010; Grueter et al., 2013; Hou, 2008; see also Liu et al., 2004) reflect a dietary pattern in which resource patches such as young leaves or fruits/seeds are spatially clumped and more widely distributed and therefore monkeys need to travel more to obtain these relatively high-quality foods. Given the large home range of snub-nosed monkeys compared to many other Asian colobines, preferred resources are likely to be hyper-dispersed across their home range over periods of weeks even months. However, the abundant and patchy resources – particularly in warm seasons – enable snub-nosed monkeys as well as Nepal gray langurs (*S. schistaceus*) to intensively exploit food resources in particular core regions of their total home range, and individuals remain in those regions for several days before traveling to other parts of their range to feed (Sugiyama, 1976; Bishop, 1979; Su et al., 1998 Kirkpatrick et al., 1998; Tan et al., 2007; Grueter et al., 2008; Sayers & Norconk, 2008; Ren et al., 2009b). This pattern of ranging and habitat utilisation may also reflect the fact that rates of food passage in colobines are extremely long (mean retention time of fibrous foods: 34-70 hours) (Dierenfeld et al., 1992; Edwards & Ullrey, 1999; Kirkpatrick et al., 2001) and individuals devote 21.2-59.1% of their daytime activity budget to resting (Korstjens et al., 2010). Thus, home range size and day range of snub-nosed monkeys appear to be driven by a combination of factors including band size, food availability and distribution, and gut physiology that requires long periods of digestion to break down difficult-to-digest foods.

This review of the feeding ecology of snub-nosed monkeys provides some information regarding how lichen sustains themselves over winter in high altitude mountainous environments; however, the role of lichenivory in the adaptive radiation of snub-nosed monkeys needs further examination. Also, the contradictory ranging pattern of the temperate snub-nosed monkeys with their band sizes means that the ecological constraints model must be questioned for these monkeys. In this chapter, I test the hypotheses that (H1) temperate snub-nosed monkeys (*R. bieti* and *R. roxellana*) generally do not follow the ecological constraints model which predicts that larger multi-level society (MLS) of each species have larger daily range lengths than smaller MLS of the same species and (H2) that bands with larger daily range lengths are also not characterised by a larger home range. Based on these two initial hypotheses, I test three additional predictions: (P1) snub-nosed monkeys inhabiting higher elevation habitats with lower temperatures are characterised by increased lichenivory throughout the entire year; (P2) increased band size in temperate snub-nosed monkeys (*R. bieti* and *R. roxellana*) is not strongly correlated with a larger daily range length or larger home range; (P3) other than sites' specifics, there may be a regular trend in seasonal daily range length and seasonal home range size of temperate snub-nosed monkeys. I predict that the patterns of seasonal daily range length and seasonal home range size vary inversely in temperate snub-nosed monkeys. Such patterns may be explained by their specialised diet, and seasonal patterns of food distribution and availability in subalpine temperate forests.

3.3 Methods

3.3.1 Literature review

For this analysis, I reviewed the available literature on feeding, ranging and activity budget of snub-nosed monkeys. The variables used in the analyses were selected based on three criteria: 1) during the study time, the focal band was not provisioned and thus I eliminated issues of feeding habits and ranging being affected by human interference; 2) data for the study were collected over a period of at least nine consecutive months; and 3) the published paper provided the results in a format that allowed me to conduct a regression analysis. I searched for English and Chinese publications by using Google Scholar with English or Chinese terms of snub-nosed monkey (金丝猴 or 仰鼻猴), feeding habit or diet (食性), time budget (活动时间分配), ranging behaviours (游走行为), home range (家域), and daily range length (日移动距离). Data were collected from 14 studies, including *R. avunculus* (n = 1), *R. brelichi* (n = 1), *R. roxellana* (n = 6) and *R. bieti* (n = 6). Besides information of band sizes, variables collected from these publications were categorised into four datasets: 1) environment: mean elevation, mean latitude, mean annual rainfall, and mean annual temperature; 2) diet: number of food plant species, feeding time allocation of leaves, fruit/seeds, flowers, buds, bark and lichens (Table Appendix I); 3) activity budget: percentage of time spent feeding, moving, resting, and socialising; 4) ranging data:

annual home range size and daily range length, home range size in each season and daily range length in each season (Table Appendix II).

For P1, I eliminated two *R. bieti* bands (Lasha and Fuhe). The Lasha band has been isolated in a small, degraded, and conifer-dominated patch by surrounding villages and high elevational pasturages (Huang et al., 2017). This has forced the monkeys to adapt to a diet with low diversity (36 plant species) that is also highly lichenivorous (>80% feeding time) compared to other southern-distributed *R. bieti* bands with obvious diet seasonality and diversity (e.g. Longmashan band feeds on 98 plant species on the adjacent mountain, Huo, 2005). The elimination of the Fuhe band is due to how the results are obtained and reported; the study uses micro-histological analysis of feces, not proportion of feeding time (Liu et al., 2004). For P2 and P3, I did not include data on *R. avunculus* and *R. brelichi* because they inhabit tropical and subtropical habitats and experience daily fission-fusion activities rather than maintaining a cohesive and stable band, as is reported for *R. bieti* and *R. avunculus* (Bleisch & Xie, 1998; Kirkpatrick et al., 1998; Dong, 2012; Qi et al., 2014). This was done to avoid the effects that changes in band cohesion and foraging party size during the day have on ranging patterns, patch size, and daily range length. Additionally, there is limited data regarding *R. avunculus* and *R. brelichi* 's ranging habits.

3.3.2 Statistical analyses

To test P1, I used a meta-analysis to explore the effects of mean elevation on the proportion of the annual time devoted to feeding on lichen, leaves, flowers, fruits/seeds, buds, and bark. I also calculated annual dietary diversity. I defined dietary diversity as the number of plant species consumed for each of the five snub-nosed monkey species (*R. bieti*, *R. roxellana*, *R. brelichi* and *R. avunculus*) based on 14 sites from 14 studies. I tested whether the proportion of feeding time devoted to lichen was correlated with the proportion of feeding time on leaves, fruits/seeds, flowers, buds, and bark by using linear regression. In addition, I assessed the influence of the environmental variables on plant food diversity and the proportion of feeding time spent on lichen, leaves, and fruits/seeds.

To test P2, I examined whether band size was correlated with feeding effort in temperate species (*R. bieti* and *R. roxellana*) in ten sites from ten studies (Table Appendix II). Feeding effort for each study band was calculated as (feeding time + moving time)/resting time (Dunn et al., 2010; Huang et al., 2017). Using linear regression, I tested whether there is a relationship (bivariate correlation) among feeding effort, daily range length, and the home range area; and whether temperate snub-nosed monkeys inhabiting larger bands are characterised by a larger daily range length and a larger home range across seasons and annually.

For P3 I focused only on temperate snub-nosed monkeys (*R. bieti* and *R. roxellana*) to compare the relative size of daily range lengths (eight studies in seven sites) between two seasons (spring-summer, summer-autumn, autumn-winter, winter-spring, spring-autumn, summer-winter) in order to define the seasonal pattern of daily range length and the sites' specificity of daily range length by using linear regressions and judging the position of discrete points relative to y=x axis. Daily range length and home range area in the previous season was used as the independent variable and the data from the latter season listed in the pair was considered the dependent variable. I did the same analysis for home range area by using the data from the same studies. Therefore, as long as the relationship of the regression line (RL) and the position of discrete points with respect to the y=x axis was known, I inferred the overall trend in the effect of season on daily range length and home range area.

3.4 Results

3.4.1 Snub-nosed monkeys living at higher elevations have higher rates of licheivory

The results indicate that mean elevation was significantly and positively correlated with time feeding on lichens (df = 11, $R^2 = 0.83$, $p < 0.0001$) and negatively correlated with time feeding on leaves (df=10, $R^2=0.47$, $p=0.014$), fruits/seeds (df = 10, $R^2 = 0.88$, $P < 0.0001$), and flowers (df = 6, $R^2 = 0.75$, $p = 0.006$), suggesting that lichens represent a dietary adaptation for snub-nosed monkeys living at higher elevations (Table Appendix III). A negative correlation was also found between mean elevation and bud consumption, although the correlation was weak (df = 8, $R^2 = 0.39$, $p = 0.052$). There was no significant correlation between elevation and the consumption of bark (df = 8, $R^2 = 0.09$, $p = 0.4$) (Fig 3.1). Similarly, at higher elevations, leaf (df = 10, $R^2 = 0.69$, $p = 0.001$), fruit/seed (df = 10, $R^2 = 0.68$, $p = 0.001$), flower (df = 10, $R^2 = 0.68$, $p = 0.001$) and bud (df = 10, $R^2 = 0.4$, $p = 0.005$) consumption were negatively correlated with lichen consumption. Additionally, as mean temperatures decreased, time spent feeding on leaves (df = 11, $R^2 = 0.38$, $p = 0.033$) and fruits/seeds (df = 10, $R^2 = 0.48$, $p = 0.012$) decreased while the proportion of time spent feeding on lichens increased (df = 11, $R^2 = 0.53$, $p = 0.015$) (Fig 3.1). The proportion of the diet composed of lichens was weakly and negatively correlated with mean latitude (df = 10, $R^2 = 0.34$, $p = 0.047$).

In general, species diversity in the diet was not correlated with either mean elevation or annual temperature. However, two of 13 sites had a disproportionate effect on results masking the relationships for the other species and bands with more representative results. Plant species dietary diversity recorded for *R. avunculus* in the tropical forest of Khau Ca and that of *R. roxellana* at Guangyingshan on the southern slopes of the Qingling Mountians is considerably lower (50 species and 53 species, respectively) than that reported for *R. roxellana* at Zhouzhi West on northern slopes of Qingling Mountians (123 species). The annual temperature and annual precipitation at Khau Ca and at Guangyingshan are higher than that at Yunhuangmiao (Table Appendix I) and the former two sites therefore should have higher biodiversity than the later one and thus should lead to higher dietary diversity for the monkey bands at Khau Ca and at Guangyingshan than that at Yunhuangmiao. After excluding these two data points, I found a significantly positive relationship between annual temperature (df = 9, $R^2 = 0.61$, $p = 0.004$) and plant species dietary diversity, and a significantly negative correlation with mean elevation and plant species dietary diversity (df = 9, $R^2 = 0.58$, $p = 0.007$). Overall, these results indicate that temperate snub-nosed monkeys (*R. bieti* and *R. roxellana*) consume more lichen in high elevational areas which have low temperatures and are characterised by reduced plant species diversity.

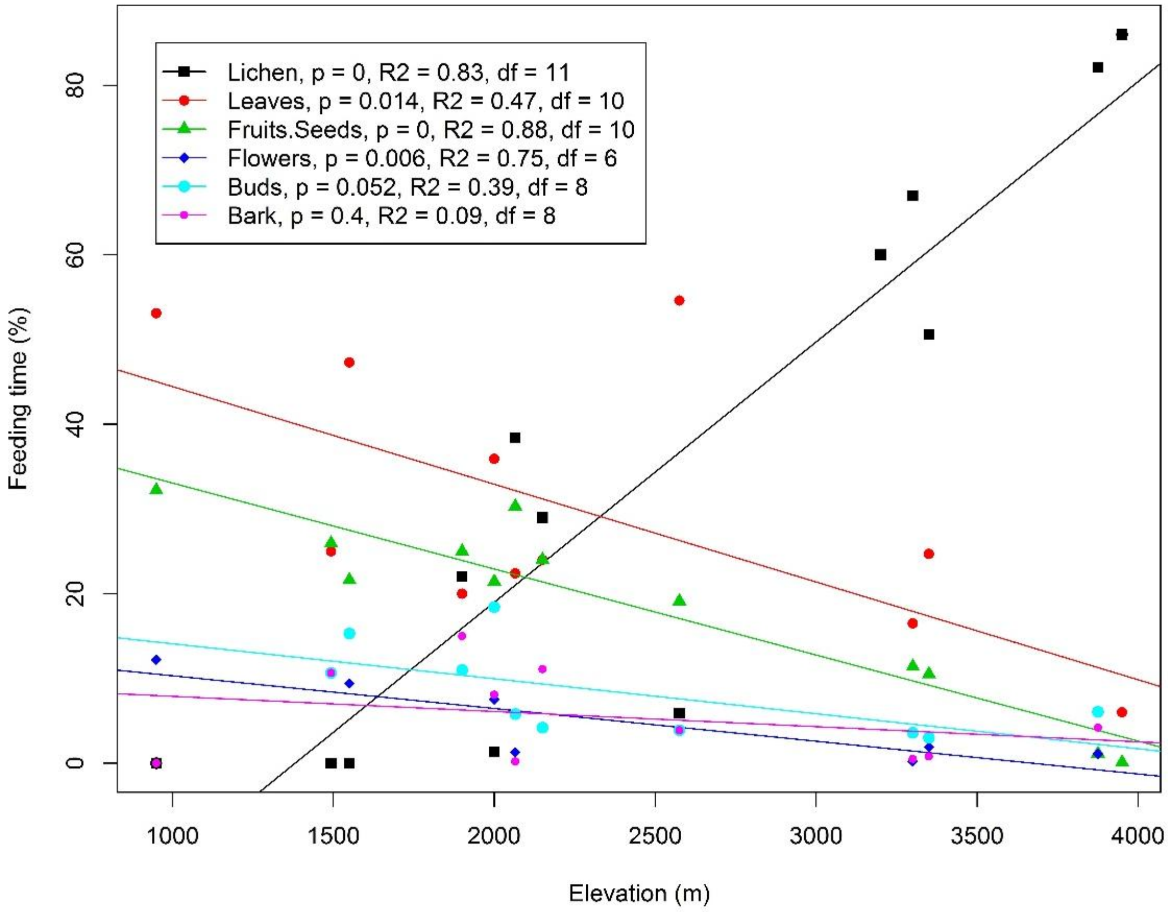


Figure 3.1 Correlation between elevation and the proportion of feeding time snub-nosed monkeys spent consuming different food items.

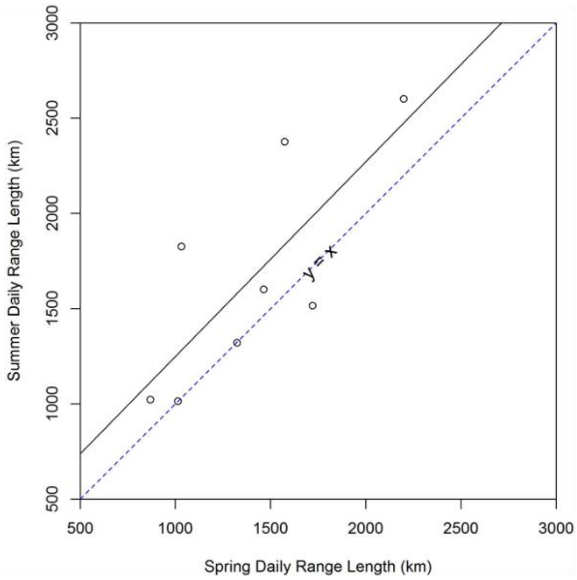
3.4.2 Ecological constraints model unsupported in temperate snub-nosed monkeys

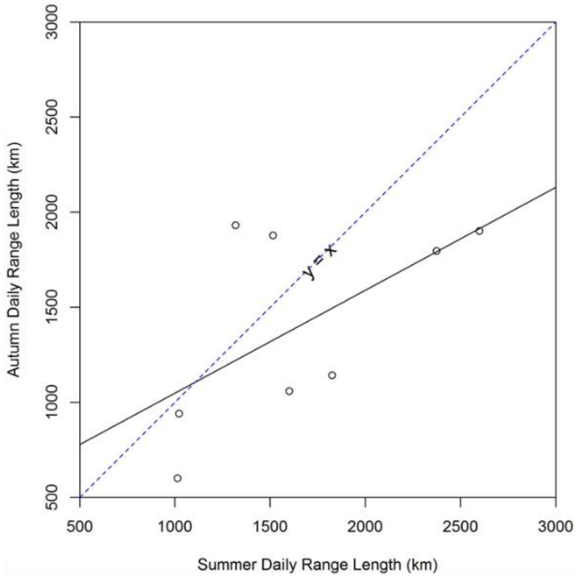
There is no significant correlation of band size with feeding effort, annual mean daily range length or annual home range size in temperate snub-nosed monkeys (Table Appendix III). In addition, band size did not correlate with daily range length in any seasons nor with home range size in spring and autumn. I did find that band size was positively correlated with home range size during both the summer (df = 10, R^2 = 0.63, *p* = 0.028) and the winter (df = 10, R^2 = 0.67, *p* = 0.017). Annual mean temperature, mean elevation, daily range length and home range area were not significantly related to feeding effort.

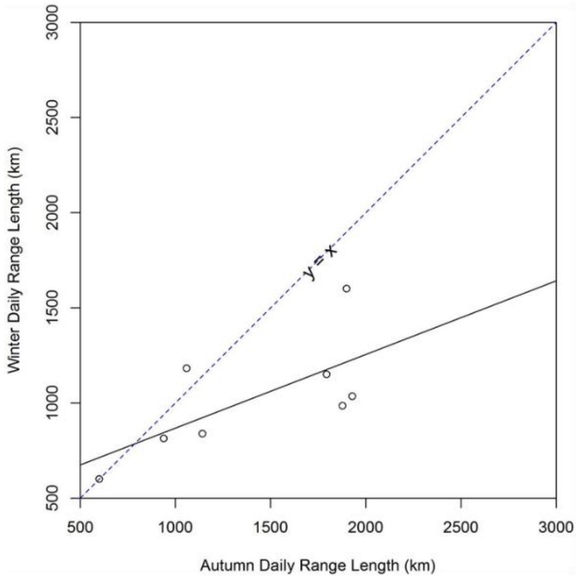
Overall, my results support P2 indicating that larger bands of temperate snub-nosed monkeys are not consistently characterised by increased feeding effort, a larger daily range length, or a larger home range size than smaller bands.

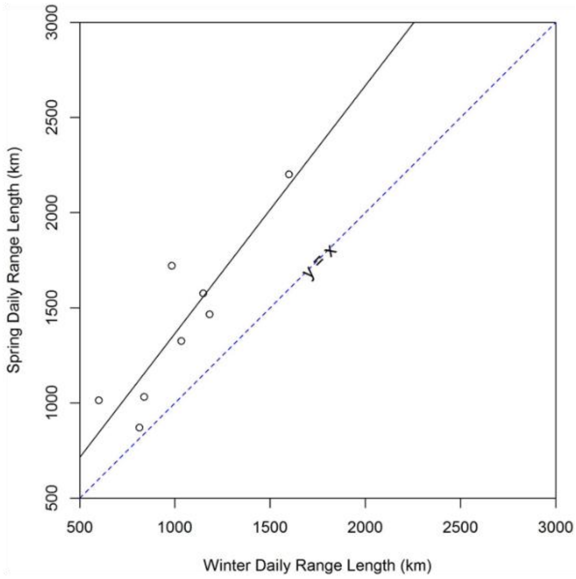
3.4.3 Seasonal changes in snub-nosed monkey ranging patterns

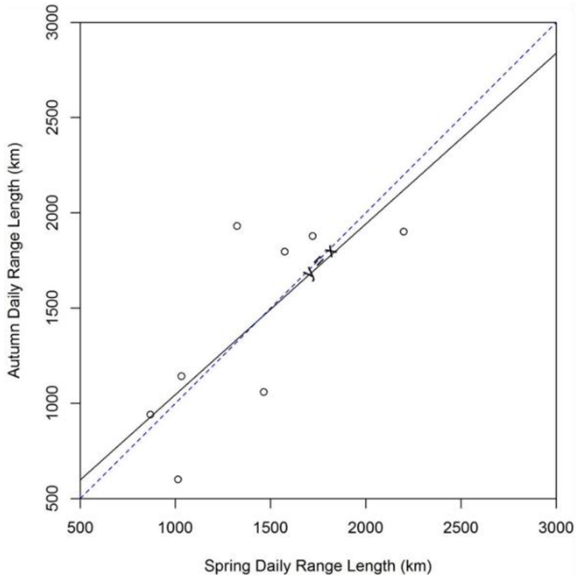
These results show that across seasons, temperate snub-nosed monkeys (*R. bieti* and *R. roxellana*) alter their ranging behaviour. Specifically, I found that temperate snub-nosed monkeys had significantly shorter daily range lengths in spring than summer [df = 6, t = 2.94, $R^2 = 0.77$, $p = 0.026$, regression line (RL) above y = x], shorter daily range lengths in winter than that spring (df = 6, t = 4.81, $R^2 = 0.89$, $p = 0.003$, RL above y = x) and significantly longer daily range lengths in summer than winter (df = 7, t = 4.10, $R^2 = 0.84$, $p = 0.045$, RL below y = x) (Fig. 3.2). In 75% (6 out of 8) of the study bands, daily range lengths in summer were larger than in autumn (df = 6, t = 1.83, $R^2 = 0.60$, $p = 0.1169$, RL almost below y = x), however this was not significant. Similarly, only two of eight (25%) bands had winter daily range lengths larger than autumn daily range lengths (df = 6, t = 2.26, $R^2 = 0.68$, $p = 0.064$, RL almost below y = x). In 5 out of 8 (62.5%) study bands, daily range lengths in spring were larger than autumn lengths (df = 6, t = 2.73, $R^2 = 0.75$, $p = 0.034$, RL almost overlaps y = x and hollow points are distributed on both sides of y = x). Overall, these results suggest that in most sites, temperate snub-nosed monkeys have the shortest daily range lengths in winter and the largest daily range lengths in summer (Figure 3.3).


Summer Daily Range Length (km)
500
1000
1500
2000
2500
3000
y = x
Spring Daily Range Length (km)


Autumn Daily Range Length (km)
500
1000
1500
2000
2500
3000
y = x
Summer Daily Range Length (km)


Winter Daily Range Length (km)
500
1000
1500
2000
2500
3000
y = x
Autumn Daily Range Length (km)


Spring Daily Range Length (km)
500
1000
1500
2000
2500
3000
y = x
Winter Daily Range Length (km)


Autumn Daily Range Length (km)
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1000
1500
2000
2500
3000
y = x
Spring Daily Range Length (km)

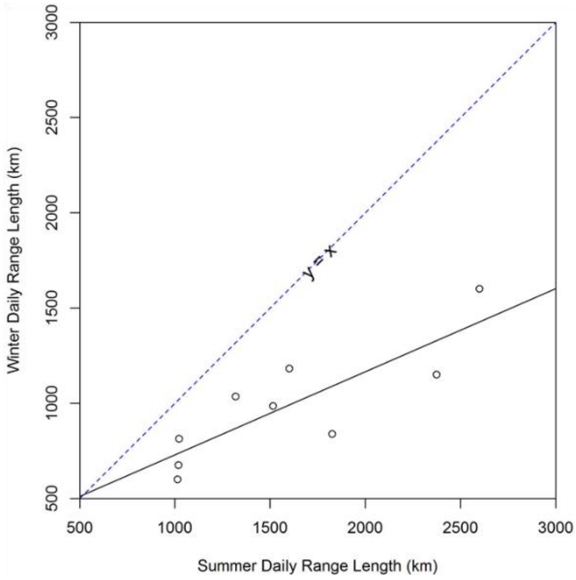

Winter Daily Range Length (km)
500
1000
1500
2000
2500
3000
y = x
Summer Daily Range Length (km)

Figure 3.2 Comparison of the relative size of daily range lengths of temperate snub-nosed monkeys (*R. bieti* and *R. roxellana*) between two seasons (spring-summer, summer-autumn, autumn-winter, winter-spring, spring-autumn, summer-winter). The regression line and hollow points' positions relative to y = x axis reflect general trends of daily range lengths and specificity of daily range length of a band between two seasons in a site, respectively.

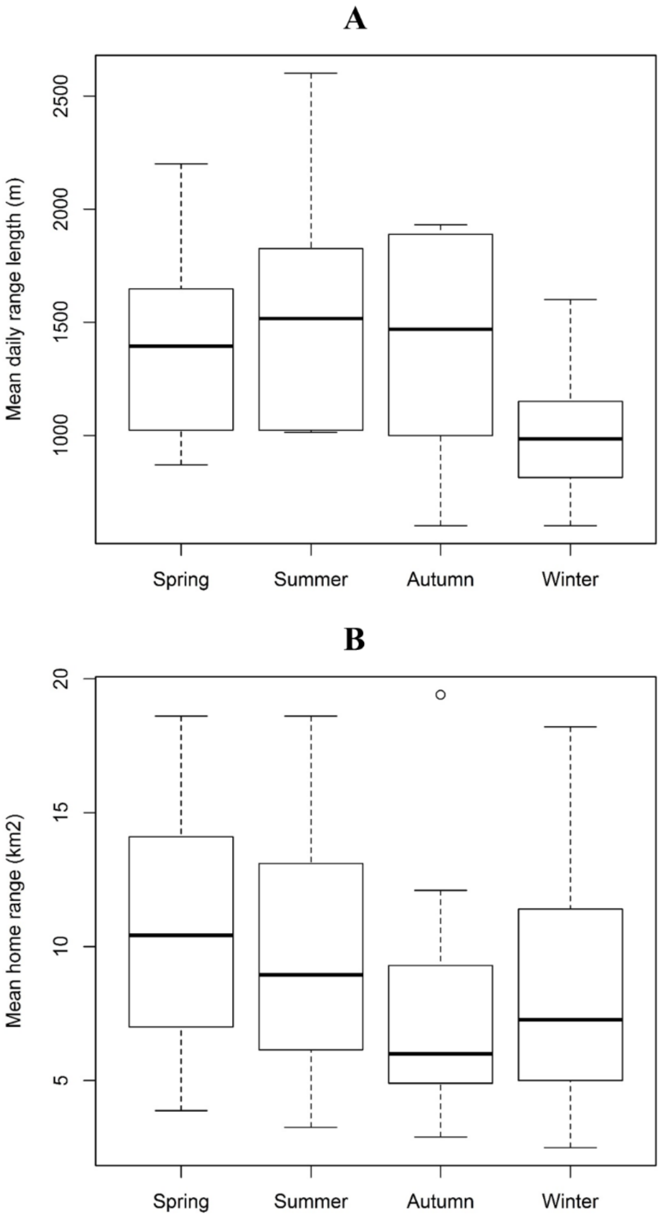
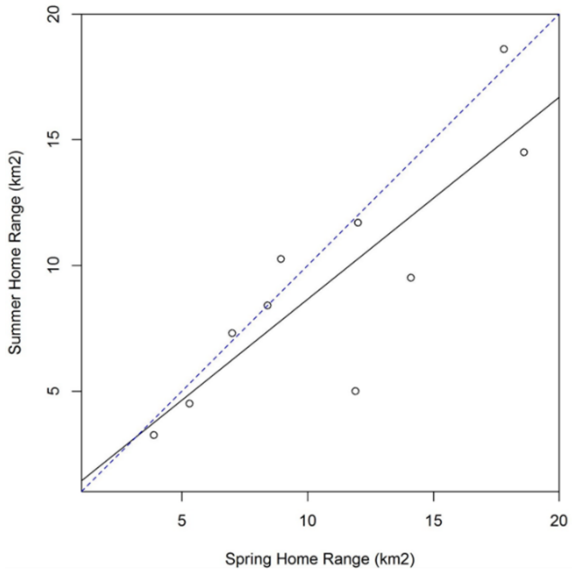
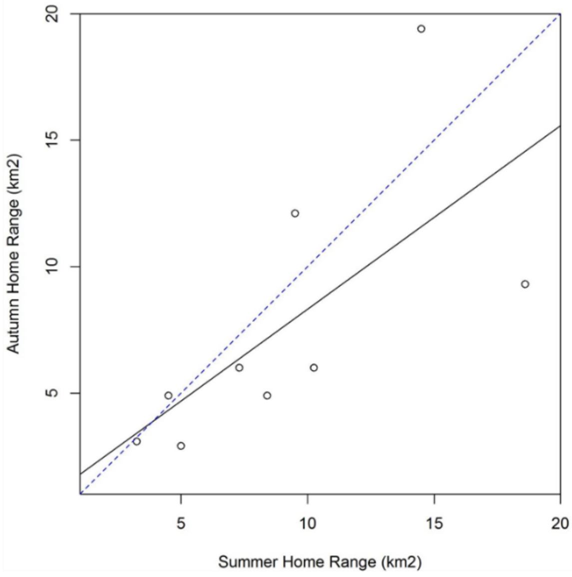
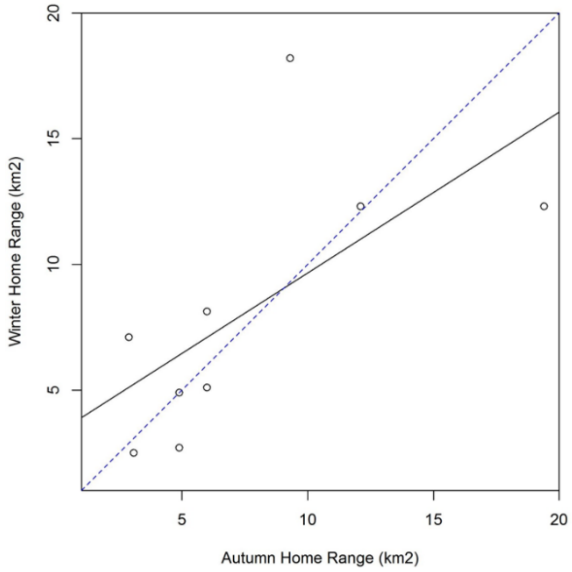


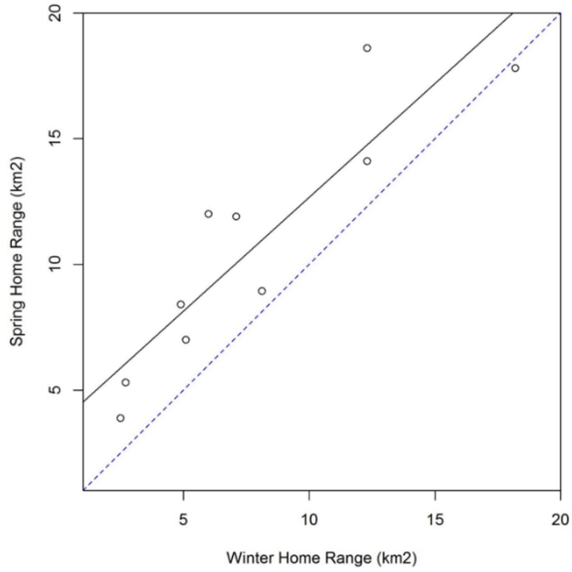
Figure 3.3 Box plot of seasonal variation in daily range length (A) and home range size (B) for temperate snub-nosed monkeys (*R. bieti* and *R. roxellana*). The maximum and minimum values in the box of plot A indicate that the daily range lengths in summer and autumn were especially variable in temperate snub-nosed monkeys across sites. However, the median lines also show they increase their path lengths from a low in winter, a high in summer, and a decline in autumn. Relatively, the box of plot B shows that the winter home range size is more variable in temperate snub-nosed monkeys across different sites. However, maximum, minimum, and median values in the plots show that temperate snub-nosed monkeys reduced the size of their home range from spring to autumn, then increased home range size in winter.

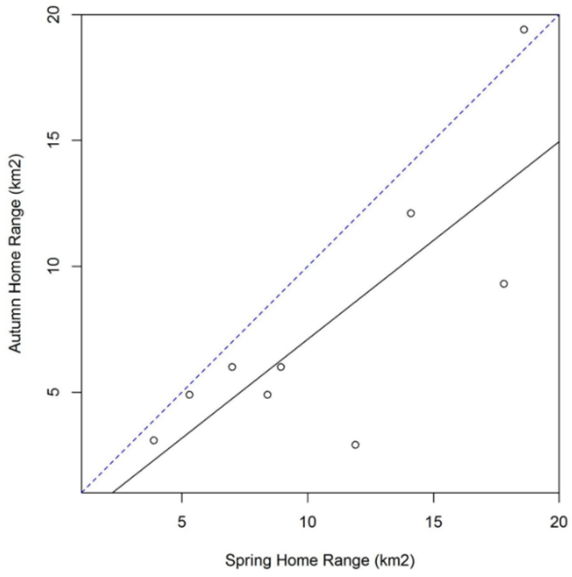
Similar to daily range length, home range size was found to vary seasonally in temperate snub-nosed monkey species. The results indicate these primates have a larger home range in spring compared to summer (df = 8, t = 4.47, $R^2 = 0.84$, $p = 0.002$, RL below y = x) and autumn (df = 7, t = 3.32, $R^2 = 0.78$, $p = 0.013$, RL below y = x), and summer compared to autumn (df = 7, t = 2.45, $R^2 = 0.68$, $p = 0.044$, RL almost below y = x) and winter (df = 10, t = 4.87, $R^2 = 0.84$, $p = 0.007$, CL below y = x) (Fig 3.4). They have a smaller home range in winter than in spring (df = 8, t = 5.82, $R^2 = 0.9$, $p = 0.0004$, RL above y = x). There was no consistent relationship between home range size in autumn and winter (df = 7, t = 2.26, $R^2 = 0.65$, $p = 0.058$, RL crosses y = x). Overall, results show that temperate snub-nosed monkeys have the largest home range in spring and the smallest home range in autumn (Fig 3.3).


Summer Home Range (km2)
Spring Home Range (km2)
5
10
15
20
5
10
15
20


Autumn Home Range (km2)
Summer Home Range (km2)
5
10
15
20
5
10
15
20


Winter Home Range (km2)
Autumn Home Range (km2)
5
10
15
20
5
10
15
20


Spring Home Range (km2)
Winter Home Range (km2)
5
10
15
20
5
10
15
20


Autumn Home Range (km2)
Spring Home Range (km2)
5
10
15
20
5
10
15
20

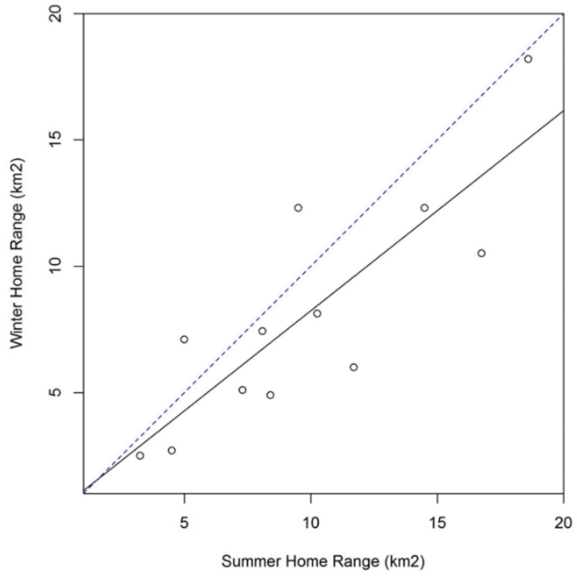

Winter Home Range (km2)
Summer Home Range (km2)
5
10
15
20
5
10
15
20

Figure 3.4 Comparison of the relative size of home ranges of temperate snub-nosed monkeys (*R. bieti* and *R. roxellana*) between two seasons (spring-summer, summer-autumn, autumn-winter, winter-spring, spring-autumn, summer-winter). The regression line and hollow points' positions relative to y=x axis reflect general trends of home ranges and specificity of home range of a band between two seasons in a site, respectively.

3.5 Discussion

3.5.1 Summary of results

Using a meta-analysis of published studies, this chanpter aimed to answer three questions: 'Do snub-nosed monkeys exhibit behavioural adaptations related to the exploitation of lichen as a dietary staple?', 'Does the ecological constraints model offer the strongest explanation to account for relationships between MLS size, daily range length, and home range size in the two species of temperate snub-nosed monkeys?', and 'Do both species of temperate snub-nosed monkeys exhibit a similar pattern of diet, ranging, and habitat utilisation or does each species exploit their habitat in different ways?'. The results suggest that snub-nosed monkeys exploiting high altitude temperate forests have an increased dietary dependency on lichen; a dependency that increases at higher elevations and in habitats characterised by colder winter temperatures. Compared with altitude, latitude is positively but weakly correlated with lichen feeding time suggestig latitude is not a critical determinant. This is evidenced by the fact that *R. roxellana* at Guanyinshan has a higher latitudinal but lower altitudinal habitat (N 33.67 °, 1900 m) and spend less time feeding (22%) on lichens than *R. bieti* at Samge (N 27.57 °, 3300 m, 67% feeding time) (Grueter et al., 2009b; Zhao et al., 2015).

I found that lichen in the diet was negatively correlated with consumption of leaves, fruits/seeds, flowers, and buds across elevation gradients, suggesting lichens may play an important role in nutrient balancing (see evidence from Svihus & Holand, 2000; Kirkpatrick et al., 2001). Moreover, the results suggest that band size was not correlated with feeding effort and that individuals in larger bands did not travel longer distances per day (e.g. daily range length) or occupy a larger home range compared to individuals in smaller bands. Larger bands were found to exploit larger home ranges in summer and winter than in spring and autumn. This suggests that preferred resources are less concentrated in summer and winter than during other times of the year in the temperate high elevational habitat of snub-nosed monkeys. However, given that temperate snub-nosed monkeys were found to have longer daily range lengths in summer and autumn and shorter daily range lengths in winter and spring, this suggests that throughout most of the year, the foraging pattern of *R. bieti* and *R. roxellana* involves the exploitation of core areas of concentrated feeding sites and once these sites are functionally depleted (insufficient to support a band), the monkeys travel to another part of their home range (e.g. core area) to feed.

3.5.2 Function of lichen feeding

A dietary dependence on lichens may both provide nutrient balancing and allow temperate snub-nosed monkey species to survive in large, super bands. Lichens are a patchy yet ubiquitously available resource that are not produced seasonally and have a nutritional profile characterised by high carbohydrate content (Kirkpatrick et al., 2001; Steenbeek & van Schaik, 2001; Marshall & Wrangham, 2007; Grueter et al., 2009b; Grueter et al., 2012; Liu et al., 2013a). Lichens consumed by temperate snub-nosed monkeys are present on common tree species such as *Abies georgei* and *Picea likiangensis*. Given that lichen is available in large and continuous food patches, temperate snub-nosed monkeys may be able to use them to live in larger bands and reduce travel distance (Janson & Goldsmith, 1995), while avoiding the ecological challenges associated with limited food availability (Grueter & van Schaik, 2009; Grueter et al., 2009a). This is supported by the fact that temperate snub-nosed monkeys live in more cohesive and more stable bands than species of snub-nosed monkeys whose diet is dominated by fruits, seeds, and leaves (e.g. *R. avunculus*, Dong 2012; *R. brelichi*, Bleisch & Xie, 1998; Yang et al., 2002). During periods of food scarcity or when lichen is unavailable, Ren et al. (2012) report that *R. bieti* bands fission into two sub-bands for periods of 2-11 days and then reunite in areas of their range with large food patches. A similar fission event was observed by Grueter et al. (2008) when valued fruit resources became rare and extremely patchy in late winter.

3.5.3 Significance of daily range length and home range dimension

Like other primates, snub-nosed monkeys seasonally change daily range length and home range area in response to temporal variation in both the abundance and spatiotemporal distribution of foods (Bennett, 1986; Boonratana, 2000; Li et al., 2000; Raño et al., 2016; Li et al., 2020). The results of the current study show that temporal changes in daily range length and home range area utilised differ between *R. bieti* and *R. roxellana*. As suggested by Grueter et al. (2013), larger bands of *R. bieti* do not travel greater distances per day than do smaller bands of *R. bieti*. By regressing daily range length against band size in *R. bieti*, the existence of scramble competition within bands could be inferred, but the magnitude of the increase in foraging effort with band size is modest (Grueter & van Schaik, 2009).

Under the subalpine environment and subjected to Asian monsoons, the rainy season begins in spring (April) and ends in autumn (October) and thus the long winter is from November to March in temperate snub-nosed monkey habitats. During spring, summer, and autumn, high-quality foods such as spring buds and young leaves (with high protein), flowers (with high protein and nonstructural carbohydrates), and fruits and seeds (with nonstructural carbohydrates) (Zhang et al., 2013; Liu et al., 2013; Hou et al., 2018) are available and distributed in small scattered patches (see seasonal food availability data: Fig 2 in Li, 2006; Fig. 4 in Xiang et al., 2007, Fig 4, 6 in Grueter et al., 2009b; Fig 2 in Hou et al., 2018). In temperate snub-nosed monkeys' habitat, mature leaves are the most available resources in late spring to early autumn and lichen and bark are available all year (Grueter et al., 2009b; Liu et al., 2013b; Zhao et al., 2015; Hou et al., 2018). Therefore, I suggest that during spring and summer – by using a high-cost, high-returns strategy – temperate species of snub-nosed monkeys increase daily range length in order to exploit more seasonally-available and nutrient-rich food items (e.g. spring buds, flowers, young leaves, unripe fruits) that are distributed in small, scattered food patches for maximising nutrient intake. In contrast, snub-nosed monkeys decrease their home ranges in spring and summer by continuing to rely on both leaves and lichens as an important dietary staple (Li, 2002; Tan et al., 2007; Xiang et al., 2010; Grueter et al., 2013). Similarly, the monkeys maintain large daily range lengths for depleting fruits and seeds in clumped but widely-scattered patches as the high-cost, high-returns strategy during autumn (see Sayers & Norconk, 2008, Xiang et al., 2013) while exploiting a smaller home range – as they can obtain adequate leaves and lichens in smaller areas (Tan et al., 2007; Grueter et al., 2008). Conversely, the increase in their home ranges from autumn to winter likely reflects the exploitation of the remaining leaves, fruits and seeds (Grueter et al., 2008) or lichens (Kirkpatrick et al. 1998; Tan et al., 2007) in scattered patches; the relatively abundant lichens or bark enable them to maintain shorter daily range lengths for energy-saving purposes (see Ding & Zhao, 2004; Guo et al., 2007; Hou, 2018). Studies by Su et al. (1998) and Kirkpatrick et al. (1998) indicate that during all seasons, temperate snub-nosed monkeys revisit the same core area and food patches over the course of about seven days and then travel a distance of 1200-2950 m to encounter another core area with sufficient food patches for the band to remain in that general location for another set of successive days (See Fig. 3 in Su et al., 1998, pp. 262; Kirkpatrick et al., 1998; Ren et al., 2009b). This general pattern of habitat utilisation – which is reported to occur thoughout most of the year – enables large bands of snub-nosed monkeys to intensively exploit (but not overexploit) a series of core areas during a given season, and then move to other parts of their range as these new resources become available.

3.5.4 Lichenivory as a preferred dietary resource for temperate snub-nosed monkeys

Compared with leaves, lichens are lower in crude protein (3.4-6.1% in lichens vs. 11-20.4% in leaves) (Kirkpatrick, 1996, Liu et al., 2013b; Hou et al., 2018), although several researchers have considered lichen to represent a nutritionally-limited fallback food consumed principally when other foods are less available (Xiang et al., 2007; Bisell, 2014). Based on my review, I argue this is a misrepresentation for temperate snub-nosed monkeys. Compared with other fallback foods such as bark and evergreen mature leaves, lichen is consumed more when more-preferred foods (i.e. spring buds, young leaves and fruits) are not available. Rather than being defined as a stable fallback food (Grueter et al., 2009), lichen is thus best considered as a sub-preferred food resource, which can efficiently provide a considerable amount of metabolisable energy [lichen rich in digestible nonstructural and structural carbohydrates (72.45-98.2%), Kirkpatrick et al, 2001; Akbulut & Yildiz, 2010] to snub-nosed monkeys and thus can support large bands of hundreds of individuals.

The digestibility of a food can be improved by higher degrees of fermentation that results from longer retention times (Lambert 1998; Milton, 1998; Remis, 2002; Lambert & Fellner, 2012). Snub-nosed monkey are specialised foregut fermenters characterised by long gut retention time (47 hours) and a highly diverse microbiome required to digest structural carbohydrates present in tree bark, twigs, and leaves (Chalker-Scott, 1999; Kirkpatrick et al., 2001; Lambert, 2002). Given similarities in the digestive adaptations required to ferment leaves and lichens (Kirkpatrick et al., 2001), it is likely that a pre-existing forestomach adaptation that evolved to ferment leaves (sacculated stomach) enabled an ancestral species of snub-nosed monkeys to expand their diet and exploit large patches of lichen present in cold alpine forests. In captivity, when lichen content was increased from 16% to 73% in *R. bieti*'s diet, the retention time of mixed foods increased from 33 to 66 hr, and the digestibility increased from 71% to 80% (Kirkpatrick et al., 2001). This also can be found in another foregut fermenter: reindeer (*Rangifer tarandus*) require 23 hours retention time to process foods grazed from winter (and lichen-heavy) vegetation (43% leafy items, 56% lichens), which is longer than food grazed from summer tundra vegetation (51% leafy items, 24% lichens) (Bergerud, 1972; Person, et al., 1975). In addition, the retention time for reindeer differs seasonally, from 10 hours in summer, when lichen digestibility is 38%, to 17-20 hours in winter, when lichen digestibility is 78% and higher than that of shrubs (White et al., 1980). This increased retention time allows reindeer to extract more energy from lichen in the winter than in the summer. Moreover, lichen (3% protein) consumption was found to significantly decrease daily water intake (2.4 vs. 3.9 litre) and its thermal energy cost per daily metabolisable energy (6.5% vs. 13.9%) for reindeer in winter compared to consumption of medium-protein feeds (10% protein) (Soppela et al., 1992). As snub-nosed monkeys show some digestive similarity as the reindeer, the monkeys may use the similar digestive strategy to realise efficient use of lichens and a reduction of daily energy expenditure.

It is also important to acknowledge that lichens may represent a more nutrient-rich food item than previously assumed given that there is a symbiosis between the fungi and cyanobacterium present in lichens, resulting in their cell walls being composed of a diverse array of non-fibre, but fibre-like, structures such as lichenin, a polysaccharide (Spribille et al., 2016). This lichen polysaccharide is a six-carbon saccharide mannose and includes β-glucans, α-glucans, and galactomannans (Olafsdottir & Ingólfsdottir, 2001), which cannot be digested by mammalian enzymes but can be effectively fermented by intestinal microbacteria (Svihus & Holand, 2000). Compared with a cellulose molecule that consisted of thousands of $C_6H_{12}O_6$ molecules, a lichenin molecule only has hundreds of $C_6H_{10}O_5$ molecules and less oxygen. This means that the digestion of lichens requires lower energy demands for hydrolysis by their intestinal microbiome and that they can release more energy for the same unit mass of consumption.

For captive *R. bieti*, Kirkpatrick et al. (2001) found the digestibility of hemicellulose (which they also hinted might actually be lichenin) to be 91%. Comparatively, hemicelluloses (containing many different sugar monomers, such as the five-carbon saccharides xylose and the six-carbon saccharides mannose) and cellulose (consisting of several hundred or more glucose units) also required fermentation by the snub-nosed monkey intestinal flora (Smith, 1999; Xu et al., 2015; Liu et al., 2018). However, the digestibility of neutral detergent fibre (including hemicellulose, cellulose and indigestible lignin) for *R. bieti* (64%) and *R. roxellana* (74.3%) in captivity (Bissell, 2014; Huang, 2014) and hemicellulose digestibility (62-73%) and cellulose digestibility (57-77%) for François langurs (*Trachypithecus francoisi*) in captivity (Nijboer et al., 2001) indicate that compared to diets high in fibrous foods (such as mature leaves, barks and twigs), the consumption of lichens is a more efficient way to gain energy for snub-nosed monkeys. Therefore, compared with other fallback foods such as bark or twigs, lichen (with relatively higher carbohydrate contents, higher digestion efficiency and lower energy cost for digestion) is preferred and foraged year-round by the temperate snub-nosed monkeys. At the site of Yuhuangmiao in the Qinglin Mountains of north-central China, the East Ridge band of *R. roxellana* living at higher elevation (2,150 m) increase lichen consumption in winter (from 62.3% of feeding time to 1.6% of feeding time in summer). In contrast, the West Ridge band inhabits lower elevational forests (2,000 m) and reduces lichen consumption (3.6% of winter feeding time) and increased tree bark consumption (36.7-49.5% of winter feeding time) in winter due to a rarity of lichens in the home range of the West Ridge band (Guo et al., 2007; Huang, 2015; Hou et al., 2018; Guo, S. T. & Huang, Z. P., pers. comm., 2018).

According to Clauss et al. (2008)'s suggestion regarding primate feeding strategies, lichenivorous snub-nosed monkeys may gradually switch digestive strategy from an “intake approach” (relative high mass intake, short mean retention time, and relatively low digestibility of leaves) to an “efficiency approach” (relative low mass intake, long mean retention time and high digestibility of lichens) in response to seasonal changes of food availability [note: lichens have lower tissue density (21 grams per liter) than food leaves (35 grams per liter), Kirkpatrick, 1996]. With a reduction in daily travel (Ding & Zhao, 2004; Li, 2009; Li et al., 2013; Hou, 2018) and lichenivory, the monkeys achieve the most energy gain-cost efficiency during the winter, when thermoregularly costs are extremely high and some females are pregnant (Xiang & Sayers, 2009; Huang et al., 2012; Guo et al., 2018). Data from other lichenivorous primates with very large band sizes also support this argument. In Nyungwe Forest in Rwanda, Angolan colobus monkeys' (*Colobus angolensis*) annual diets include 20.6-32% lichen – and it can be up to 57% of their monthly diet (Vedder & Fashing, 2002; Miller et al., 2020). This study band contains over 300 individuals but remains cohesive throughout the entire year (Fashing et al., 2007). Similarly, a large band of Barbary macaques (*Macaca sylvanus*) in Akfadou (Algeria) including 88 individuals (usual band size for this species: 25-55 individuals) in cold and humid oak forests, and year-round lichen consumption accounted for 43% in their feeding time in winter (Ménard, 2002). Overall, these examples indicate that high digestibility and high carbohydrate content combined with an energy conservation for digestion makes lichens extremely high in energy gain-cost efficiency compared with other natural winter foods for temperate snub-nosed monkeys. This may help temperate snub-nosed monkeys to maintain a super large and cohesive band in alpine mountains during the winter, when energy costs may double due to increased thermoregularly costs (Guo et al., 2018).

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Chapter 4: The Natural History and Conservation Status of the Black Snub-Nosed Monkeys

4 An Introduction to the Black Snub-Nosed Monkeys

The black, or Myanmar, snub-nosed monkey (*Rhinopithecus strykeri*) was first discovered in 2010 by a field research team from Flora & Fauna International (FFI, Myanmar) and the Biodiversity and Nature Conservation Association (BANCA) in an area covering approximately 560 km^2 of mountain forests located in the watershed of the Maw and Lakin rivers, the tributaries to the N'mai Hka River in the northern Kachin State of Myanmar (Geissmann et al. 2011). *R. strykeri* was then identified and described as a new species based on its morphological characteristics (fur, beard and tail) and geographic distribution in 2011. (Geissmann et al. 2011) A later year, this species was also found in Gaoligong Mountains National Nature Reserve (GLGMNNR) in China, an area approximately 50 km southeast of the monkey populations in Myanmar (Long et al., 2012). Before an official scientific name was given to this species, the local Lisu people in China knew it as "*Meyah*" (Chinese: 弥阿) (Long et al., 2012) and the Lawwaw people in Myanmar called it "*Myuk na tok te*", both meaning "black monkey with an upturned nose" (Pappas, 2010; Ma et al., 2014). In China in general, the common name of this species is Nujiang snub-nosed monkey or "*Nujiang Jin-si-hou*".

4.1 Morphological Characteristics and Life History

Based on Geissmann et al.'s (2011) study, adult males (estimated > 6 years of age) have large bodies (head-body length: 55.5 cm, body weight: 17 kg) mainly covered by long black fur. Their crown consists of a long, thin and forward-curved crest of black hairs (Fig. 4.1 a), while the blackish-brown back fur appears around the cheek, the front chest, and the inner sides of the upper arms and upper legs. Only the ear tufts, chin beard and the perineal or testicle area are distinctly white (Fig. 4.1 b). The tail is approximately 1.4 times (78 cm) the body length. While originally it was thought that only the facial bare skin is pale pink, I found that the soles of the hands and feet of two captive *R. strykeri* are also pink, which contradicts Geissmann et al. (2011)'s description that skin on the extremities is black (Fig. 4.1 c). Adult female body size (based on two captive individuals, head-body length is 52-54 cm, tail length 63-66 cm, and weight 9-11.6 kg) is smaller than that of the male and is covered with the same colour hairs (Fig. 4.1 d). The crown of the adult female is not long nor as distinct as the adult male. Therefore, slight sexual dimorphism in body size can be seen in *R. strykeri* (see also cranial and bental sizes of a male and a femal specimen in Table Appendix IV). Newborn to one-year-old infants are fully covered in whitish gray fur and have pale blue faces (Fig. 4.1 e). Between one to two years of age, their fur gradually transitions to the adult colouration. Aung et al. (2011) recorded head-body, tail and hind foot lengths of a less than one years old dead infant to be 19 cm, 22 cm and 7 cm, respectively (Fig. 4.1 e).

The lifespan and fecundity of *R. strykeri* are still unknown. Due to phylogenetic conservatism, this species is likely similar to the closely-related black-and-white snub-nosed monkey (*R. bieti*) (Liedigk et al. 2012; Zhou et al. 2014; Yu et al., 2016). Geissmann et al. (2011) estimated that the sexual maturity of *R. strykeri* is six years old but captured *R. bieti* females can have their first birth at five years old (Ji et al., 1998). This is supported by my own observation of a captive female *R. strykeri* at the Yaojiaping Wildlife Rescue Centre in the GLGMNNR, which reached sexual maturity at approximately 5.5 years old (age of first menarche). I judge that the pregnant females may give birth started form February or March as the newborn infants appeared in their mother's arms were see in early of April, which is similar to golden snub-nosed monkey (*R. roxellana*) and *R. bieti* (Zhang et al., 2000; Xiang & Sayers, 2009; Huang et al., 2012). According to the age-at-maturity and fur colouration changes, 0-2 year-olds can be defined as infants, 2-4 year-olds as juveniles, 4-6 year-olds as subadults and > 6 year-olds as adults. Further research is needed to confirm these age categories.

Figure 4.1 Morphological features of *Rhinopithecus strykeri*. a. an adult male. Photo provided by Lushui Sub-Administration Bureau of GLGMNNR; b. the fur coloration of perineal area, photo by Dong Shaohua; c. the skin colouration of the palms of the hands; d. an adult female grooming a juvenile female; e. A dead infant, photo by Saw Soe Aung & Thet Naing Aung (FFI).

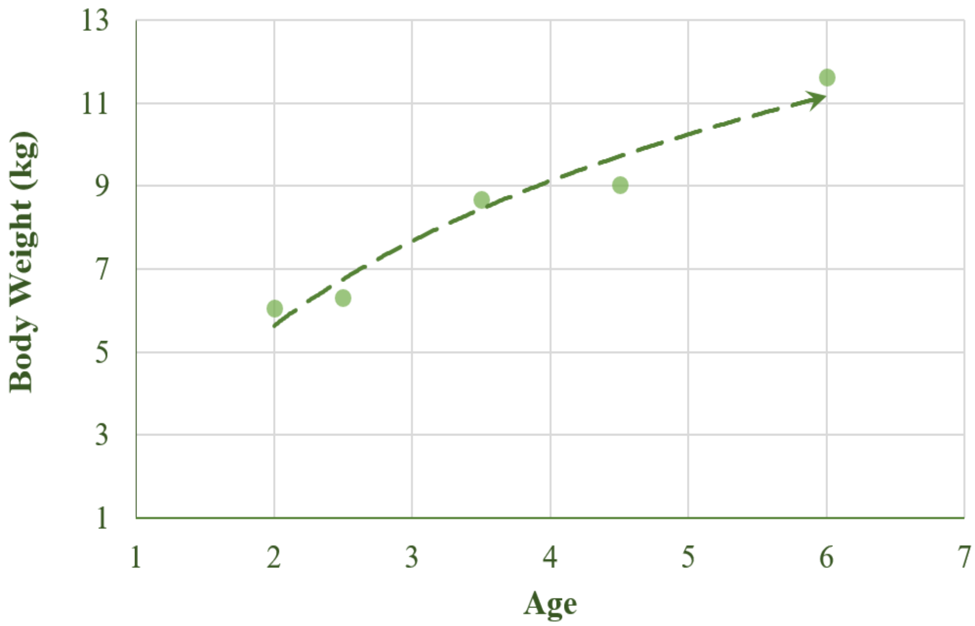

Figure 4.2 Weight growth curve across ages for a captive female.

4.2 Geographic Distribution and Population Status

Intensive field studies between 2010-2017 by FFI and BANCA in Myanmar (see Meyer et al., 2017) along with the those by the Institute of Eastern Himalaya Biodiversity Research (IEHBR), Dali University (field surveys, interviews, camera trap studies, see Li et al., 2014; Ma et al., 2014; Chen et al., 2015; Yang et al., 2018) between 2012-2017 show that *R. strykeri* occurs in mountain forests of the middle-south segment of the Gaoligong Mountains (GLGMs) in northern Sino-Myanmar border regions. The species' east and west range borders are divided by the N'mai Hka River and Salween River (Fig. 4.3).

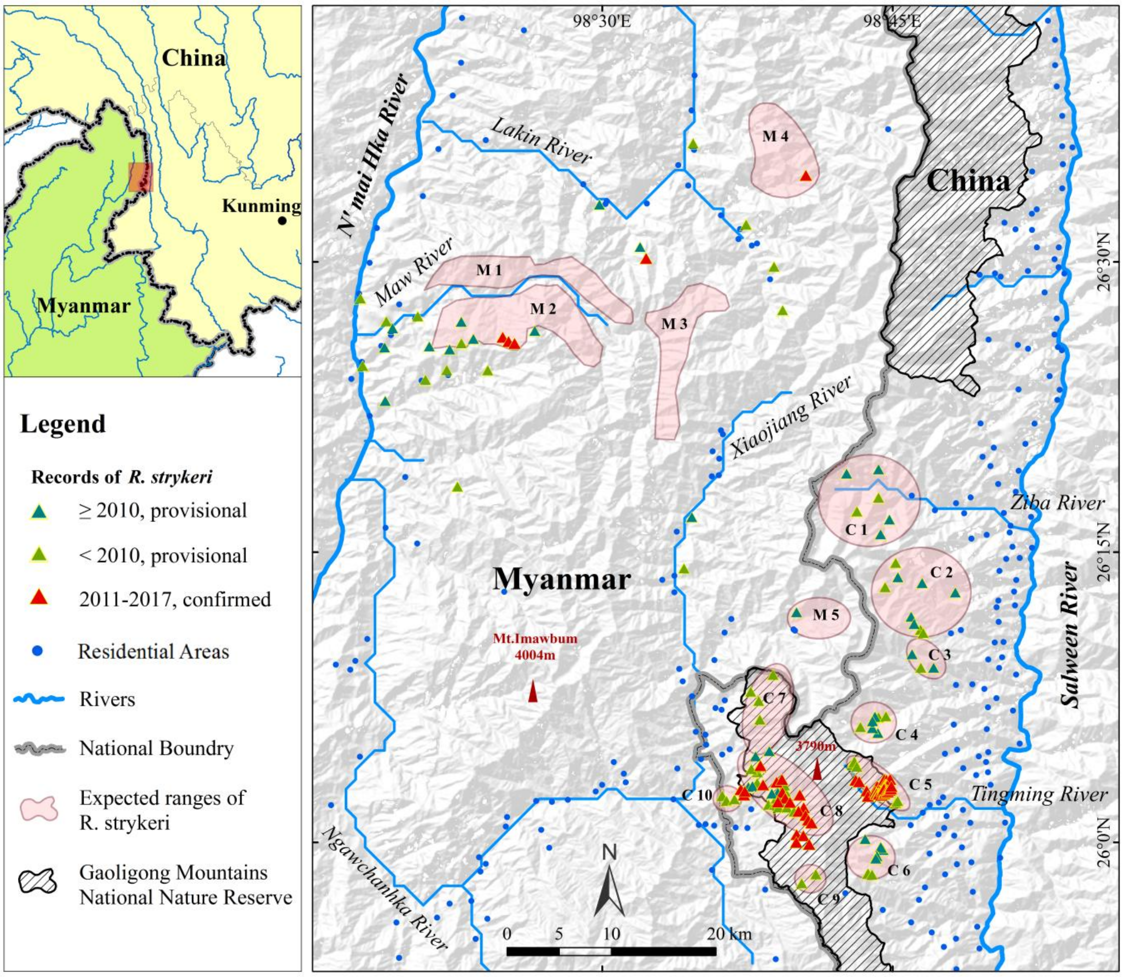

Figure 4.3 Distribution of *Rhinopithecus strykeri* in Sino-Myanmar border. Red triangles: recent records being confirmed by direct sighting and camera trapping. Green triangles: historical records obtained from interview-based surveys and specimens collected from local people. This map was modified from Figure 9 in Meyer et al., (2017) and the population information in China was updated by Yin Yang in 2019. See Appendix A in Meyer et al. (2017) for detailed information of proposed *R. strykeri* sub-populations M1-M5 and C1-C10.

In Myanmar, the species distribution range mainly encompasses the forests in the Imawbum Mountains (a mountain range on the eastern slopes of the Gaoligong Mountain System) surrounded by the Lakin, Maw and Ngawchanhka (Xiaojiang in Chinese) rivers, covering approximately 560 km^2 (N26.35°-26.63°; E98.35°-98.68°) in northern Kachin State (Fig. 4.3). Initially, interview evidence suggested three distribution ranges and three or four sub-populations for this species totalling 260-330 individuals in the Maw River area. However, based on more recent interview information along with skull and/or skin collections from the local villagers, Meyer et al. (2017) estimated that there are probably at least three to four complete domestic sub-populations (M1-M4) and one transboundary sub-population (M5) in Myanmar (Fig. 4.3). That said, field encounters and camera trapping records only confirm three subpopulations: one sub-population north of the Maw River (Group M1/M3: <100 individuals, confirmed by direct sightings and being suggested to be merged as same sub-population due to their close distance and individuals are found within the gap distance of the M1 and M3), a second sub-population south of Maw River (Group M2: 40-50 individuals, confirmed by direct sightings and camera traps), and a third sub-population northeast of Lakin River (Group M4: > 10 individuals, confirmed by direct sightings) (Meyer et al., 2017).

In China, the distribution of *R. strykeri* covers both the western and eastern slopes of the GLGMs in Lushui County, Yunnan Province. Based on interview surveys, Ma et al. (2014) estimated that there might be as many as 490-620 individuals in ten sub-populations (C1-C10 in Fig. 4.3) located around

N25.96°-26.32° and E98.59°-98.81° in China’s GLGMs. Among these estimated sub-population, only Pianma sub-population (C5: > 100 individuals) has been confirmed by field observation and camera trap records (Long et al., 2012; Li et al., 2014; Chen et al., 2015). The sub-population C10 has probably already been extirpated, as I observed that the forest of the C10's area has been seriously degraded and surrounded by human settlements. C7 and C2 are transboundary sub-populations reported by interviewees and the C2 thus may be also the transboundary group M5 (Meyer et al., 2017). In 2015, my field team and I discovered the Luoma sub-population (C8: > 70 individuals) via intensive field study and camera trapping (see details in Chapter 5).

Based on the above information there appear to be only five confirmed sub-populations (M1/M3, M2, M4, C8, and C5) of this species based on field surveys. As a result, Meyer et al. (2017) cautiously estimate a minimum of only 400 individuals existing in the Sino-Myanmar border areas.

4.3 Habitat and Environment

4.3.1 The Gaoligong Mountains

The Gaoligong Mountains[1], which sit at 600 m to 6318 m a.s.l and form the western wall of the Salween River Valley, are approximately 500 km long, crossing five latitudes (N24°-29°) and running north to south in the Northern Sino-Myanmar border region (Chaplin et al., 2005). The large variation in elevational gradient, the great latitudinal range, diverse host rocks and monsoon conditions within the GLGMs create a variety of microclimates that lead to increased diversity of flora and fauna in this region (Chaplin et al., 2005), including 4294 seed plant species (Li et al., 2000), 486 avian species (Dumbacher et al., 2011), and 115 mammal species (Xue, 1995). This mountain also sits at the nexus of

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[1] Gao-高-li-黎-gong-贡 (or Gauri Gvong in Lawwaw language), which means "The Mountain of the Gaoligong Tribe", was originally named by ancient Jingpo people and has been accepted by 16 other ethnic groups in the Sino-Myanmar area. This name was first recorded in Fan Chuo（樊绰）'s book *Yunnan annals*（云南志）, which was completed in the Tang Dynasty around the 860s.

three globally important biodiversity hotspots: the Himalayan, Indo-Burmese, and the mountains of Southwest China (Meyers et al., 2000), and forms the core part and western boundary of the Three Parallel Rivers World Natural Heritage (UNESCO, 2003). Compared to the prominent biodiversity, it is worth noting that flat surface area (slope < 3°) accounts for only 7.8% of whole GLGMs region (Chaplin et al., 2005), while the lowland agricultural area accounts for 5.57% of the total land in Nujiang Autonomous Prefecture, but these areas support more than 70% (374,000 people in 16 ethnic groups in China, including 173,000 living in extreme poverty) of the total local human population (Basnet et al., 2017).

4.3.2 Climate

The GLGMs are found in an ocean-continent transitional climate and are significantly affected by southwest monsoons form the Indian Ocean (Xue, 1995). From the foot of the mountain to its top, there are concretely seven different types of climate zones, including: south subtropical (< 1300 m a.s.l.), mid subtropical (1300-1800 m a.s.l.), north subtropical (1800-2100 m a.s.l.), warm temperate (2100-2700 m a.s.l.), mid temperate (2600-3000 m a.s.l.), cold temperate (2900-3700 m) and subfrigid (3600-4200 m a.s.l.) (Xue, 1995; Xu, 1997, Table 4.1). Using the 2012-2018 monthly meteorological data provided by the Lishui County Meteorological Bureau, I calculated the mean annual precipitation increases from 903 mm and the mean annual temperature decreases from 13.3 ℃ on the eastern slope (Luzhang Township, 1804 m a.s.l.) to 1,926 mm and 7 ℃ at the mountain top (Pianma Pass, 3,184 m a.s.l., Xu, 1995), and then the mean annual precipitation decreases to 1,386 mm and the mean annual temperature increases to 13.6 ℃ on the western slope (Pianma Township, 1,786 m a.s.l.) (Fig. 4.4). The unique multi-level climate and a subtropical-subfrigid transitional ecosystem result in a special climate landscape that there are four seasons in a same mountain and the microclimates are different within five km.

Table 4.1 Vertical climatic zones in the Gaoligong Mountains.

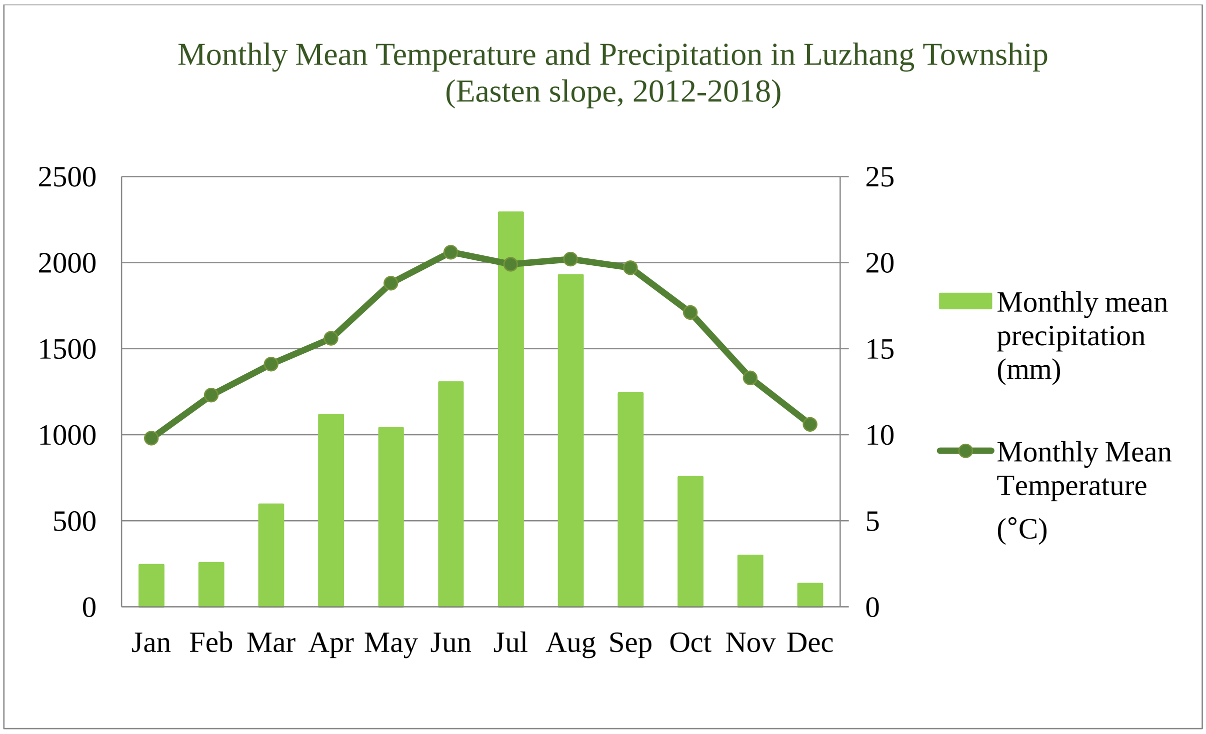
Climatic zones	Elevation (m)	≥10℃ Accumulated temperature*	Days	Mean annual temperature	Mean temperature in coldest month
Subtropical Zone	1,800 - 2,100	4,200 - 5,200	260 - 310	13 -16	7 - 9
Temperate zone	2,100 - 2,700	3,200 - 4,200	210 - 260	11 - 13	2 - 7
	2,700 - 3,000	1,600 - 3,200	140 - 210	7 - 11	1 - 4
Subfrigid Zone	3,000 - 3,600	< 1,600	< 140	< 7	< 1

***Accumulated temperature is defined as the accumulated excess of temperature above a given standard temperature.**

Source: Xue, 1995

4.3.3 Habitat vegetation

R. strykeri inhabits mountainous forests with steep slopes at altitudes between 1720 and 3190 m a.s.l. in Myanmar (Geissmann et al. 2011) and between 2200 - 3300 m a.s.l. in China (Ma et al. 2014; Chen et al., 2015; Ren et al., 2017). Correspondingly, along the elevational gradient, vegetation includes semi-moist evergreen broadleaf forest (1500-2200 m a.s.l.), mid-montane moist evergreen broadleaf forest (2200-2800 m a.s.l.), temperate-cool coniferous (Yunnan Hemlock *Tsuga dumosa* - broadleaf mixed forest) forest (2700-3200 m a.s.l.), cold-temperate bamboo bushes (2700-3600 m a.s.l.), and cold-temperate brush meadow (3000-3600 m a.s.l.) (Xue, 1995; Li et al., 2000) (Fig. 4.5).



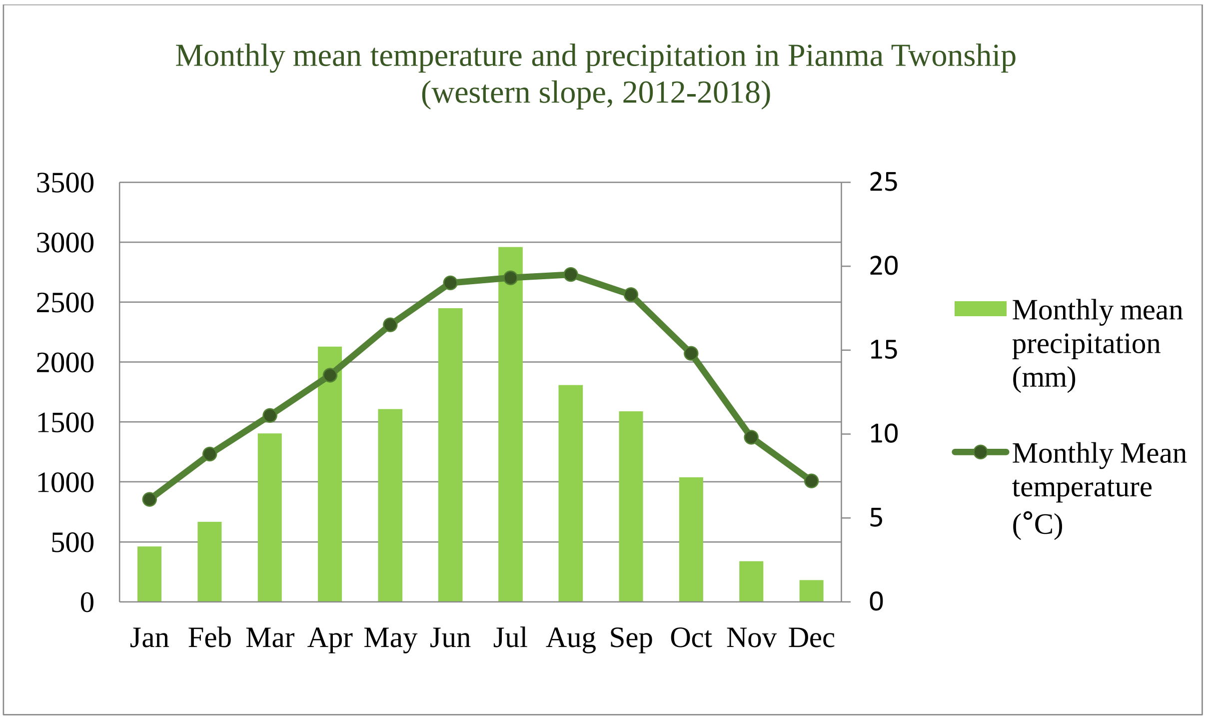


Figure 4.4 Monthly mean temperature and monthly mean precipitation in Luzhang Township and Pianma Township during in 2012 -2018.

According to Xue (1995), Xu (1997), Li et al. (2000) and my own field observations, the three vegetative types that mainly appear in the habitat of this species are as follows (see also Table 4.2):

Semi-moist evergreen broadleaf forest on the eastern slopes of GLGMs in China is largely replaced by *Pinnus yunnanensis* at lower elevations or *Alnus nepalensis* forests at higher elevations along with secondary shrubs and herbs due to severely anthropogenic disturbance. Primary forest only remains in very precipitous mountainous regions. In Myanmar, there are still large areas of semi-moist evergreen broadleaf forest remaining on the western slopes of GLGMs. However, the spread of large scale commercial logging and slash-and-burn agriculture have left the habitat of Myanmar's monkey populations riddled with a thousand gaping wounds. The communities of the semi-moist evergreen broadleaf forest in this region appear to be dominated by few species (e.g. *Castanopsis fleuryi, Lithocarpus truncatus*), but are also mixed with other evergreen or deciduous broadleaf trees. Among these trees, half are tropical, and half are temperate, species.

The mid-montane moist evergreen broadleaf forest is distributed on the top of the subtropical vegetation belts. Due to abundant rainfall, this vegetation is well-developed and makes up the main part of the GLGMs forest system, maintained in a relatively undisturbed state. The forest canopy is primarily made up of *Lithocarpus, Cyclobalanopsis, Castanopsis, Manglietia, Michelia, Schima,* and *Machilus,* and most tall trunks are covered with epiphytes and moss. There are also some deciduous species in this forest type, including *Fagus*, *Litsea*, *Acer*, *Prunus*, *Sorbus*, *Schefflera*, *Dipentodon sinicus,* and *Tetracentron sinensis* that are found in the secondary forest level. If this vegetation belt is destroyed by either human or natural disturbances then the land will become naturally dominated by mono-dominants, like the *Alnus nepalensis* forest.

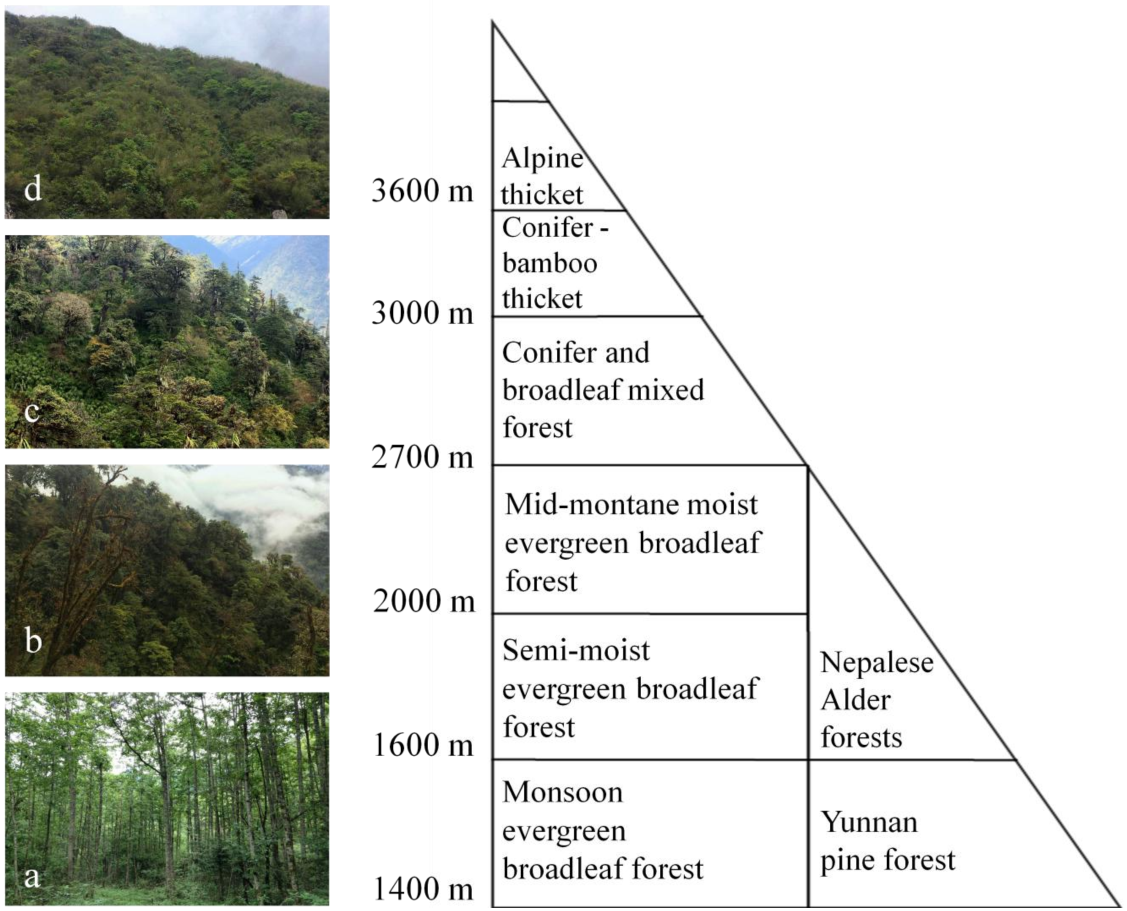


Figure 4.5 Typical vegetation zones in Gaoligong Mountains. Some typical forest landscapes in Gaoligong Mountains: a. degraded secondary forest (Nepalese Alder forest); b. mid-montane moist evergreen broadleaf forest; c. conifer and broadleaf mixed forest; d. bamboo and shrub thicket.

Table 4.2 Dominant tree species of different forests in the Gaoligong Mountains (Source: Xue, 1995)

Dominate species: > 20% growing stock

Location	Forest Type	Tree Level	Tree Species	Mean Height (m)	Mean DBH (cm)	Number of Quadrat (Size: 0.1 ha)	Average seeding number per hectare	Mean growing stock (%)	Dominated species
Yaojiaping	Mid-montane moist evergreen broadleaf forest (*Castanopsis* - *Lithocarpus* forest)	1	*Quercus glauca*	28.5	89.7	6	60	30.8	Yes
			Lithocarpus leucostachyus	33	103.4	3	30	24.7	Yes
			Schima argentea	31.6	71.7	5	50	22.4	Yes
			Lithocarpus hancei	28.8	67.1	5	50	12.4	
		2	*Rhododendron* spp.	10.8	21.9	12	120	40.3	Yes
			Quercus glauca	9.4	13	9	90	20.2	Yes
			Vaccinium bracteatum	11.2	19	6	60	14.8	
Yaojiaping	Mid-montane moist evergreen broadleaf forest (*Cyclobalanopsis* forest)	1	*Cyclobalanopsis oxyodon*	24.8	58.7	14	140	59.6	Yes
			Lindera flavinervia	23.3	51.4	4	40	14.3	
			Pinus armandii	35	80	1	10	11.4	

		2	*Rhododendron* spp.	12.5	24.3	12	120	33.3	Yes
			Cyclobalanopsis oxyodon	13.6	24.5	12	120	31.5	Yes
			Vaccinium bracteatum	12	14.5	8	80	6.1	
Pianma Pass	Mixed hemlock-evergreen broadleaf forest	1	*Tsuga dumosa*	70	20	-	-	100	Yes
		2	*Rhododendron* spp.	26	12	-	-	58.9	Yes
			Symplocos spp.	22	9	-	-	12	
Yaojiaping	Degraded deciduous broad-leaved forest	1	*Alnus nepalensis*	11.6	13.9	37	370	81	Yes

The **Yunnan hemlock-broadleaf mixed forest** is a subalpine coniferous broadleaved forest characterised by *Tsuga dumosa*, which is a representative montane species in GLGMs occurring under temperate and cool ecological conditions along with other deciduous broadleaf forests dominated by various species of *Acer*, *Symplocos*, *Betula*, *Lindera*, *Lithocarpus Schefflera* and *Gamblea ciliata* or *Rhododendron* forests with shrubs or bamboo thickets to structure blocks of mixed forest.

4.4 Social Organisation and Ranging Patterns

Based on video data and direct observation of the Pianma band, Li et al. (2014) provided the first evidence that *R. strykeri* lives in multilevel societies similar to the other four snub-nosed monkey species. This means their band is primarily composed of several one male-multi-female units (OMUs) and one or two all male units (AMUs) with a mean age-sex ratio of adult male-to-adult female of 1:2.1, a mean infant-to-adult female ratio of 1:4.7, and a mean adult-to-immature ratio of 2.5:1. By using 222 photos from camera traps, Chen et al. (2015) identified the 90 individuals in this band that are mainly found in the elevational range of 2400-3100 m in a 22.9 km^2 home range. This elevational range is roughly consistent with the 2600-3100 m range suggested by Ma et al. (2014) based on interview surveys. The 22.9 km^2 home range, however, is much larger than Li et al. (2014)'s 12 km^2 home range calculation based on GPS records. If Long et al. (2012)'s track coordinate points are added, the home range by Chen et al. (2015) would increase by one km^2. These varied home ranges can be explained by the fact that there are probably at least two sympatric bands of snub-nosed monkeys living in Pianma area with partially overlapping home ranges (Ma et al., 2014; Pu Sancai & Liu Pu, pers. comm., 2016). Therefore, more field work is needed to confirm home range and sub-population conditions in Pianma in the future.

I do not have enough of my own data to analyse home range area for the Luoma band, but I did collect five days of path length records from 3rd - 7th January 2016 (GPS coordinates were recorded every hour starting from leaving the sleeping site in the moring to entering a new one at night) that show a mean daily range length of 984 m, ranging from 525 m to 2330 m, with mean speeds from 60.8 m/h to 211.8 m/h. The day they moved the least they seemed to stay primarily around their sleeping sites. I found the mean elevation of sleeping sites to be 3086 m a.s.l. and the mean distance between sleeping sites on the first four days I tracked them to be 303 m. However, the band crossed a valley and changed its sleeping site to another ridge on the last day, significantly increasing this distance. These results suggest that *R. strykeri* is likely similar to other snub-nosed monkey species that often stay in the same areas for successive days, especially when they use winter foods (Su et al., 1998; Ren et al., 2009). Moreover, all six sleeping sites (one in autumn and five in winter) I recorded are all above 3000 m a.s.l., which shows that they sleep at a high elevation and then forage by descending toward the valleys, returning to high-elevation areas to sleep. Quan et al. (2011) proposed that snub-nosed monkeys use high-elevation range areas that are correlated with solar radiation and sunshine duration. In my study, I only recorded one time that the monkey band descended to lower elevations in winter because of snowing, which is similar to *R. bieti* (Kirkpatrick& Long 1994; Li et al., 2008) and *R. roxellana* (Li et al., 2010).

4.5 Conservation Assessment

R. strykeri has been listed as Critically Endangered by the IUCN Red List of Threatened Species due to criteria A4cd (Geissmann et al., 2012). Meyer et al. (2017) re-evaluated the conservation status of this species recently and they suggest the current IUCN Red List status remain unchanged due to the scarcity of reliable data (see the criteria in Appendix B in Meyer et al., 2017). The monkeys have faced a number of major threats such as hunting, wildlife trade, and habitat destruction. Long term survival of this species requires conservation management to address these problems.

4.5.1 Poaching pressure

The Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) have listed all snub-nosed monkey species on Appendix I that prohibits international trade in specimens of these species except when the purpose of the import is not commercial. In China, all snub-nosed monkey species have been listed as first class national protected animals by *Wild Animal Conservation Law of the People's Republic of China*. In Myanmar, *R. strykeri* has been required to protect under the Myanmar's *Wildlife Protection and Protected Areas Law*. Therefore, hunting and trading of this species are illegal both in international and local levels.

In Myanmar, at least 48 individuals have been harvested in 27 hunting events by 35 hunters from 15 villages in Myanmar between 1992 and 2013; at least 22 of these individuals were hunted between 2009 and 2010 (Appendix A, pp. 46-56 in Meyer et al., 2017). After 2013, I noted four additional hunting/poaching events in Myanmar:

1.) two infants of Shortridge’s langurs (*Trachypithecus shortridgei*) were smuggled from Myanmar to China in April 2015;

2.) a freshly killed adult female *R. strykeri* was sold to a person in China in March 2016 (Fig. 4.5a); and

3.) the Nujiang prefecture forest police seized 44 monkey skeletons smuggled from Myanmar in April 2017, including: one *R. strykeri*, 16 *T. shortridgei*, 23 Assamese macaques (*Macaca assamensis*) and four stumptail macaques (*M. arctoides*) (Fig. 4.6 b).

4.) a adult male Skywalker hoolock gibbon (*Hoolock tianxing*) were caught in Myanmar and then sold to a villager in China as pet animal in January 2012. This individual then was rescued by the Forestry administration of Gongshan County in April, 2018.

In China, Ma et al. (2014) reported 15 *R. strykeri* were hunted before 2000 and four after the year. Although there are relatively few incidents of hunting *R. strykeri* recorded in China in recent years, a camera trap in Luoma forest captured images of two hunters with a shotgun in the monkey's core range in 2016 (Fig. 4.6 c).

Hunting has a long, historical tradition among the Lisu, Jingpo, Nu and Lawwaw people in the GLGMs region, including as a sense of identity. Hunting with traditional methods (e.g. cross bows and snares) and the traditional animist beliefs have little effect on wildlife populations (Harris & Shilai, 1997). However, today's demand for bush meat is not just about meeting the needs of supplemental protein and self-contained traditional medicine for local people. Meyer et al. (2017) report that animals including bears (*Ursus* spp.), muntjak (*Muntiacus* spp.), serow (*Capricornis* spp.) and wild boar (*Sus scrofa*) sell from 500-10,000 RMB (73-1458 USD)/animal depending on the species and weight of body parts, while macaque and langur heads or skulls can be sold for 100-150 RMB (14.6 USD), and bones are sold for 88 RMB (13 USD)/kg. Hunting and wildlife trading enables some hunters to earn 10,000 RMB (1458 USD) annually (Momberg et al. 2010; Meyer et al., 2017). Obviously, the illegal wildlife trade will stimulate changes in locals' hunting behaviours. For example, some local ethnic groups have turned to using factory-made shotguns with bullets and iron traps to cope with the surge in demand for bush meat and traditional medicine.

Figure 4.6 Recent evidence of hunting pressure on *Rhinopithecus strykeri*. a. a female adult shot in the neck by automatic firearm in Myanmar and then being sold to China in April 2016; b. 44 monkey skeletons smuggled from Myanmar in April 2017 (photo taken in February 2018); c. caught by a camera trap, two hunters with a gun appeared in Luoma *R. strykeri*'s core habitat in May 2016.

4.5.2 Logging and deforestation

Before 2016, habitat destruction caused by large-scale commercial logging in the habitat of *R. strykeri* in Myanmar was a very serious risk to the species' population sustainability. Blasting from mechanical logging and the creation of large logging roads involves the use of heavy equipment and winches that can reach ridges that incline up to 70 ° (Geissmann et al., 2012). Because the weathering layer of the metamorphic rock mass is rather thick in GLGMs, the loss of tree cover can then result in frequent landslides, rockfall and heavy debris flow following rain that causes even further damage for vegetation at lower areas (Zhang et al., 2007; Geissmann et al., 2012; Yin Yang, Pers. observ.). In Myanmar habitat destruction also comes from disordered slash and burn agriculture, which expands from the valley of N'mai Hka river up to elevations of 2,000 (2100) m (Geissmann et al., 2012). The moor-burning also frequently causes uncontrolled forest fires crossing the border to the GLGMNNR, thus becoming a threat to *R. strykeri*'s habitat in China (e.g. two border-crossing forest fires occurred in January 2017). Moreover, firewood collections are also traditional and cultural customs of ethnic groups in protected areas. This activity is often taken place at the edges of the forests. This activity sometimes occurred not for family consumption but also for fire pit culture (e.g. the fire pit is the place where the Lisu and Jinpo people discuss important clan or village affairs together and it is a symbol of the light and continuation of the family and the ethnic group) even the electricity being able to use at the same time. Therefore, the frequent firewood collection as many others bring threats on the integrity of the monkeys' habitat. Between 2000 and 2015, at least 91.2 km^2 of *R. strykeri* habitat was lost in Myanmar due to logging and slash-and-burn cultivation (Chapter 8). The good news is that the large-scale commercial logging has primarily been stopped (Ngwe Lwin, pers. comm., August 2017) due to the Chinese Government strictly implementing a transboundary ban on illegal activities such as logging and mining since 2015 (Fig. 4.7).

The transboundary logging ban has also put more pressure on the protection of forests in the GLGMNNR. One GLGMNNR internal report (2017) states that the illegal logging of valuable trees has become uncontrolled in the reserve, and is occurring more frequently and in more places than before the ban. In addition, the loggers have used firearms to deter forest rangers, which have resulted in at least one ranger being injured when the ranger tried to stop a logging event in the reserve. The ranger sometimes also seem to be hesitant to let the people in the same village getting into trouble. Also, when some loggers stay in the mountain for timber, their relatives will try to monitor the forest rangers in same villages in order to inform the loggers to evacuate in time when the rangers also enter the mountain for patrol. More of logging impacts and driving factors will be expanded on in Chapter 6.

Figure 4.7 Pianma town is the main entry site for logs from Northern Myanmar into China. Pictures taken in the same place in 2014 (source from mongabay.com) and 2018 shows that the logging has stopped.

After *R. strykeri* was discovered in the Sino-Myanmar border areas, staff from FFI Myanmar programs and GLGMNNR, along with a number of Chinese scientists and government officials from the forestry department, have made great efforts to secure the long-term survival in this species. There are a number of conservation successes, such as dramatic reductions in hunting records of *R. strykeri* in Myanmar. This came about after intensive awareness campaigns were conducted in all 54 villages in the monkeys' area and comprehensive surveys of two sub-populations of *R. strykeri* in China enabled people to recognise some of its natural histories. My colleagues and I describe all past conservation achievements and suggest future steps towards long-term protection for *R. strykeri* in section 5 in the technical report *Conservation Status of the Myanmar or Black Snub-nosed Monkey* (see Meyer et al., 2017).

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Chapter 5 The Critically Endangered Black Snub-Nosed Monkey *Rhinopithecus strykeri* Found in the Salween River Basin, China

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The snub-nosed monkey genus *Rhinopithecus* was previously thought to comprise four rare, threatened primates (Tonkin snub-nosed monkey *R. avunculus*, gray snub-nosed monkey *R. brelichi*, golden snub-nosed monkey *R. roxellana* and black-and-white snub-nosed mokey *R. bieti*), known only from isolated parts of southern China and northern Vietnam (IUCN, 2016). *Rhinopithecus strykeri*, known as the Myanmar snub-nosed monkey, or Nujiang snub-nosed monkey in China, was discovered in 2010 in forests above 1,720 m altitude in Kachin state, north-east Myanmar (Geissmann et al., 2011). A second population was discovered the following year in Pianma on the western slopes of Gaoligong Mountains, in Yunnan, China (Fig. 5.1; Long et al., 2012). *R. strykeri* primarily inhabits mid-montane moist evergreen broad-leaved forest and coniferous broad-leaved mixed forest (Geissmann et al., 2011; Chen et al., 2015). Genetic evidence indicates that it separated from the black-and-white snub-nosed monkey clade in the Late Pleistocene, c. 0.24 million years ago (Liedigk et al., 2012) when two major barriers, the Mekong and the Salween Rivers, may have physically isolated segments of the parent population. All species of snub-nosed monkeys are reported to live in a multilevel or modular society, composed of several one male-multi-female units that travel together to form a band, and one or more all-male units that follow the band (Kirkpatrick & Grueter, 2010).

Interviews (Geissmann et al., 2011; Ma et al., 2014) suggested there may be up to 14 sub-populations of Myanmar snub-nosed monkeys, with a total population of < 950 individuals, in four separate areas close to the Sino-Myanmar border: one in Pianma, China, and three in Myanmar. In China field observations of *R. strykeri* in Pianma have been conducted since 2011 (Chen et al., 2015), although limited behavioural data have been collected because of the rugged terrain and long rainy season from April to October. This band, with > 100 individuals, lives at 2,400 – 3,300 m and has an estimated home range of 22.9 km^2 (Chen et al., 2015).

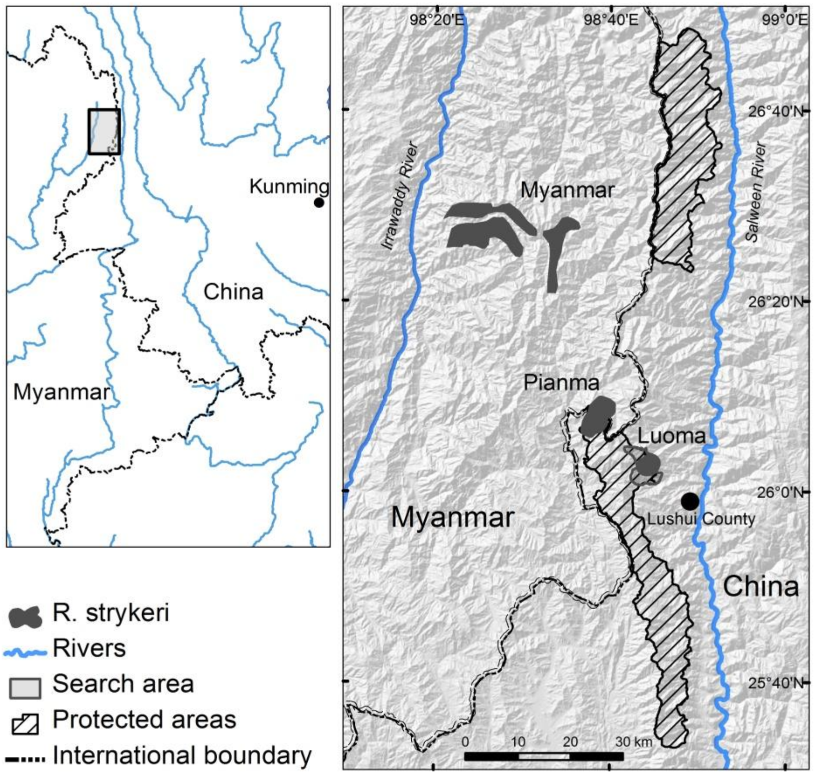

Figure 5.1 Locations of the known populations of the *Rhinopithecus strykeri* in the Sino-Myanmar border region.

Ma et al. (2014) predicted the species would also occur on the eastern slope of the Gaoligong Mountains, in China, where we searched during October 2013–September 2015 (see search area in Fig. 5.1). By using outline transects (Plumptre et al., 2013), one or two researchers and two field guides searched the species along each sub-ridge in the habitat, during 08.00 – 19.00, 1 – 3 weeks per month depending on the weather. We located the species, and took >

200 photographs and 27 videos (Fig. 5.2). We also found evidence of chewed branches and faeces consistent with those collected from the species at other sites.

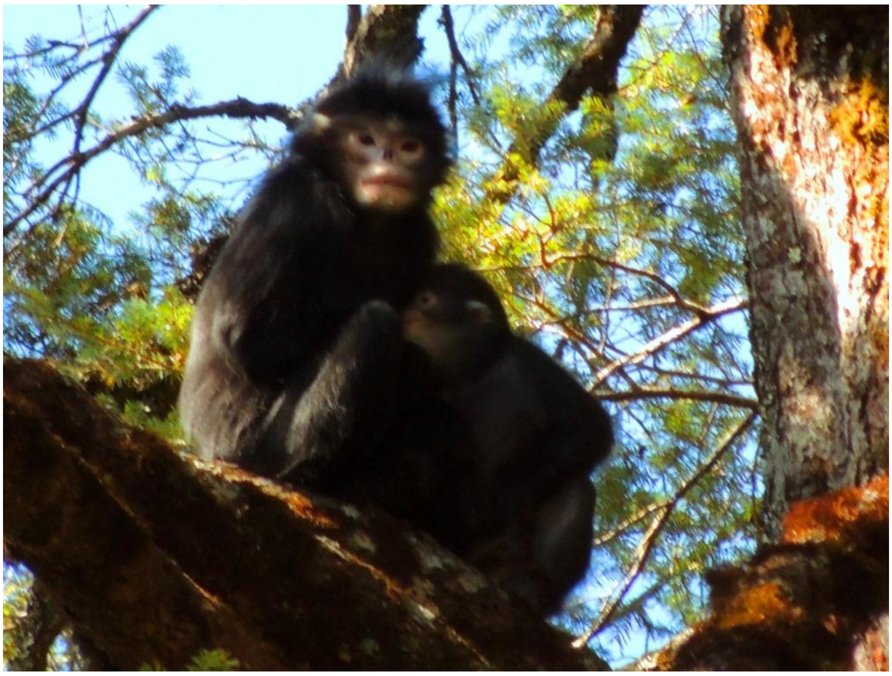

Figure 5.2 Mother and infant of *Rhinopithecus strykeri* in the newly discovered population in Luoma, China, near the border with Myanmar (photo by Dong Shaohua).

In addition to direct observation, we also set up 19 camera traps (Acorn Ltl-6210 MC), from April/2015 to September/2015, on wildlife trails and five food trees at a height of 15-20 m. The trees included three Toufucha (*Gamble aciliata*) and two Shanxiangguo (*Dodecadenia grandiflora*) which were identified by our local field guides as feeding trees for black snub-nosed monkeys. Using the camera traps we obtained 54 successive photos of the species in one event. The camera trap was in one of the toufucha tree crowns.

We collected faeces on May 5 2015, at 3180 m, and confirmed they were the faeces of Myanmar snub-nosed monkeys based upon their shape, size and contents, which included the remains of leaves and conifer needles. At 19.00 on September 16, 2015, we observed Myanmar snub-nosed monkeys fighting and vocalizing loudly in a patch of hemlock Tsuga dumosa–bamboo/rhododendron forest; they then moved to a sleeping site on the ridge of the mountain at about 19.30. The next day we followed them during 7.00 – 16.30. We ceased to follow the band thereafter because of heavy rain and dense fog. We had followed the band for approximately 9½ hours, at a mean travel speed of 16.82 m per hour. Assuming that the band would have continued travelling and foraging for at least another 2½ hours, we estimate the snub-nosed monkeys would have travelled c. 2018 m on that day.

To estimate band size we reviewed the 54 infrared photographs, identifying 18 adult or subadult monkeys, with 10 appearing in infrared photographs during 17.53 – 17.58 and eight in other infrared photographs during 18.06 – 18.10. The mean passing rate of snub-nosed monkeys was 2.3 individuals per minute. Given that several distinguishable adult males appeared in these photographs, and none of the monkeys were carrying infants, we assume these individuals were members of an all-male unit. In addition, on one occasion we observed c. 20 infant and juvenile monkeys. Given the sex-age ratios of males to females (1:2.1), females to infants (4.1:7), and adults to immatures (2.5:1) in the Pianma *R. strykeri* population (Li et al., 2014), we tentatively estimate that this population comprises > 70 individuals.

The Luoma population lies c. 14 km in a straight line south-east of the Pianma population reported by Long et al. (2012) and 80 km south-east of the Myanmar site reported by Geissmann et al. (2011). The Luoma population lives in the core zone of the Gaoligong Mountain National Nature Reserve, one day’s walk from the nearest village. The forests are mostly mixed broad-leaf–hemlock forest or hemlock–rhododendron forest at 2,600–3,300 m, and are mostly pristine, with little anthropogenic disturbance. Compared to the declining populations in Myanmar (c. 300 individuals; Geissmann et al., 2011) and Pianma (c. 100 individuals; Chen et al., 2015), which have suffered from the threat of hunting and habitat degradation (Geissmann et al., 2011; Yin Yang, pers. obs.), the Luoma population may have potential to be a source population for the formation of new additional *R. strykeri* populations, and therefore requires strict and effective conservation management.

Additionally, living in forests dominated by hemlock, the ecology of the Luoma population may differ from that of the other populations, which inhabit moist evergreen broadleaved forests. Based on 358 interviews with local people in the region, individuals of this population may travel to forests outside the Reserve, close to Chengan Township, where they may occasionally be hunted by the Lisu people (Ma et al., 2014). The border of the reserve may therefore need to be adjusted to ensure the full protection of the population. We aim to conduct further research and to develop a conservation action plan to protect nearby forests, which also harbour other threatened species, such as the Endangered red panda *Ailurus fulgens styani*, Vulnerable Mishmi takin *Budorcas taxicolor taxicolor*, and Vulnerable Sclater's monal *Lophophorus sclateri*. In addition, further surveys to locate other potential *R. strykeri* populations and to determine accurately the population size in the Sino-Myanmar region are required for a full assessment of the conservation status and needs of this Critically Endangered primate.

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Chapter 6 First Insights into the Feeding Habits of the Critically Endangered Black Snub-Nosed Monkey *Rhinopithecus strykeri* (Colobinae, Primates)

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6.1 Introduction

Solving the challenges faced in locating, acquiring, and processing a nutritionally balanced diet can act as a strong selective pressure impacting a species' morphology, sensory ecology, dietary adaptations (Grueter et al., 2009b; Huang et al., 2017), habitat use and ranging (Matsuda et al. 2009; Behie and Pavelka 2015), reproductive strategies (Gould et al., 2015), and population density and group size (Hanya & Chapman 2013; Ma et al., 2017). In the case of frugivorous primates, factors such as patch productivity, patch distribution, and whether individual trees of the same species fruit synchronously or asynchronously, play a critical role in foraging decisions (Garber, 1989; Chapman et al., 2012). In contrast, folivorous primates are reported to preferentially feed on young leaves over mature leaves, and to only consume certain parts of the leaf. This is often attributed to these items having higher ratios of protein-to-fiber and/or low levels of secondary compounds, which are known to influence colobine feeding. More recently, it has become apparent that all primates may select food items in order to ensure they are consuming a nutritionally balanced diet (Felton et al., 2009; Behie & Pavelka, 2012; Righini et al., 2017). Here, we refer to selection as the individual act or decision to consume one plant species or food type (i.e., plant part) over another. We define a feeding preference as the consistent selection of one food type or plant species over another, when both resources are available to the forager.

Given that many primates inhabit anthropogenically disturbed forests whose plant species composition and distribution may differ from that of undisturbed forests, a detailed understanding of a species' feeding ecology and dietary preferences is critical for maintaining and conserving wild populations (see Chaves et al., 2012; Milich et al., 2014; Behie & Pavelka, 2015).

Studies linking feeding ecology and conservation are especially important for Asian primates, as 73% of 119 species are Vulnerable, Endangered or Critically Endangered, and 95% have declining populations, principally in response to the needs of a large and expanding human population, and the destruction and transformation of natural landscapes for purposes of industrial agriculture, logging, transportation, oil and gas exploration, and the grazing and ranching of livestock (Estrada et al., 2017). One primate radiation of particular concern is snub-nosed monkeys, genus *Rhinopithecus*. Snub-nosed monkeys are members of the subfamily Colobinae and characterized by an enlarged, low acid, and multi-chambered stomach that contains a diverse microbiome for the breakdown of difficult to digest complex carbohydrates present in leaves and tree bark (Kay & Davies, 1994; Lambert, 1998; Guo et al., 2018). Currently, there exists five species of snub-nosed monkeys distributed in Vietnam, Myanmar, and China. Three of these species, the Tonkin snub-nosed monkey (*R. avunculus*), the black snub-nosed monkey (*R. strykeri*), and the gray snub-nosed monkey (*R. brelichi*) each has a remaining population numbering less than 700 individuals (Li et al., 2018). Population estimates for the black-and-white snub-nosed monkey (*R. bieti*) and the golden snub-nosed monkey (*R. roxellana*) are < 3000 and 22,500 individuals respectively (Li et al., 2018). Based on whole genome sequences of four of the five *Rhinopithecus* species, genetic heterozygosity is extremely low (0.015–0.068%, Zhou et al., 2016) increasing the likelihood of population decline and extinction.

In 2010, the black snub-nosed monkey or Myanmar snub-nosed monkey (*Rhinopithecus strykeri*) was discovered in the Gaoligong Mountains (GLGMs) of northeastern Kachin state, Myanmar (Geissmann et al. 2011). A second population of this species was subsequently found in northwestern Yunnan, China (Long et al. 2012). Black snub-nosed monkeys are distributed across a range of only 3,575 km^2 from E98°20′–98°50′, N25°40′–26°50′ in the region of the Sino–Myanmar border (Ren et al. 2017). This primate species faces a high risk of extinction from hunting and forest destruction (Meyer et al. 2017). Based on field surveys (2010–2017) and camera trap evidence, only 400 individuals distributed across five sub-populations have been confirmed in the wild (Meyer et al. 2017). Difficulties in following and observing unhabituated populations living in high-altitude forests characterized by steep mountainous terrain, thick subtropical evergreen broad-leaved vegetation, and extended raining periods with dense fog have resulted in minimal information on their behavior and ecology.

Given the importance of determining dietary preferences for developing effective conservation management plans, we collected two complementary data sets to document the black snub-nosed monkey diet. First, we conducted field observations of a wild population. Second, we initiated a series of cafeteria-style feeding trials of plant foods collected from the native habitat of black snub-nosed monkeys and fed these samples to the only two known individuals in captivity. This method was chosen based on the historical and effective use of cafeteria trials to identify food palatability and dietary preferences of Artiodactyls (Alonso-Díaz et al., 2008; Wang et al., 2015; Rea et al., 2017). Moreover, this technique was used to successfully identify food preferences in captive black-and-white snub-nosed monkeys (*R.*

bieti) (Wu & He, 1989), with the results later confirmed during field observations of wild populations (Kirkpatrick, 1996). This method also has been used to confirm that the food preferences of captive Douc langurs (*Pygathrix nemaeus*) were similar to the food preferences of wild populations. That is, both wild captive and wild individuals preferentially selected foods with high protein-to-fiber ratios and low levels of alkaloids, tannins, and other plant secondary compounds (Otto, 2005).

The specific aims of our study were to (1) develop a list of the plant species eaten by wild and captive black snub-nosed monkeys, (2) examine patterns of food selectivity and food preferences, (3) determine the habitat characteristics (forest type, elevation) of the plant species consumed, and (4) use these data to develop a conservation plan for this Critically Endangered primate species.

6.2 Methods

6.2.1 Study site

The field study was conducted in a forested area located between the town of Pianma (26°08′N, 98°35′E) and the town of Luzhang (26°00′N, 98°44′E) in the Lushui region of the Gaoligong Mountains National Nature Reserve (GLGMNNR). Steep peaks and deep gullies typify the landscape resulting in sharp vertical differentiation in climate, vegetation, soil, and natural plant communities (Chaplin, 2005). Following an ascending elevational gradient, the vegetation changes from well-developed subtropical evergreen broadleaf forest at 1,600–2,800 m (semi-moist evergreen broadleaf forest below 2,000 m; mid-montane moist evergreen broadleaf forest from 1900 to 2800 m) to coniferous broadleaf mixed forest (dominated by Himalayan hemlock, *Tsuga dumosa*) between 2,700 and 3,200 m, to predominantly mixed hemlock-bamboo thicket at 3,000–3,400 m or to bamboo thicket and alpine scrub from 3100 to 3,600 m. In this region, most of the old evergreen broadleaf forest below 2,300 m has been replaced by secondary deciduous broadleaf forest (dominated by *Alnus nepalensis* with some *Betula* spp., *Populus yunnanensis* and *Pinus yunnanensis*) due to forest-crop rotation by the Lisu people who live on these lands (Fig. 4.4). The forest zones used by *R. strykeri* in the GLGMs are reported to range from 1,720 to 3,300 m (Geissmann et al., 2011). The tree canopy in these forests can be divided into five strata (emergent 30–25 m, canopy 20 m, understory 10 m, shrub 5 m, and herbaceous layer 1 m) that contain a diverse set of mosses, epiphytes, parasitic plants, and lianas.

The climatic conditions are characterized by (1) an annual average temperature of between 14 and 17 ℃; (2) an average temperature during the coldest month of not lower than 7 ℃; (3) an annual rainfall of 1,000–3,900 mm (with highest rainfall totals from February to September); and (4) rainfall totals of > 50 mm during all months of the resulting in no obvious periods of drought or water shortage (Li et al., 2000).

6.2.2 Feeding study of the wild black snub-nosed monkeys

Over the course of our 16-month study period (September 2015–December 2016), we collected dietary and ranging data on one unhabituated wild population of black snub-nosed monkeys, hereafter called the Luoma group (see Yang et al., 2018, Fig. 6.1). To locate the monkeys, we constructed and walked line transects (see Plumptre et al., 2013) along each sub-ridge in their potential home range. We walked these transects between the hours of 8:00–19:00 for 7–21 days per month (203 working days) depending on weather. We followed the Luoma group when encountered, and obtained a total of 80 h (16 Sep 2015; 3–7 Jan 2016; 2–3 May 2016) of direct field observations. The number of contact hours was limited by the challenges encountered when following monkeys that moved across steep cliffs, thick forest, and in dense fog. Observations ceased when the group entered a sleeping site or the animals were lost. When possible, we observed the monkeys from a distance of 50–100 m and recorded all food items consumed that could be unambiguously identified by the researcher. We also collected discarded food remains from the ground that had bite marks, consistent with those made by black snub-nosed monkeys and recorded these as potential food items. If this food item also was consumed by the captive black snub-nosed monkeys during the cafeteria-style feeding trials (see Sect. 6.2.3), we scored it as a likely food item consumed by wild snub-nosed monkeys. Samples of all food items consumed by members of the Luoma group were dried and taken to the Kunming Institute of Botany, Chinese Academy of Sciences (KUN) for identification.

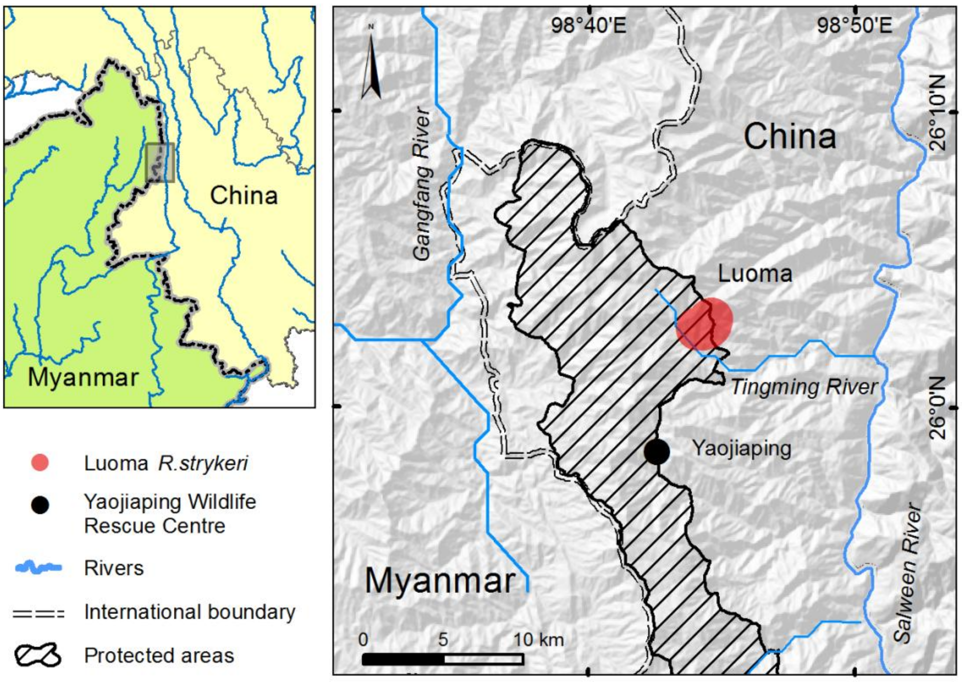

Figure 6.1 Location of the Luoma *Rhinopithecus strykeri* study band and the Yaojiaping Wildlife Rescue Centre.

6.2.3 Cafeteria-style feeding trials

We obtained additional data on the diet of black snub-nosed monkeys by conducting cafeteria-style feeding trials in the Yaojiaping Wildlife Rescue Centre (25°58′28.30″N, 98°42′35.84″E, elevation of 2510 m, Fig. 6.1), located in the Gaoligong Mountains National Nature Reserve (GLGMNNR). The center is situated in a mountain valley with easy access to *R. strykeri*'s natural habitat (Ren et al., 2017), and far from local villages. Trials were done with the only two captive individuals of the species. At the time, these animals were confiscated, both were juveniles (females) and both had been in captivity for several months. Two senior primate researchers, Professors Wen Xiao and Long Yongchen, deemed that their reintroduction into the wild would carry considerable risk; consequently, both females were housed in a 210 m^2 × 7-m-high enclosure, designed to simulate a semi-natural environment.

We presented these two captive black snub-nosed monkeys with over 600 wild plant species (containing food items such as buds, young leaves, mature leaves, twigs, flowers, fruits, and bark), lichens (n = 30 species), and bamboo shoots (n = 3). Nine to 12 plant species were provided per day. These plants were randomly harvested and collected from the primary altitudinal range (1,600–3,600 m) and forest types inhabited based on our field observations of wild *R. strykeri*. At least four large branches of each food type were collected in the monkeys' habitat each morning, of which one was used as a voucher specimen and the others for the feeding trial. At collection, the fresh branches were placed in large sealed canvas bags and brought to the rescue centre before 14:00. The branches were placed in water to keep them fresh and provisioned to the captive monkeys either that afternoon or the following day. Animals were offered food three times each day—at 7:00 (1 h before the keeper fed the captive black snub-nosed monkeys a large quantity of native plants and artificially grown foods such as pears, apples, and horsebeans), at 15:00, and at 17:00 (after the monkeys were fed at 13:00 and 16:00). This feeding schedule was used to reduce the likelihood that the monkeys consumed a food item due to hunger rather than preference or palatability. All food species were offered at least three times (including at least once in the early morning prior to the monkeys 7:00 am feeding time) on different days to confirm palatability and the monkeys' preference for each item. A digital camera or telescope was used to record the food items eaten. Although we acknowledge that some food items consumed by the captive black snub-nosed monkeys might not be included in their natural diet, given the difficulties of studying these Critically Endangered primates in the wild, cafeteria trials may represent the most effective way, at present, of obtaining dietary and ecological information that is needed to develop an effective species survivorship plan and to protect their remaining habitat.

Selectivity was defined when (1) the captive black snub-nosed monkeys consumed only a small subset of the total number of species/food types (buds, leaves, flowers, fruits/seeds, twigs, bark, petioles, bamboo shoots and pseudobulbs) presented (in this case we could distinguish satiation from selectivity if this same feeding preference was observed across multiple feeding trials and at different times of the day) or (2) the captive monkeys selected a tree species that was uncommon in their range; for example, they rejected fruits and young leaves of dominant tree species such as *Alnus nepalensis* and *Tsuga dumosa*, but favored the fruits and young leaves of uncommon species such as *Cerasus clarofolia*, *Sorbus insignis*, and *Gamblea ciliata*.We also documented the captive monkeys' first and most frequently selected species and preferred food items (e.g., leaves, fruits, buds, etc.).

A food item was designated as preferred when the monkeys exhibited any of the following behaviors: (1) giving a "Hen" vocalization when they encountered the food item and then ran directly toward the food item; (2) rapidly grabbing several branches of a particular food plant, or acted aggressively to acquire that food item and then maintained a distance from its cage mate; (3) concentrating their feeding efforts on different food items of the same plant species (e.g., leaves, bark, fruits) before moving to other provisioned food items of a different plant species; and (4) exhibiting a specific behavior indicating high arousal interest (e.g., alternating right-and-left movements of their limbs three or four times in rapid succession when first observing that food item). If both captive monkeys exhibited these same behavioral responses to the same food species/item across multiple trials, we considered this behavioral pattern to represent a ‘food preference’. Each food plant × food item was presented to the captive monkeys during each of four seasons of the year, if that food item was available in the wild. We conducted cafeteria feeding trails from November 2016 to October 2017 and combined the data for both captive individuals in our analysis.

6.2.4 Measuring wild plant species distribution patterns

Each plant species (n = 170) fed on by either the wild or captive monkeys was marked with the date collected, location (using a handheld GPS unit), habitat type (see notes in Table Appendix V), life form (tree, shrub, or herb; evergreen, deciduous, or perennial; epiphytic or climbing), characteristics of the plant (such as color of flowers, fruits and stems), altitude, and rarity index (defined as common and uncommon, based on the lead author’s general assessment of the availability of that species compared to that of the dominant species present in the environment and botanical data reported by Xue (1995) (Table 4.2). The published volumes of the Flora of China, Flora Yunnanica, and Flora of Gaoligong Mountainswere used to assist in species identification. In addition, botanists from the KUN and Southwest Forestry University were consulted in species classification, which was confirmed based on the inspection of 30,000 plant specimens collected from the GLGMs, and housed at the voucher herbarium at the KUN. Over 98% of the 170 plant species consumed by the black snub-nosed monkeys were identified to the species level.

We collected 654 plant samples (e.g., fruits, leaves, flowers) from the 170 food plant species consumed by the black snub-nosed monkeys growing at an altitude of between 1,600 and 3600 m. Additionally, we examined the collection of 30,000 plant specimens present in the herbarium in KUN and identified 2,456 voucher samples of the 170 species consumed by the monkeys. Thus, in total, we combined the ecological information obtained for the 2,456 voucher specimens with the 654 food plants samples (170 plant species) consumed by the captive and wild black snub-nosed monkeys to develop a database of the number and altitudinal distribution of potential food species within each major habitat type and to calculate species richness along the altitudinal gradient of 1,600–3,600 m. This was accomplished by assigning each botanical sample to one of 20 altitudinal bands corresponding to 100-m intervals from 1,600 to 3,600 m. The occurrence of each food plant was regarded as appearing at every 100-m band between its upper and lower limits, following Bhattarai et al. (2004) and Wang et al. (2007). The upper and lower limits of the altitude band were used to define the altitudinal distribution of each plant taxa. For example, *Sorbus coronata* was collected from elevations between 2,350 and 2,970 m, and therefore it is assumed to be restricted to an elevation of 2,300–3,000 m. This method, termed ‘interpolation of species ranges,’ has been employed broadly to examine plant species richness patterns along elevation gradients in the Himalayan region (Wang et al., 2007; Baniya, 2010; Acharya et al., 2011; Rai et al., 2015). We applied a Generalized Linear Model in Python (version 3.5) to calculate the potential regression equation of food species richness along each elevation gradient as well as the centralized distribution range of most food plants. The response variable, species richness, is count (discrete) data and has a bell-shaped distribution corresponding to the independent variable, 20 elevation bands of 1,600–3,600 m. The regression curve was fitted based on a normal distribution using an exponential scale. We recorded habitat information (e.g., elevation, slope, habitat type, species richness) for each plant species consumed by the captive and wild black snub-nosed monkeys as well as for all food plant species recorded at this elevational range as documented in the book the Flora of Gaoligong Mountains for counting the number of food plant species occurring in different habitat types of the Gaoligong Mountains.

This research was conducted in accordance with the guidelines of the Animal Experimentation Ethics Committee of the Australian National University for the Ethical Treatment of Nonhuman Primates (A2015/04). Procedures also adhered to the legal requirements of China, and were approved by the Forestry Department of Yunnan Province and Nujiang Prefecture Forestry Department.

6.3 Results

6.3.1 Diet of free-ranging black snub-nosed monkeys

Snub-nosed monkeys of the Luoma group were observed to consume 22 food items, from 14 plant species and ten plant families. In addition, the monkeys consumed four species of lichen from three families (Table Appendix V). Each of these food items and food species also were eaten by the captive monkeys during feeding trials. Additionally, bamboo shoots from at least one species were confirmed to be part of the wild snub-nosed monkeys natural diet based on their presence in fecal remains.

6.3.2 Diet of the captive snub-nosed monkeys

Diversity

We presented the two captive *Rhinopithecus strykeri* with a highly diverse set of food options including > 600 plant species and 30 lichen species. Of these, the captive animals consumed 170 plant species belonging to 75 genera and 41 plant families, and 15 lichen species from four genera and three families (Table Appendix V). Overall, the monkeys consumed young and mature leaves, fruits, seeds, flowers, buds, bark, petioles, pseudobulbs, and bamboo shoots from 105 tree, 21 shrub, 20 liana, 17 epiphytic, and seven herb species. This resulted in the consumption of 593 food items (species × plant parts) that included: 21.9% young leaves, 16% mature leaves, 15.7% fruits/seeds, 13.8% buds, 15.2% flowers, 7.8% young twigs, 3.7% bark, 2.5% lichen, 1.5% petioles, and 1.9% other food types such as bamboo shoots, rootstocks, and pseudobulbs (Fig. 6.2).

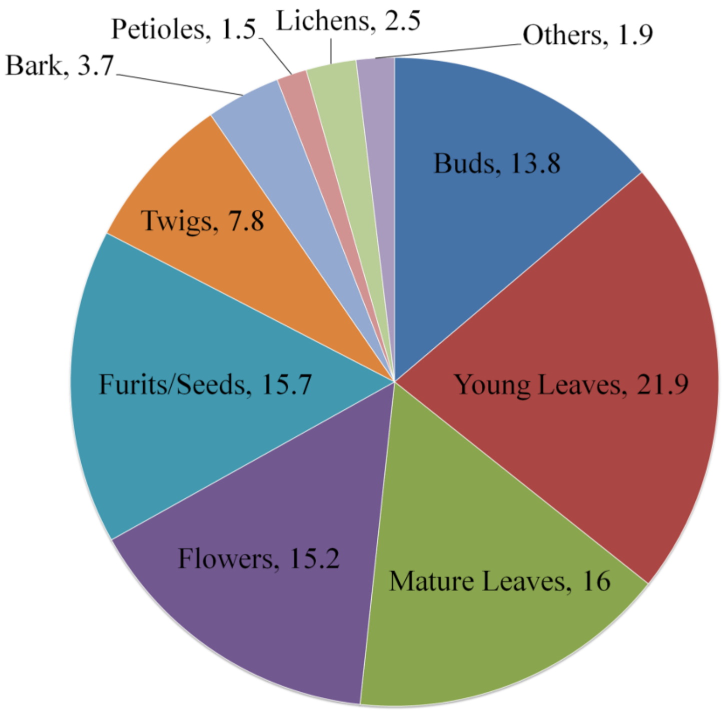

Figure 6.2 Proportions of different food types consumed by the two female captive *Rhinopithecus strykeri* (data are based on individual food items from 170 plant species and 15 lichen species)

On five occasions, we observed the captive black snub-nosed monkeys preying on arthropods (two stick insects, *Cnipsus colorantis* and *Ramulus* sp. and a katydid, *Mirollia* sp.) and on one occasion they attempted to capture a tree shrew (*Tupaia* sp.). In addition, on one occasion, small particles of soil were extracted and ingested by one of the captive individuals when dead trees were replaced in the enclosure. The captive monkeys also extracted *Carex nitidiutriculata* rootstocks from the ground and drank water from an artificial stream.

Selectivity

The captive monkeys showed evidence of dietary selectivity. Among all plant families, Rosaceae (24 species) was the taxa from which snub-nosed monkeys consumed the most species of leaves, followed by Lauraceae (13 species), Aceraceae (11 species), Araliaceae (eight species) and Betulaceae (seven species). Among the Rosaceae, the monkeys rejected buds and young leaves but consumed mature leaves of *Amygdalus persica* while, in the case of another Rosaceae species, *Prunus salicina*, they consumed the buds and young leaves and not the mature leaves. In some cases, the captive monkeys were found to discriminate between the food items of closely related species. For example, they consumed the buds, leaves, flowers, fruits, and bark of *Skimmia arborescens* (Rutaceae) but did not consume any food items of *Skimmia laureola*.

We offered 30 species of lichens to the captive monkeys, and 15 of these were consumed. Some common species of lichen in their habitat such as Cetrelia cetrarioides, Everniastrum cirrhatum, and Stica nylanderiana were not eaten by the captive monkeys.

Li et al., (2000) reported that six bamboo species existed in *R. strykeri*'s habitat at an altitude of between 1,900-3,300 m; and we obtained shoots from three of these species (*Fargesia contracta*, *F. orbiculata*, and *F. papyrifera*). The monkeys selected *Fargesia contracta* and *Fargesia orbiculata* over *Fargesia papyrifera.* This may be because the latter has an astringent taste (based on local knowledge) and their shoots are covered by thick sheaths with dense setose surfaces on both sides. Additionally, the pseudobulbs, flowers, and fruits of some epiphytic orchids were eaten by the captive monkeys, whereas the leaves and sheaths of the same species were dropped. Based on our preference criteria, the monkeys were found to prefer 134 individual food items, principally leaves and fruits from 78 plant species (Table Appendix V).

6.3.3 Distribution patterns of wild plant foods

Food plant richness in the forests occupied by black snub-nosed monkeys was best represented as a bell-shaped curve along an altitudinal gradient (Regression equation: $y = 319.290050 \times N(3.739439 \times (e^{(0.000613 \times (x - 2664.418641)) - 1})$, $R^2 = 0.981104$), with the greatest proportion of food plant species (> 50%) present at an intermediate elevation (2,215–3,016 m) and then gradually declining at lower (1,600–2,216 m) and higher elevations (3,017–3,600 m) (Fig. 6.3). Additionally, the different vegetation types contained different numbers of plant species consumed by the black snub-nosed monkeys. Food plant biodiversity was as follows (from highest to lowest): primary evergreen broadleaf forest > secondary evergreen broadleaf forest > mixed hemlock-broadleaf forest > secondary deciduous broadleaf forest > primary deciduous broadleaf forest > hemlock-bamboo thickets or alpine shrub forest (Table 6.1). These results suggest that the natural distribution of food resources exploited by black snub-nosed monkeys may constrain this species' altitudinal range, with primary evergreen broadleaf forest, secondary evergreen broadleaf forest, and mixed hemlock—broadleaf forest containing the greatest concentration of foods consumed.

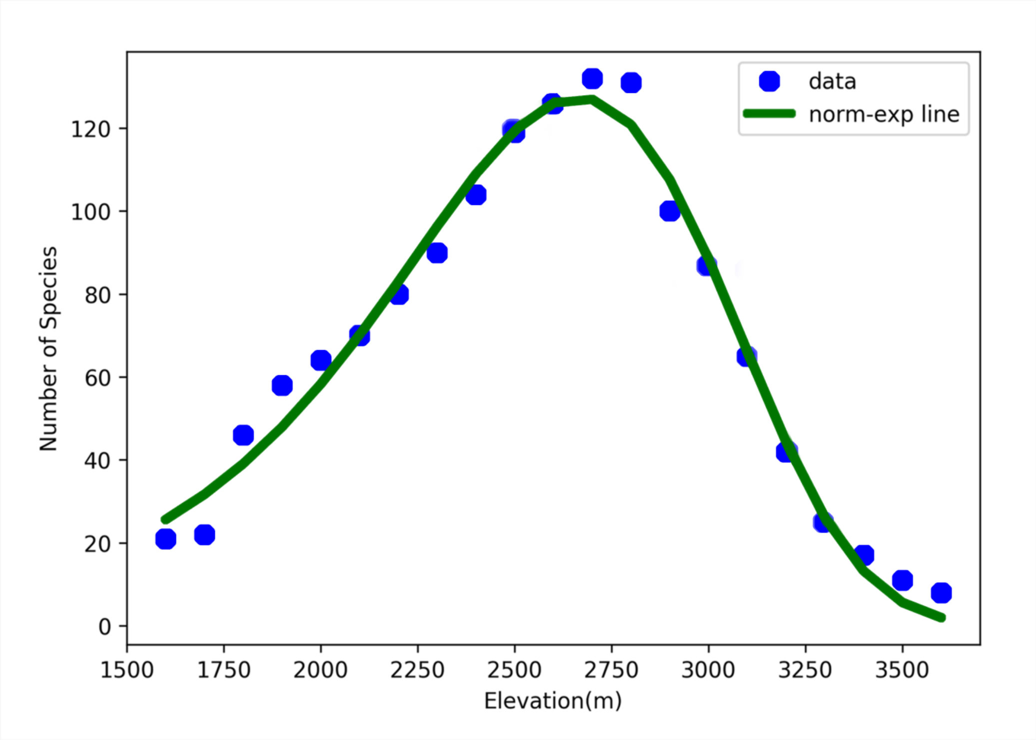

Figure 6.3 Relationship between food plant species richness (excluding lichens) and elevation (altitudinal range from 1,600 to 3,600 m).

Table 6.1 The number of food plant species occurring in different habitat types of the Gaoligong Mountains.

Habitat Type	**Number of food species**
Primary evergreen broadleaf forest	163
Secondary evergreen broadleaf forest community	127
Primary deciduous broadleaf forest	34
Degraded secondary evergreen broadleaf forest	68
Degraded secondary deciduous broadleaf forest	40
Mixed hemlock - evergreen broadleaf forest or hemlock – rhododendron forest	100
Hemlock-bamboo thickets	22
Bamboo thicket and alpine scrub	25
Note that in total we examined the distribution of 170 plant species consumed by the black snub-nosed monkey and individual species were commonly distributed in more than one habitat type.	

6.4 Discussion

In this study, we provide the first description of diet, feeding ecology, and food choice of the Critically Endangered black snub-nosed monkey, *Rhinopithecus strykeri*. Only 400 individuals are estimated to remain in the wild and therefore data on the elevational distribution and forest types of their food species are essential for developing effective conservation and management plans. Although our ability to follow a wild unhabituated population of black snub-nosed monkeys at close sighting distances was limited by the rugged mountainous terrain, based on 80 h of field observations and year-long feeding trials of the only two black snub-nosed monkeys in captivity, we found that *R. strykeri* exploits a wide range of plant species and food types, including difficult to digest plant tissues such as bark, twigs, and mature leaves. Our observations of the feeding behavior of the wild and captive black snub-nosed monkeys indicated that young leaves, fruit/seeds, mature leaves, buds, and lichen were among the most common food items consumed and that the food plant species most commonly selected were principally distributed at an elevation of between 2,200 and 3,000 m.

6.4.1 Dietary selectivity and ecological distinctions in the feeding behaviour of the genus *Rhinopithecus*

The Gaoligong Mountain Nature Reserve is characterized by higher plant biodiversity than other forested habitats that contain Chinese species of *Rhinopithecus* (Supplementary Appendix II) and is known to harbor more than 4300 plant species from over 1086 genera and 210 families, almost 50% of which are tropical in origin (Li et al., 2000). The habitat zone exploited by *R. strykeri* lies within an elevation of between 1,700 and 3,200 m (Geissmann et al., 2011) and is characterized by high plant species diversity and structurally complex canopies. Although our field observation and cafeteria-style trials provide only a partial inventory of the diet of *R. strykeri*, our results indicate that black snub-nosed monkeys consume a diverse set of plant species, and that food items from seven plant families (e.g., Rosaceae, Lauraceae, Aceraceae, Araliaceae, Betulaceae, Ericaceae, and Actinidiaceae) were consumed most frequently. As is the case for other species of snub-nosed monkeys (Bleisch, & Xie, 1998; Li, 2006; Xiang et al., 2007), these findings suggest that *R. strykeri* is a selective feeder targeting specific plant taxa across their range rather than consuming the most common tree species present in the environment. This has critical implications for black snub-nosed monkey conservation, especially because the distributional range of this species has suffered severe habitat destruction and degradation associated with human activities (Ren et al., 2017).

Based on the information currently available, all five Rhinopithecus species are reported to consume leaves, fruits, insects, buds, and bamboo (Fig. 4.4) and at least four taxa, *R. strykeri*, *R. roxellana*, *R. bieti*, and *R. brelichi* are best described as extractive foragers for foods such as tubers, bamboo shoots, and invertebrates (Ren et al., 2008; Grueter et al., 2009a; Xiang et al., 2013; Huang, 2015). For example, Meyer et al. (2017) report that, in the wild, *R. strykeri* dig to acquire bamboo shoots and search for fallen fruits on the forest floor. In addition, geophagy, which we observed in captive *R. strykeri*, also has been reported in wild but semi-provisioned *R. bieti* (Li et al., 2014) and *R. roxellana* (Huang, 2015), and in wild *R. brelichi* (Yang et al., 2002). Many species of monkeys engage in geophagy, however, in the case of colobine primates, geophagy is likely to play an important role in detoxifying plant secondary compounds and in maintaining the high pH required for their specialized stomach to support a rich and diverse microbiota (Krishnamani & Mahaney, 2000).

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Lichen is reported to account for 50–80% of feeding time in several populations of *R. bieti* inhabiting mixed deciduous broadleaf and conifer forests in the Yunling Mountains at elevation of 2,500–4,600 m (Liu et al., 2004; Kirkpatrick, 1996; Xiang et al., 2007; Grueter et al., 2009b; Li, 2010; Huang et al., 2017). Golden snub-nosed monkeys (*R. roxellana*) also consume lichen, however, depending on their habitat type, this can range from 1 to 43% of total feeding time (Li et al., 2013; Liu et al., 2013; Guo et al., 2007; Huang, 2015; Hou et al., 2018). Lichen has not been reported in the diet of *R. avunculus* or *R. brelichi* (Table Appendix VI). It appears that, compared to subtropical and tropical snub-nosed monkeys, temperate snub-nosed monkeys inhabiting high altitude forests are more dependent on lichen as a dietary staple. Given our limited number of observation hours of wild black snub-nosed monkeys, the degree to which lichens account for a large percentage of the annual or seasonal diet remains unclear. However, we have noted that on the eastern slopes of the Gaoligong Mountains, which are dominated by mixed coniferous and broadleaved trees contain greater lichen loads than the western slopes which are dominated by evergreen broadleaved trees (pers. observ.). Thus it is possible that different populations of *R. strykeri* will show differences in diet, with individuals on the eastern slopes consuming more lichen during the winter and individuals inhabiting the western slopes consuming more leaves and buds during the winter. This hypothesis requires further investigation.

In the present study, we integrated data from field observations and cafeteria-style trials to better understand the feeding ecology of the black snub-nosed monkey. We acknowledge that the food list we obtained for *R. strykeri* is incomplete, as food resources for thousands of tree species in the GLGMs were not presented to the captive black snub-nosed monkeys. Moreover, our experiment may contain a sex bias in food selection, as the only two captive individuals are female. Although these limitations exist, our results are the first to describe the diet of the Critically Endangered black snub-nosed monkey and offer an initial framework to generate and test hypotheses regarding the feeding ecology and habitat preferences of this primate species, and to compare its diet to that of other species of snub-nosed monkeys

6.4.2 Conservation implications

The data we obtained on altitudinal differences in the plant species richness, distribution, and diversity of food resources in the GLGMs closely correlated with the presumed elevational distribution of *R. strykeri* based on interview surveys (2,600–3,100 m, Ma et al. 2014), camera trap records (2,400–3,400 m, Chen et al. 2015), habitat distribution modeling (2,300–3,200 m, Ren et al. 2017), and our field observations. Today, most of the area below an elevation of 2,300 m are outside of the boundary of the GLGMNNR and are naturally dominated by dense stands of *Alnus nepalensis* (Table 4.2 and Fig. 4.4). These secondary deciduous forests are near villages inhabited by local minorities, who use these trees as an important source of firewood and construction material. In addition, much of the forest in these lower elevations has been cut for grazing domesticated animals. Thus, the habitat below 2300 meters is highly disturbed and not suitable for black snub-nosed monkeys. According to

Ma et al. (2014) and Chen et al. (2015), wild black snub-nosed monkeys rarely use these disturbed Alnus nepalensisforests. Based on the currently available information, it appears that primary mid-montane moist evergreen broadleaf forests and mixed hemlock–broadleaf forests are the most suitable habitats for *R. strykeri*, and protecting these forests should be an immediate conservation priority.

The local ethnic minorities in this area are very poor and depend on local forest resources. Thus, illegal felling of forest trees, livestock grazing, and exploitation of non-timber forest products (e.g. Yunnan goldthread *Coptis teeta*, paris *Paris* spp., orchids and mushrooms) in the altitudinal range inhabited by black snub-nosed monkeys were common during the study period. In particular, trees of high commercial value or food trees used by *R. strykeri*, such as *Acer* spp., *Dodecadenia grandiflora* var. *griffithii*, *Manglietia insignis*, *Michelia* spp., *Magnolia campbellii*, *Juniperus recurva* var. *coxii*, and *Taxus* spp., were frequently logged by local residents and sold to nearby wood factories to make luxury furniture or tea trays (Fig. 6.4 and Table 6.2), resulting in the conversion of primary forest into degraded forest and changes in floristic composition. A reduction in food availability caused by habitat degradation and fragmentation force primates to broaden their diet to include foods that may contain lower energy, higher secondary metabolites, and a nutrient imbalance (Cristóbal-Azkarate & Arroyo-Rodríguez 2007; Chaves et al. 2012). This may result in reduced fecundity and reduced infant and adult survivorship (Mbora et al. 2009), leading to population decline and increased extinction risk (Li et al. 2018). Since 2015, the Chinese government has implemented and vigorously enforced a ban on illegal trans-boundary logging of Myanmar's northern forests. However, this has resulted in increased logging pressure in the GLGMNNR. The local timber-processing industry in Pianma Township is particularly powerful, and this has led to continued illegal logging in the reserve. We suggest therefore that in order to conserve *R. strykeri* and its habitat, (1) the Nujiang Prefecture

Government should create a red list and strictly prohibit, enforce, and severely prosecute those selling these primary forest trees in the local wood markets; (2) with the help of NGOs, the reserve and the surrounding communities must work together to set buffer zones outside the protected area and begin a program of natural forest restoration that includes planting food trees and non-timber forest products that are part of the black snub-nosed monkey diet; (3) the government needs to restructure the local timber-processing industry such that native trees are banned from logging.

Figure 6.4 Large-scale commercial logging and illegal logging have threatened *Rhinopithecus strykeri*'s habitats. **a**. Illegal logging of *Taxus yunnanensis* within in the monkeys' home range in the Gaoligong National Nature Reserve. **b**. Logs of high commercial value tree species in Pianma Township. **c**. Luxury furniture and tea trays produced in Pianma Township. **d**. A spontaneous market of wild orchids and other forest products in Liuku Township.

Table 6.2 Local utilizations of forest products in *Rhinopithecus strykeri*'s habitat, Lushui County, Yunnan.

Species	Utilizations of local plants	Effects on the *R. strykeri*	Foods for *R. strykeri*
Taiwania flousiana	Logging for furniture and tea tray manufacture	Habitat fragmentation and degradation	Unknown
Cupressus duclouxiana	Logging for furniture and tea tray manufacture	Habitat fragmentation and degradation	No
Taxus chinensis	Logging for chopping board, furniture and tea tray manufacture	Habitat degradation	Potential*
Taxus yunnanensis	Logging for chopping board, furniture and tea tray manufacture	Habitat degradation	Potential
Magnolia rostrata	Traditional medicine (tree bark)	Habitat degradation; foraging pressure	Yes
Manglietia insignis	Logging for furniture manufacture	Habitat fragmentation and degradation; foraging pressure	Yes
Michelia doltsopa	Logging for woodcarving, furniture and tea tray manufacture	Habitat fragmentation and degradation; foraging pressure	Yes
Michelia Floribunda	Logging for woodcarving, furniture and tea tray manufacture	Habitat fragmentation and degradation; foraging pressure	Yes
Schisandra spp.	Traditional medicine	Frequent human disturbances	Yes
Dodecadenia grandiflora var. *griffithii*	Logging or tree burr extraction for woodcarving, furniture and tea tray manufacture	Habitat fragmentation and degradation; foraging pressure	Yes
Phoebe puwensis	Logging for woodcarving, furniture and tea tray manufacture	Habitat fragmentation and degradation; foraging pressure	Unknown
Acer spp.	Logging for construction materials and furniture manufacture	Habitat fragmentation and degradation; foraging pressure	Yes
Betula alnoides	Logging for construction materials and furniture manufacture	Habitat fragmentation and degradation; foraging pressure	Yes
Betula utilis	Logging for construction materials and firewood	Habitat fragmentation and degradation; foraging pressure	Yes
Exbucklandia populnea	Logging for wood panels and woodcarving	Habitat fragmentation and degradation; foraging pressure	Unknown
Coptis teeta	Traditional medicine	Frequent human disturbances	No
Aralia chinensis	Vegetable	Frequent human disturbances	Potential
Panax japonicus var.	Traditional medicine	Frequent human	Potential

bipinnatifidus		disturbances	
Fritillaria cirrhossa	Traditional medicine	Frequent human disturbances	Unknown
Commelina Benghalensis	Vegetable & traditional medicine	Frequent human disturbances	No
Ligusticum brachylobum	Traditional medicine	Frequent human disturbances	No
Paris spp.	Traditional medicine	Frequent human disturbances	No
Dendrobium hookerianum	Traditional medicine and gardening	Foraging pressure; frequent human disturbances	Yes
Dendrobium spp.	Traditional medicine and gardening	Frequent human disturbances	Potential
Coelogyne occultata	Traditional medicine and gardening	Foraging pressure; frequent human disturbances	Yes
Coelogyne corymbosa	Traditional medicine and gardening	Foraging pressure; frequent human disturbances	Yes
Dendrobium spp.	Traditional medicine and gardening	Frequent human disturbances	Potential

* 'Potential' means such species is a food plant comsumed by other snub-nosed species

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Chapter 7 Cafeteria-style Feeding Trials Provide New Insights into the Diet and Nutritional Strategies of the Black Snub-Nosed Monkey (*Rhinopithecus strykeri*): Implications for Conservation

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7.1 Introduction

Primates are reported to select foods that allow individuals to achieve a nutritionally balanced diet in response to the changing nutritional requirements associated with growth and maintenance, remaining thermoneutral in extremely hot or cold temperatures, and fulfilling the costs of reproduction (Felton et al., 2009; Rothman et al., 2008; Righini et al., 2017). However, in forests that are highly fragmented or selectively logged, changes in tree species composition can result in decreased food availability (Rode et al., 2006; Chaves et al., 2012), changes in the nutrient and water content of available foods, and increased social and nutritional stress (Rode et al., 2006; Irwin et al., 2015; Dunham and Rodriguez-Saona, 2018), leading to a decrease in health, fecundity, and offspring survivorship (Rode et al., 2006; Baranga et al, 2013). For example, Tana River mangabeys (*Cercocebus galeritus*) inhabiting a highly disturbed and fragmented forest were found to experience higher gastrointestinal parasite loads and reduced fertility compared to individuals inhabiting an undisturbed forest (Mbora et al., 2009). A similar pattern of decreased fertility, increased population decline, and increased parasite loads has been reported in other primate species inhabiting habitats characterized by anthropogenic disturbance (Gillespie et al., 2005).

China is home to 25 primate species, of which 80% are listed by the IUCN as Vulnerable, Endangered, or Critically Endangered (Li et al. 2018). The primate extinction crisis in China is a direct result of China's history of human expansion (currently totaling some 1.4 billion people), the rapid transformation of its natural landscape into agricultural fields, monocultures, pastures, mines, transportation networks, and vast urban centers, and an extraordinary period of economic growth over the past several decades (Pan et al. 2016; Li et al., 2018). Efforts to prevent China's impending primate extinction crisis will require an aggressive program of habitat restoration and protection, which includes the planting of native trees that provide a nutrient profile consistent with the nutritional requirements of China's remaining species of primates.

Snub-nosed monkeys (*Rhinopithecus*) represent an endangered genus of Asian colobines, with four of the five extant species native to China (Meyer et al., 2017). As in the case of other colobine monkeys, *Rhinopithecus* have an elongated, multichambered, low acid forestomach that contains a diverse microbiota required to assist in the fermentation of difficult-to-digest foods that contain high levels of structural carbohydrates and secondary compounds, such as mature leaves, twigs, tree bark, winter buds, and lichen (Chivers, 1994; Zhou et al., 2014; Hou et al., 2018). Snub-nosed monkeys inhabit a wide range of tropical and alpine forest environments ranging from 600 m to 4,600 m in elevation (Kirkpatrick and Grueter, 2010). Studies of the diet and nutritional ecology of four snub-nosed monkey species (*R. bieti*, *R. roxellana*, *R. brelichi* and *R. avunculus*) indicate that individuals prioritize nutrients while limiting their intake of plant secondary metabolites (Bleisch et al., 1998; Zhang et al., 2013; Lan Anh et al., 2014; Hou et al., 2018). For example, wild black-and-white snub-nosed monkeys (*R. bieti*) were found to consume leaves that were higher in protein and phosphorus and lower in tannins than not consumed leaves (Zhang et al., 2013). In golden snub-nosed monkeys (*R. roxellana*), the protein content of the diet in spring was higher than during the winter (Guo et al., 2018). In contrast, an almost two-fold increase in the consumption of water-soluble carbohydrates and metabolisable energy per day in winter was argued to enable these primates to remain thermoneutral during the coldest months of the year (Guo et al., 2018).

The black snub-nosed monkey (*Rhinopithecus strykeri*), also known as the Myanmar snub-nosed monkey, is Critically Endangered, with an estimated population size of less than 400 individuals remaining in the wild (Meyer et al., 2017). These primates live in the high-altitude forests (mid-montane moist evergreen broad-leaved forest and coniferous broadleaved mixed forest, 1900-3500 m a.s.l.) of the Gaoligong Mountains along the Sino-Myanmar border (Meyer et al., 2017; Yang et al., 2018). Little is known regarding *R. strykeri*'s behavior and ecology, which is due in part to the fact that this species was first discovered in 2010, and the difficulties of observing them on steep mountain slopes and across rugged forested terrain characterized by extended periods of fog and a seven-month rainy season.

In the current study, we used the results of systematic cafeteria-style feeding trials to identify the nutrient content of fresh food items collected in the home range of wild black snub-nosed monkeys that were either consumed or avoided by captive black snub-nosed monkeys. Cafeteria-style feeding trials represent an innovative tool in which wild plant foods are presented to captive animals as part of a series of food preference trials. This method has been successfully employed to study food selection, food palatability, food avoidance, and nutrient balancing in artiodactyls and rodents whose diets are difficult to study directly in the wild (Wang et al., 2015; Averill et al., 2016; Berl et al., 2017; Gasperini et al., 2018). Specifically, we collected 600 wild plant species from the natural habitat of *R. strykeri* and presented them to the only two individuals of this species in captivity. The monkeys consumed 318 leafy food items (e.g. buds, young leaves and mature leaves) from 170 plant species (105 tree, 21 shrub, 20 liana, 17 epiphytic, 7 herb species) (Yang et al., 2019). Using this technique, we examined (i) the taxonomic identity of wild native plant species readily consumed by captive *R. strykeri*; (ii) differences in the macronutrient content of foods consumed compared to those that were avoided; and (iii) whether the native leafy foods (buds, young and mature leaves) consumed by captive *R. strykeri* differed in nutrient content across seasons (spring and autumn). Finally, based on our analyses, we identified several nutrient-rich native plant species whose food items are readily consumed by the monkeys. We recommend that ex- and in-situ conservation programs act to expand the habitat available to *R. strykeri* by planting these nutrient-rich species as part of a natural reforestation program designed to connect forest fragments and to expand habitat and resource availability for this Critically Endangered primate species.

7.2 Methods

7.2.1 Study site and subjects

Mt. Gaoligong, which is home to *R. strykeri*, has a relatively mild (annual average temperature ranging from 5.4 ℃ at an elevation of 3,210 m to 21.2 ℃ at an elevation of 755 m), humid montane climate (average annual precipitation ranging from 1,012 at an elevation of 910 m to 3,904 mm at an elevation of 3,210 m) (Li et al., 2000). This mountain ecosystem supports 4,187 seed plant species from over 1,086 genera and 210 families (Li et al., 2000).

Our study subjects represented the only two black snub-nosed monkeys (a five-year-old female and a three-year-old female) in captivity. These individuals were confiscated from the pet trade by the Nujiang Forestry and Grassland Administration and were housed together at the Yaojiaping Wildlife Rescue Centre in a 210 m^2 × 7 m high enclosure designed to simulate a semi-natural environment. This rescue center is a non-exhibit facility belonging to the

Gaoligong Mountains National Nature Reserve (25°58'28.30" N, 98°42'35.84" E, elevation of 2,510 m). During the course of our study, keepers daily supplemented the cafeteria-style diet to the monkeys with one or two pears or apples, approximately 800 grams of broad beans or peas, as well as native plants (i.e *Actinidia* spp., *Ligustrum delavayanum*, *Senecio scandens*) (additional information on the two captive individuals can be found in Yang et al. 2019).

7.2.2 Cafeteria feeding trials, plant collections, and phytochemical analysis

Cafeteria-style feeding trials were conducted from November 2016 to October 2017 (see Yang et al., 2019). Plant samples were presented to the monkeys (e.g. buds, young leaves, mature leaves) either on the same day they were collected in the wild or on the following day. We also collected a voucher specimen for taxonomic identification. All voucher specimens later were identified at the Kunming Botanical Gardens (Yang et al., 2019). In order to determine the nutritional content of individual food items consumed and food items rejected, we collected plant samples from the wild that were as close as possible in appearance and ripeness to samples used in the cafeteria trials (Rothman et al., 2012). We preferentially collected samples of food items that the monkeys were observed to readily consume during our feeding trials. Samples of food items consumed and rejected were from relatively common or abundant plant species in the natural home range of black snub-nosed monkeys. Immediately upon collection, each fresh sample (≥200 grams) was placed in a large plastic sealed Ziploc bag (20 cm × 30 cm) and sample color, food type and the species' scientific name were recorded. Within 1-2 h after collection, each sample was weighed using an electronic load cell scale (precision 0. 1 g) and dried to a constant weight at 60℃ in a recirculation oven. The dried samples were then sealed into a new Ziploc bag to prevent exposure to moisture. We weighed the dried samples in the bag to calculate moisture content [$\text{Moisture\%} = (W_{Fresh}-W_{Dry}) / W_{Fresh} \times 100\%$, W = Weight]. All samples were later shipped to the Key Laboratory of Animal Nutrition and Food Science of Yunnan Province at the Yunnan Agricultural University for phytochemical analysis.

In the lab, we milled the samples and sieved the material through a 1-mm screen for analysis of fiber content and then through a 0.45-mm screen for the analysis of other nutrients. We calculated dry matter by placing the dried field samples into an oven (105 ℃) for one hour to eliminate adhesive water (Rothman et al., 2012). We analyzed all nutrition components following the Association of Official Analytical Chemists (AOAC) methods for animal feed (AOAC, 2016). Crude Protein (CP) was calculated by multiplying the nitrogen value of each plant part by 6.25 using a Kjeldahl procedure with a copper catalyst (SAC, 2009). Diethyl ether was used to extract Crude Lipids (CL) via the Soxhlet extraction method (SAC, 2003). Crude Ash (CA) content was obtained by burning 2 g of dry matter at 550˚C for three hours in a muffle furnace. Calcium and Phosphorus were assessed using the AOAC method (2016). Fiber fractions, Neutral Detergent Fiber with residual ash (NDF), Acid Detergent Fiber with residual ash (ADF), and Acid Detergent Lignin (ADL) were determined sequentially using a Tecator Swenden-M6-1020 fiber analyzer. Total non-structural carbohydrate (TNC) content was calculated by subtracting the percentage of CP, CL, CA, and NDF from the total dry mass (Rothman et al., 2012).

Huang (2014) reported that NDF digestibility by captive golden snub-nosed monkeys was 74.3% based on feeding experiments conducted on five adult males. As per Conklin-Britain et al. (2006), we used a value of 12 kJ/g of available energy/g fiber after subtracting 4 kJ/g consumed by anaerobic microbes to calculate the physiological fuel value of NDF, which was 12 × 0.743 = 9 kJ/g (Hou et al., 2018; Chen et al., 2018). We calculated the metabolisable energy (ME) of each food item per 100g using standard conversion factors (CP 17 kJ/g , CL 37 kJ/g, TNC 16 kJ/g, and NDF 9 kJ/g) (Conklin-Brittain et al., 2006) and the formula (ME kJ = 100 g × CP% × 17 kJ/g + 100 g × CL% × 37 kJ/g + 100 g × TNC% × 16 kJ/g + 100 g × NDF% × 9 kJ/g).

7.2.3 Data analysis

We used two-tailed t-tests (Ruxton and Neuhäuser, 2010) in R Statistics software (version 3.3.1, R Core Team, 2016) to compare nutrient content among different food items and different plant species. Data were square root transformed prior to analysis to conform to assumptions of normality. Specifically, we compared (1) all selected food items vs. all non-selected food items, (2) consumed young leaves vs. non-consumed young leaves, (3) consumed mature leaves vs. non-consumed mature leaves, (4) consumed buds vs. non-consumed buds, (5) consumed young leaves vs. consumed mature leaves, (6) all food items consumed in spring vs. those not consumed in spring and (7) spring food items vs. autumn food items.

In addition, "Random Forests", a data mining model with strong predictive ability was used for data classification, prediction and feature importance analysis (Breiman and Cutler, 2001). This model has comparatively high prediction accuracy, provides a tool to balance classes in unbalanced data sets, and offers an evaluation of those variables that are most important in the classification (Breiman and Cutler, 2001). In order to explore whether the nutritional characteristics of ingested vs. non-ingested plant items could be predicted based on their nutritional profile, we used the unsupervised Random Forests algorithm to test classification accuracy (Text Appendix I). To this end, we used the Random Forests model to determine the relative importance or weighted contribution of 10 nutrient components (Moisture, CP, CL, TNC, Calcium, Phosphorus, ADF, NDF, ADL, and ME) in predicting the monkeys' food preferences in R using the function Variable Importance Plot (Liaw & Wiener, 2002). All statistical tests were done in R 3.3.1 (R Core Team, 2016).

Finally, we examined the CP/NDF ratio of food items from 70 plant species presented to the monkeys. In several primate studies, the protein-to-fiber ratio of leaves has been found to be positively correlated with dietary preference (Wasserman & Chapman, 2003; Rothman et al., 2015; Matsuda et al., 2017). We designated food plant species characterized by leaves with a CP/NDF ratio above the mean value of all tested food items as important candidates for ex- and in-situ forest regeneration programs. We also identified additional nutrition indicators of importance based on the results of the Random Forests analysis. When the value of an indicator was greater than the mean for all food plant species (regardless of how much greater), we assigned a "+". The greater the number of "+'s" assigned to a plant species, the greater value we assumed that tree species had for in- and ex-situ conservation programs.

7.3 Results

We analyzed the nutritional composition of 176 wild plant samples from 108 plant species collected in the natural habitat of *R. strykeri*. The data for twenty-two items (11 consumed items and 11 non-consumed items) included one repeated sample whereas the other 132 items contained only a single sample for nutritional content analysis. We used the mean value of these repeated samples in all comparisons and analyses. Our data set included 100 individual food items (14 buds, 60 young leaves, 26 mature leaves) from 70 species (4 herb, 12 liana, 10 shrub, and 44 tree species) that were consumed by the captive monkeys, and 54 non-consumed items (9 buds, 21 young leaves, and 24 mature leaves) from 48 species (two herbs, three lianas, 8 shrubs and 35 tree species) that were rejected by the captive monkeys (Table Appendix VII, Table Appendix VIII).

7.3.1 Nutritional differences between consumed and non-consumed plant items

Moisture

In general, consumed plant tissues contained higher values of moisture than non-consumed items (t = 5.63, $p < 0.001$; Table 1), and this applied equally to young leaves (t = 3.842, $p < 0.001$) and mature leaves (t = 4.943, $p < 0.001$).

Crude Protein

Consumed plant tissues exhibited higher levels of CP (t = 3.976, $p < 0.001$) than did non-consumed items. This pattern was consistent in both young leaves (t = 2.702, p = 0.01) and mature leaves (t = 2.469, p = 0.017).

Total nonstructural carbohydrates

TNC content was significantly higher in food items consumed by captive black snub-nosed monkeys compared to food items rejected (t = 3.423, p = 0.001). The pattern remained constant across seasons. For example, food items consumed during the spring were higher in TNC than non-consumed items (t = 2.258, p = 0.027), and food items consumed in the autumn contained higher levels of TNC than non-consumed items (t = 2.164, p = 0.036).

Metabolisable Energy

The level of ME was higher in consumed items compared to non-consumed items (t = 4.514, $p < 0.001$). ME also was higher in young leaves and mature leaves that were consumed by the monkeys compared to young and mature leaves that were offered but not consumed (young leaves: t = 2.305, p = 0.029; mature leaves: t = 3.36, p = 0.002).

Crude Lipids

CL did not differ significantly between consumed and non-consumed food items (t = -0.193, p = 0.847).

Fiber

NDF was lower in consumed items than in non-consumed items (t = -5.73, $p < 0.001$), and this was the case for both young leaves (t = -2.708, p = 0.011) and mature leaves (t = -3.566, p = 0.001). In contrast, there were no significant differences in ADF and ADL between consumed and non-consumed leaves (Table Appendix IX).

Phosphorus

Phosphorus was significantly higher in consumed items than in non-consumed food items (t = 4.223, $p < 0.001$). This pattern was consistent in both young leaves (t = 3.109, p = 0.003) and mature leaves (t = 2.068, p = 0.044)

Calcium

The calcium content of consumed and non-consumed plant tissues did not differ (t = -1.332, p = 0.186). The ratio calcium:phosphorus, however, was significantly lower in consumed vs. non-consumed foods (t = 3.078, p = 0.003; Table Appendix IX).

Finally, the ratios of moisture:NDF and CP:NDF were significantly higher in consumed vs. non-consumed foods (moisture: NDF: t = 6.576, $p < 0.001$; CP:NDF: t = 6.227, $p < 0.001$).

7.3.2 Seasonal differences in the nutritional content of consumed and non-consumed food items

There was evidence of marked differences in the nutrient content of foods consumed by captive black snub-nosed monkeys across seasons. In the spring, consumed food items contained higher levels of moisture (t = 7.192, $p < 0.001$), CP (t = 4.97, $p < 0.001$), and phosphorus (t = 7.768, p < 0.001) but lower CL (t = -2.249, p = 0.03), NDF (t = -3.813, $p < 0.001$), and calcium (t = -7.779, $p < 0.001$) compared to food items consumed in autumn. In contrast, there were no significant differences in ADF, ADL, TNC and ME between foods consumed during these two seasons of the year. Given the seasonal differences reported above, food items consumed in spring were characterized by higher ratios moisture:NDF (t =

5.522, p < 0.001) and CP:NDF (*t* = 5.036, *p* < 0.001), and a lower ratio of calcium:phosphorus (*t* = -7.966, *p* < 0.001) than food items consumed in autumn.

Random Forest Analysis

Based on nutrient content, the prediction accuracy of our Random Forests algorithm of 'consumed items that were part of the diet of *R. strykeri*' and 'non-consumed food items that were rejected by *R. strykeri*' was 73.4% (Text Appendix I). Of the 100 consumed food items, only 15 items were classified in the model as non-consumed items (omission error rate: 15%). Twenty-six of the 54 non-consumed items were classified as part of the diet of *R. strykeri* (omission error rate: 48.2%). Of all 10 nutritional variables, moisture, NDF, ME, CP, Phosphorus and TNC were predicted as the most important nutritional components affecting food selection in captive *R. strykeri*. In contrast, calcium, CL, ADF, and ADL were classified to be the variables of least importance (Figure 7.1).

7.3.3 Food tree selection: implications for conservation

Based on the results of the Random Forests analysis, we selected the mean values of moisture, ME, TNC and phosphorus, along with the CP:NDF ratio as indicators of foods that are likely to be of high nutritional value to wild black snub-nosed monkeys. The mean values of CP:NDF, moisture, ME, TNC and phosphorus for food items consumed during the spring were 0.8, 80.1%, 1360.9 kJ/100g, 35.6%, and 0.46%, and these mean values for food items consumed in autumn were 0.43, 70.9%, 1318.2 kJ/100g, 32.5%, and 0.20%, respectively. Based on the results of our nutritional analysis, we identified 32 plant species (16 trees, eight lianas, five shrubs, and three herbs) that had a CP:NDF ratio higher than the mean for spring or autumn food items (at least one "+"). Among these species, we identified18 plant species (nine trees, five lianas, three shrubs, and one herb) as high priority resources (with ≥ "++++") for ex- and in-situ conservation (e.g. corridor design or habitat restoration; Table 7.1) given both their level of CP relative to NDF and palatability to the monkeys.

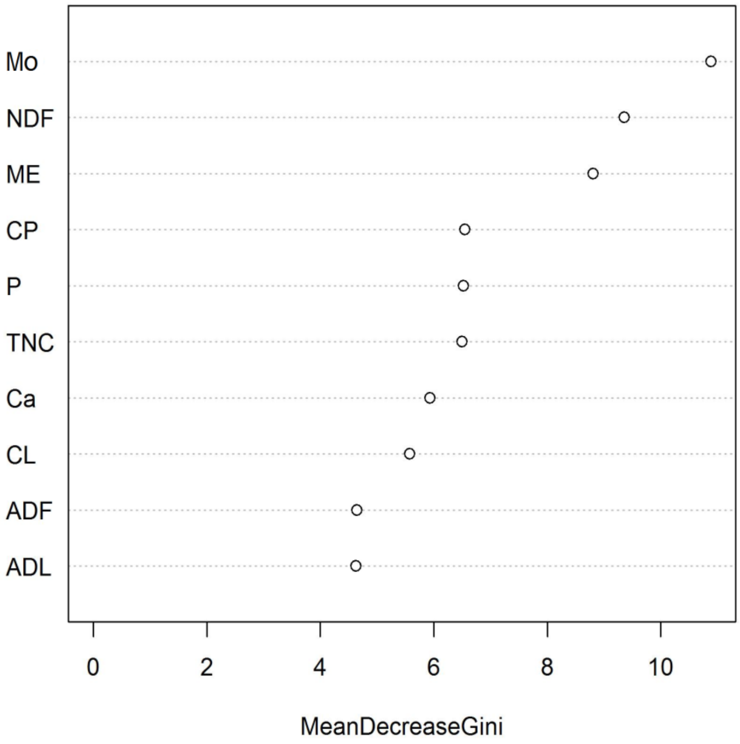

Figure 7.1 Random forest variable importance plot for predicting principal components of the 10 nutritional variables affecting food selection in captive *R. strykeri*. Values on the x-axis show the predicted importance of different nutritional components affecting food selection. Mo = Moisture, NDF = Neutral Detergent Fiber, ME = Metabolisable Energy, CP = Crude Protein, P = Phosphorus, TNC = Total Nonstructural Carbohydrates, Ca = Calcium, CL = Crude Lipids, ADF = Acid Detergent Fiber, ADL = Acid Detergent Fiber.

Table 7.1 Tree species proposed for habitat restoration or corridor establishment programs to improve the conservation of *Rhinopithecus strykeri*. Many of these trees contain food items that represent a good source of nutrients for the monkeys.

Name	Family	Food item	Nutritional indicators					Priority level	Other items also consumed by captive *R. strykeri* (Yang et al., 2019)	Type
			CP/NDF	ME	P	FW	TNC			
Acer oliverianum	Aceraceae	BUD	0.94	1,379.55	0.56	78.8	37.67	++++		Tree
		YL	1.18	1,490.32	0.26	77	60.27	+++		
Acer wardii	Aceraceae	BUD	0.83	1,429.49	0.59	82.2	30.22	++++	Fruits/seeds, young twigs	Tree
		YL	1.56	1,364.04	0.57	84.5	41.79	+++++		
Actinidia kungshanensis	Actinidiaceae	YL	0.81	1,292.94	0.53	86.9	24.75	++++	Buds, Fruits, flowers, young twigs	Liana
		ML	0.47	1,293.83	0.22	74.5	35.08	+++		
Actinidia pilosula	Actinidiaceae	YL	1.48	1,379.39	0.86	82.8	31.87	++++	Buds, Fruits, flowers, young twigs	Liana
		ML	0.59	1,441.62	0.14	65.9	44.23	+++		
Betula cylindrostachya	Betulaceae	YL	1.13	1,493.87	0.55	74.8	35.74	++++	Buds, mature leaves	Tree
Betula luminifera	Betulaceae	BUD	0.91	1,529.16	0.49	73.9	32.91	+++	Young leaves	Tree
		ML	0.54	1,409.05	0.24	65.8	32.58	++++		
Cerasus caudata	Rosaceae	YL	0.86	1,356.5	0.61	78.3	22.65	++	Buds, fruits, matures leaves, young twigs and bark	Tree
Cerasus clarofolia	Rosaceae	YL	1.02	1,395.89	0.45	78.6	46.23	+++	Buds, fruits, matures leaves, young twigs and bark	Tree
Padus napaulensis	Rosaceae	YL	0.7	1,449.55	0.14	69.4	51.9	+++	Buds, fruits, young twigs	Tree
Sorbus oligodonta	Rosaceae	ML	1.1	1,439.77	0.52	84.8	36.1	+++	Young leaves, fruits	Tree
Sorbus rhamnoides	Rosaceae	YL	0.91	1,362.45	0.55	36.01	84.8	++++	Mature leaves, fruits	Tree
Clematis clarkeana var. stenophylla	Ranunculaceae	ML	0.6	1,235.68	0.24	79.2	20.69	+++	Young leaves, flowers	Herb
Dipentodon sinicus	Dipentodontaceae	BUD	0.64	1,506.91	0.55	79.8	55.01	++++	Young leaves, mature	Tree

									leaves	
Helwingia himalaica	Cornaceae	ML	0.55	1,409.1	0.17	77.4	44.95	++++		Herb
Holboellia fargesii	Lardizabalaceae	YL	1.12	1,393.31	0.56	82	52.1	+++++	Buds, mature leaves, flowers, fruits	Liana
Hydrangea heteromalla	Hydrangeaceae	YL	1.6	1,349.06	0.54	84.7	25.55	+++	Mature leaves, flowers, young twigs	Shrub
Leycesteria gracilis	Caprifoliaceae	YL	1.19	1,447.46	0.51	76.2	48.99	++++		Shrub
Ligustrum delavayanum	Oleaceae	ML	0.43	1,288.86	0.19	64.2	29.92	+	Buds, young leaves, flowers, fruits/seeds, bark	Shrub
Lithocarpus pachyphyllus	Fagaceae	BUD	1.02	1,637.17	0.55	73	49.47	+++	Young leaves, fruits	Tree
Litsea cubeba	Lauraceae	YL	1.8	1,491.03	0.55	75.5	49.51	++++	Buds, mature leaves, flowers, fruits	Tree
Litsea rubescens	Lauraceae	YL	1.35	1,369.93	0.48	79.5	43.8	++++	Buds, flowers, fruits	Tree
		ML	0.56	1,488.26	0.23	68.4	26.18	+++		
Lonicera acuminata	Caprifoliaceae	ML	0.7	1,449.55	0.14	69.4	51.9	+++	Buds, young leaves, flowers, fruits	Liana
Pentapanax racemosus	Araliaceae	YL	0.92	1,400.15	0.62	85	45.38	+++++	Buds, mature leaves	Shrub
Polygala fallax	Polygalaceae	YL	1.23	1,479.44	2.13	82.4	38.97	+++++	Mature leaves, fruits/seeds	Shrub (Tree)
		ML	0.52	1,296.49	0.52	75.9	21.72	+++		
Populus davidiana	Salicaceae	YL	0.81	1,403.87	0.41	81.3	35.49	+++	Mature leave	Tree
Salix daliensis	Salicaceae	YL	0.88	1,343.85	0.56	83.3	24.39	+++	Buds	Tree
Sabia japonica	Menispermaceae	BUD	0.87	1,273.41	0.7	87	16.72	+++	Buds	Liana
		YL	0.89	1,377.9	0.45	82.7	45.11	++++		
Sabia Parviflora	Menispermaceae	ML	0.67	1,295.45	0.27	78	27.48	+++	Buds, young leaves, flowers, fruits	Liana
Schisandra neglecta	Schisandraceae	YL	0.87	1,413.95	0.37	83.4	48.94	+++++	Buds, flowers, fruits	Liana
		ML	0.71	1,412.37	0.27	79.6	35.43	++++		
Schisandra rubriflora	Schisandraceae	BUD	0.92	1,387.12	0.43	81	33.03	+++	Young leaves, mature leaves, flowers, fruits	Liana
Senecio scandens	Asteraceae	YL	0.9	1,336.25	0.48	86	27.53	+++	Buds, flowers, young twigs	Herb

		ML	0.58	1,219.5	0.37	80.06	11.56	+++		
Skimmia arborescens	Rutaceae	YL	1.1	1,439.77	0.52	84.8	36.01	+++++	Buds, mature leaves, flowers, fruits, young twigs and bark	Tree

Notes: ML= mature leaves, BUD = buds, YL = young leaves, FW = Free Water, CP = Crude Protein, ME = Metabolisable Energy, P = Phosphorus, NDF = Neutral Detergent Fiber, TNC = Total Nonstructural Carbohydrate

7.4 Discussion

The results of our cafeteria-style feeding trials demonstrate that captive *R. strykeri* select food items higher in water, protein, total nonstructural carbohydrates, metabolisable energy and phosphorus content and lower in neutral detergent fibre compared to food items not consumed. In addition, the Random Forest model indicated that trends in *R. strykeri*'s food choice can be predicted based on nutritional composition and that the monkeys selected items or adjusted nutrient intake based principally on the amount of moisture, NDF, ME, CP, phosphorus, and TNC in a food item. A comparison of our study with the published literature suggests that all *Rhinopithecus* species (*R. avunculus*: Lan Anh et al., 2014; *R. brelichi*: Bleisch et al., 1998; *R. bieti*: Zhang et al., 2013; *R. roxellana*: Tie, 2009; Hou, 2018), as well as several other colobines (e.g. *Colobus polykomos*, *C. guereza*, *Nasalis larvatus*, *Pygathrix nigripes*; *Presbytis rubicunda*) share a common set of nutritional strategies characterized by prioritizing the intake of water, protein, essential minerals, carbohydrates, and energy (Dasilva, 1994; Yeager et al., 1997; Fashing et al., 2007; Hanya and Bernard, 2015; Guo et al., 2018). However, the proportion of fiber intake relative to protein intake by *R. strykeri,* and other *Rhinopithecus species* differs from that reported in other folivorous primates. We discuss this in greater detail below.

7.4.1 Nutrient needs and comparisons

Folivorous primates are reported to meet most of their water requirements through the ingestion of fruits and young leaves rather than from drinking water directly from terrestrial or arboreal water sources (Cerling et al., 2004; Rothman et al., 2008). While water is important for the maintenance of homeostasis, transport of hormones and nutrients, multiple hydrolytic processes such as microbial fermentation of fibrous vegetation, and excretion of waste products also require large amounts of water (National Research Council, 2003; Dias et al., 2014). Three species of howler monkeys (*Alouatta palliata*, *A. caraya* and *A. pigra*), whose yearly diets can include from 58.2-82.4% leaves, based on feeding time, increased water intake during periods in which the consumption of mature leaves with lower levels of moisture and greater amounts of fiber and secondary compounds increased (Glander, 1978; Bicca-Marques, 1992; Dias et al., 2014). In the Tonkin snub-nosed monkey (*R. avunculus*) and in the golden snub-nosed monkey (*R. roxellana*), individuals were reported to select leaves lower in fiber and higher in moisture during all periods of the year (Tie, 2009; Lan Anh et al., 2014). Similarly, we found that the mean moisture content of foods consumed by captive *R. strykeri* was as high as 77.7%, with limited variation between food types (buds: 79.7%; young leaves: 80.2%; mature leaves: 70.9% of the same species). Additionally, we observed that captive *R. strykeri* did not consume food items that were wilted due to sun exposure, suggesting that water content may play an important role in food selection. This is supported by the fact that consumed items had a higher mean water content than non-consumed items during both spring and autumn. In addition, a high water content may indicate that selected leaves had higher cell-sap to cell-wall ratios, which are positively associated with digestibility and the availability of soluble nutrients (Oates et al., 1980; Baranga, 1983).

The protein-to-fiber ratio of a food item has been used as a general indicator of foliage nutritional quality (Rothman et al., 2015; Matsuda et al., 2017). The mean crude protein content of leaves consumed by the captive *R. strykeri* in our study was 20.7% (young leaves: 22.67% and mature leaves: 16.19%), which is within the range (15–22%) of acceptable protein levels suggested by the National Research Council for captive primates (National Research Council, 2003). The crude protein content of these mature leaves was higher than values reported for leaves consumed by *R. brelichi* in Fanjingshan in summer/autumn (CP: 12.64%, Bleisch et al., 1998) and *R. bieti* in Mt. Longma in autumn (CP: 11.01%, Huang et al., 2010). The crude protein content of young leaves consumed by the captive *R. strykeri* was, however, similar to young leaves consumed by wild *R. roxellana* in spring at two different field sites (CP: 19.97%, Hou, 2018; CP: 22.69%, Li, 2015). The mean NDF of leaves consumed by the captive *R. strykeri* in spring and in autumn in Mt. Gaoligong were 32.16% and 41.86%, respectively. These values are generally similar to those repoted for *R. brelichi* (NDF: 46% in summer/autumn, Bleisch et al., 1998) and *R. roxellana* (NDF: 32.23% in spring, Hou, 2018), but higher than those reported for *R. bieti* (NDF: 34.17% in autumn, Huang et al., 2010). The protein-to-fiber ratio of food items consumed by *R. strykeri* in our study was higher than for non-consumed items, which is consistent with previous findings for African and Asian colobines (*Procolobus badius*, *C. guereza*, Wasserman & Chapman, 2003; *Presbytis rubicunda*, Hanya and Bernard, 2015; *R. bieti*; Huang et al., 2010; and *R. roxellana*, Hou et al., 2018). However, the fiber content of leaves consumed by *R. avunculus* in Khau Ca (ADF, Lan Anh et al., 2014), *R. bieti* in Longma Mountains (NDF, Huang et al., 2010), *R. roxellana* in Shennongjia,(Crude Fiber, Liu et al., 2013; ADF, Li, 2015), *R. roxellana* in the Qingling mountains (NDF & ADF, Hou, 2018) and *R. brelichi* in Fanjingshan (NDF, ADF and lignin, Bleisch et al., 1998) did not differ from that of non-consumed leaves. Similarly, we also found that ADF and ADL of food items consumed by captive *R. strykeri* did not differ significantly from non-consumed food items. Taken together, these data suggest that the protein concentration of foliage rather than the fiber concentration may be a nutritional priority for *R. strykeri*, and that protein and fiber selection for many primate species may represent two separate goals or processes (Ganzhorn et al. 2017).

In terms of micronutrients, leaves eaten by the captive *R. strykeri* were higher in phosphorus than leaves that were avoided, a pattern similar to that documented in wild *R. bieti* (Zhang et al., 2013) and the proboscis monkey (*N. larvatus*) (Yeager et al., 1997). Phosphorus deficiency can lead to a loss of appetite, net loss of minerals in bones, and growth retardation

(Barboza et al., 2009). As the calcium content of foods is often negatively correlated with phosphorous content (Baranga, 1983, Yeager et al., 1997; Abee et al., 2012, Zhang et al., 2013; also supported in our study), captive *R. strykeri* were found to select leaves with a lower calcium-to-phosphorus ratio than that in non-consumed leaves.

Our results provide evidence of seasonal differences in food choice by captive black snub-nosed monkeys. Compared with consumed food items in spring, autumn food items selected by captive *R. strykeri* were higher in lipids, total non-structural carbohydrates and metabolisable energy. In the temperate forests inhabited by black snub-nosed monkeys, many trees are deciduous and lose their leaves in late autumn and winter. However, the leaves of evergreen trees such as *Michelia doltsopa*, *Skimmia arborescens*, and *Machilus rufipes* are available throughout the year. Compared to non-consumed evergreen leaves, consumed evergreen leaves were slightly higher in carbohydrates (37.6% vs. 29.9%) and metabolisable energy (1340 kj/100g vs. 1265 kj/100g). Wild *R. strykeri*, similar to *R. avunculus* and *R. brelichi*, may switch to evergreen leaves when other resources are less available (Dong, 2012; Xiang et al., 2012). This dietary adjustment is similar to that reported for other *Rhinopithecus* species that exploit foods high in carbohydrates and energy (e.g. evergreen leaves, lichen, bark, seeds, and buds) during the cold winter months (*R. bieti*: Grueter et al. 2009; *R. roxellena*: Guo et al. 2007; Liu et al. 2013, Hou et al. 2018). In the case of *R. roxellena,* increased energy consumption during the winter has been shown to offset the increased costs needed to remain thermoneutral when winter temperatures drop below 0 ˚C (Guo et al., 2018).

7.4.2 Comparison of cafeteria feeding trails and other methods

Results of our Random Forests modeling suggest that captive *R. strykeri* selected leaves based on their nutritional and chemical compositions. In addition, the higher classification error rate for non-consumed leaves than consumed leaves may reflect the fact that we were unable to measure antifeedants (e.g. condensed tannins, phenolics, and silicates) in our plant samples and it is likely that the presence/proportion of plant secondary compounds affected food choice in black snub-nosed monkeys (Simmen et al., 2003; Lan Anh et al., 2014). Given differences in the nutritional composition of consumed and non-consumed food items, our results indicate that captive *R. strykeri* followed a general pattern of nutrient consumption that is consistent with other colobine species in terms of consuming foods that are high in water, protein, neutral detergent fiber, total nonstructural carbohydrates, metabolisable energy, and phosphorus. This highlights the importance of phylogeny and foregut fermentation in understanding primate nutritional ecology.

Although this study is based on the food choices of only two *R. strykeri*, our results represent a first-step in understanding the nutritional ecology of this Critically Endangered primate. Moreover, our findings suggest that cafeteria-style feeding trials can represent a reliable and complementary method for studying the feeding habits of primates whose diets are difficult to accurately document in the wild (see Table Appendix X). However, we note that each of the two captive *R. strykeri* lived in the wild for less than one year, and therefore may have been unfamiliar with several natural foods that are consumed by wild individuals (Whiten & van de Waal, 2018). In addition, although we conducted feeding trials and nutritional analyses of 108 plant species, other plant species in the GLGMs were not tested. Moreover, our study focused on only two seasons of the year, spring and autumn, and therefore it remains unclear the degree to which the results presented here also characterize patterns of food choice and nutrient intake in *R. strykeri* during the summer and winter. Finally, field observations are needed on wild populations of black snub-nosed monkeys to confirm the results of our study and to augment our understanding of their feeding ecology.

7.4.3 Recommendations for in and ex situ conservation

A key first step for habitat restoration and conservation planning is determining which native tree species best fit the dietary and nutritional needs of the local animal community (Sakamaki et al., 2011; Oliveira et al., 2011). Currently, most of *R. strykeri*'s remaining habitat is located outside of protected areas where large-scale deforestation has led to forest fragmentation and population isolation, which makes habitat restoration critically important (Ren et al., 2017). Our results offer the first systematic evidence for the importance of particular plant species and food items in the diet of black snub-nosed monkeys. We argue that conservation planning to save this Critically Endangered primate species should include these plant species in reforestation and habitat restoration projects.

Finally, identifying nutritionally appropriate diets is critical in maintaining healthy populations of captive primates (Janssen, 1994; Schwitzer et al., 2006). Currently, the foods offered to the captive *R. strykeri* in the Gaoligong Wildlife Rescue Centre follow the high sugar-low fiber diet (beans, apples or pears, vegetables with limited access to natural foods) commonly fed to colobines (i.e. *Pygathrix nemaeus*, *Nasalis larvatus*, *R. roxellana*, *R. brelichi* and *R. bieti*) kept in zoos (Schwitzer et al., 2006; Matsuda et al., 2018; Chen et al., 2018; Hale et al., 2019; Yin Yang Pers. Obs.). This unbalanced diet may contribute to gastrointestinal disorders because foods high sugar and/or high starch, increase fermentation rates and the occurrence of tympanites, resulting in the rapid production of volatile fatty acids that can decrease the pH of their multichambered forestomach and kill the microbiome required for fermentation (Sutherland-Smith et al., 1998; Clayton et al., 2018). Loss of dietary diversity likely decreases the abundance of microbiota associated with butyrate production, which is critical to maintian colonocyte health and immune defense (Hale et al., 2019). Therefore, we recommend that the diet of captive black snub-nosed monkeys should include diverse food plants that offer a nutritional profile similar to that of foods present in their natural habitat. This change may serve to increase the fiber content of their diet and improve the survivability of captive individuals of this Critically Endangered primate species.

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Chapter 8 Habitat Evaluation and Conservation Framework of the Newly Discovered and Critically Endangered Black Snub-Nosed Monkey

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8. 1 Introduction

The recently described black snub-nosed monkey (*Rhinopithecus strykeri*), alternatively known as the Myanmar or Nujiang snub-nosed monkey, is found in the high altitude forests of north-eastern Kachin state, Myanmar (Geissmann et al., 2011) as well as in the middle segment of the Gaoligong Mountains, Yunnan, China (Long et al., 2012; Yang et al., 2018). There are up to 14 sub-populations (10 in China and 4 in Myanmar, < 950 individuals) of black snub-nosed monkeys living in the northern Sino-Myanmar border area according to interviews (Geissmann et al., 2011; Ma et al., 2014). As a slow breeding species, combined with the prevalence of hunting and the extensive loss of habitat, this species is likely experiencing a rapid demographic decline (Geissmann et al., 2011). With the booming economy and population growth of the region, deforestation for agricultural cultivation and infrastructure development pose potential perils to this IUCN-Critically Endangered primate.

To save this species from the brink of extinction, it is important to study its habitat and its distribution, but, due to the extreme ruggedness of the terrain and the long wet season, almost no information on population distribution and habitat status are available for conservation planning. Where the data are poor in given study areas, employing MAXENT can offer reliable assessments for a species' possible distribution, and can support conservation planning (Thorn et al., 2009; Peck et al., 2011; Ingberman et al., 2016) by prioritizing appropriate habitats for new protected areas (Urbina-Cardona & Loyola, 2008; Campos & Jack, 2013) or guiding prospective land use planning and management (Illera et al., 2010), by identifying least-cost corridors for habitat connectivity (Liu et al., 2016; Luo et al., 2016; Schaffer-Smith et al., 2016), and by pinpointing ideal reintroduction sites (Thorn et al., 2009; Cilliers et al., 2013). The function of MAXENT is to generate a model across one study area based on existing information concerning environmental variables which is then used in another area to make predictions and inference of maximum entropy occurrence under similar environmental constraints (Phillips & Dudík, 2008; Thorn et al., 2009; Araújo & Peterson, 2012). Compared to other ecological niche algorithms, MAXENT as a present-only model can moderately offset for imperfect, limited species occurrence data sets and reach near-maximum accuracy levels under these circumstances (Hernandez et al., 2006, 2008; Giovanelli et al., 2010).

Based on climatic variables and two sets of locality records obtained by interview-based survey (Gessiman et al. 2011; Ma et al. 2014; Fig. 8.1), a MAXENT model was built up to map the habitat of *R. strykeri*. Forest cover maps in the 2000s and 2015, based on Landsat images, were used to mask out non-woodland areas from the habitat. We then assessed habitat changes from 2000 to 2015. According to the results, we identified additional areas where *R. strykeri* may occur, proposed specific conservation priorities, and determined crucial areas in which urgent protection measures should be taken to ensure the long survival of this rare and little known primate.

8. 2 Methods

8.2.1 Study area

The Gaoligong Mountains rise from the low altitude (183 m) drainage of the N'mai River in Myanmar to an altitude of 6,318 m in the southeast of Zayü County (Chaplin, 2005). The altitudinal gradient yields a vertical zonation of climate, soil composition and solar radiation, generating diverse vegetation types and a rich flora and fauna diversity (Chaplin, 2005). The vegetation in the Gaoligong Mountains, from the valleys to the peaks, goes from tropical monsoon forest (< 1,000 m a.s.l.), monsoon evergreen broad-leaved forest (1,100-1,800 m), semi moist evergreen broad-leaved forest (1,700-2,500 m) and mid-montane moist evergreen broad-leaved forest above (1,900-2,800 m), coniferous broadleaved mixed forest (2,700-3,500 m), to the alpine bush zone (> 3,400 m) (Li et al., 2000).

According to Geissmann et al. (2011) and Ma et al. (2014), the known and potential *R. strykeri* populations are restricted to a narrow range of 98°20'-98°49'E, 25°58'-26°31'N. Two insurmountable natural barriers, the N 'Mai Hka and Salween Rivers, would appear to have physically blocked their expansion to other areas. Thus, the study area for this project ranged from the China State Road 320 (named Road NH3 in Myanmar) in the south to the provincial border between Yunnan and the Tibet Autonomous Region in the north, and from the N' Mai Hka River in the west to the Salween River in the east, with a total area of about 41,350 km^2 (97.05-98.91E, 24.02-28.40N, Fig 8.1).

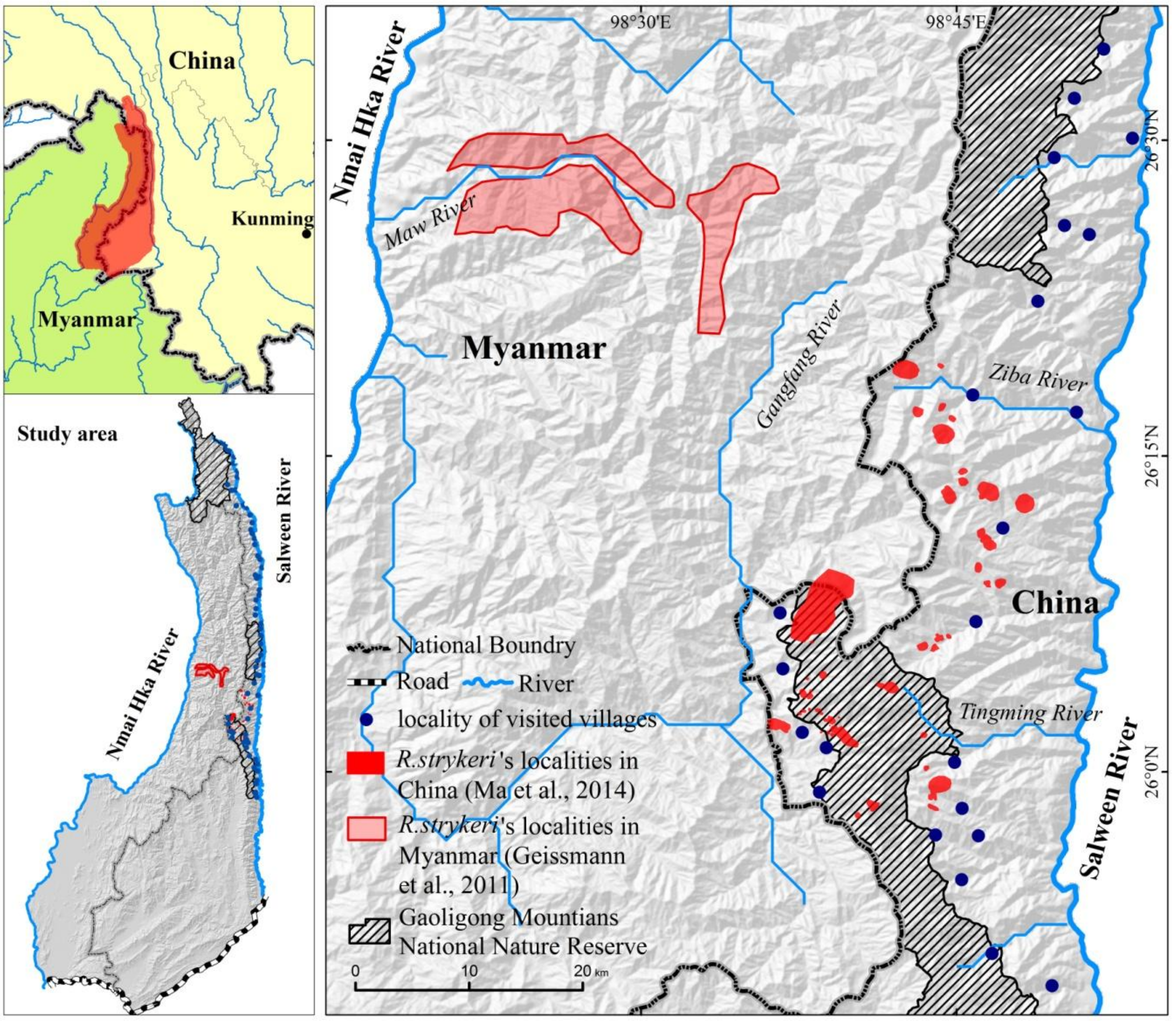

Figure 8.1 Map of the study area and distribution of *Rhinopithecus strykeri* in the Sino-Myanmar border region.

8.2.2 Species occurrence extraction

Interview-based survey can cost-effectively and statistically access the number and distribution of rare and poorly known large animals, especially in the case of primates with a distinctive appearance such as the black snub-nosed monkey (Meijaard et al., 2011; Cui et al., 2015; Ma et al. 2015; Turvey et al. 2015). Between April 2011 and December 2012, our research mission visited 68 villages near the high-altitude forests and interviewed 358 old hunters or mountain villagers who entered the forests frequently in Gongshan, Fugong and

Lushui counties, on the west bank of the Salween River in China (Ma et al., 2014). As a result, a total of 72 valid distributional records was obtained from 67 respondents, and two sub-populations of black snub nosed monkeys then were confirmed in Pianma (Ma et al. 2014) and Luoma (Yang et al., 2018). These 72 records, which were further merged into 46 polygons with sizes ranging from 16 to 1,727 ha, were used as species occurrence data in China since *R. strykeri* is not easily mistaken for any other primate given its peculiar appearance (Fig. 8.1). Three possible distribution records reported by Geissmann et al. (2011) in the Maw River areas of Myanmar, with areas of 3,854 ha, 4,126 ha, and 6,108 ha, were also included in the species occurrence data.

8.2.3 Data acquisition

Data on both climate and vegetation were used to predict the distribution of the black snub-nosed monkey. Thirty two Landsat ETM+/OLI images were downloaded from USGS Earth Explorer (http://earthexplorer.usgs.gov/) to map forest cover over the study area (see Table Appendix XI). Climatic data which we used include monthly maximum, minimum and mean temperature, monthly precipitation, and 19 variables (see Table Appendix XII). All these climatic data were acquired from the WorldClim (http://www.worldclim.org, see Hijmans et al. 2005). As with the Landsat images, the WorldClim (30 arcsec spatial resolution, equal to 0.00833°) were transformed to a Universal Transverse Mercator coordinate system (UTM, WGS84 datum, Zone 47 N) and spatially interpolated to 30 m resolution using bilinear interpolation.

8.2.4 Forest mapping

510 sets of 30 × 30 m squares (9 squares per set) were systematically sampled as “ground-truth data” from the study area by visual interpretation of high resolution images on Google

Earth (Google Inc, Table 8.1, Figure Appendix I). Each square was deemed woodland if the tree cover was higher than 40%; otherwise, it was determined as non-woodland. The whole project area was divided into woodland area and non-woodland area, based on Landsat TM/ETM+/OLI images (resolution = 30 m) using a random forest algorithm with the "ground-truth data" (Breiman, 2001). The classification process was implemented using R Statistics software (http://www.r-project.org/) with the Random Forest package (Liaw & Wiener, 2002) and Raster package (Hijmans, 2016).

Table 8.1. Hierarchical definition of forests.

Name	Definition
Fs: forest + shurb	Tree crown cover degree > 40%，including forest and shrub.
Fw: forest = Fc + Fo	Tree crown height >= 3m
Fc: closed-forest	Tree crown cover degree >= 70%
Fo: open-forest	40% =< Tree crown cover degree < 70%
Sh: shrub	Plant cover degree >= 40%, but not fulfilling the other conditions of forests

8.2.5 Species distribution modelling

MAXENT is a popular species distribution model, with a history of superior performance (Boubli & Lima, 2009; Thorn et al., 2009; Peck et al., 2011; Chetan et al., 2014; Hernandez et al. 2006; 2008). Species occurrence sites and environmental data are required to build a MAXENT model. At first, 10,000 pixels (30 m resolution) were randomly selected from the study area as background data (pseudo-absence data), 2,000 samples of which were used as pseudo-absence data for the MAXENT model, and the remaining 8,000 samples were used for validation. Two thirds of the species occurrence records, 33 of 49 distribution polygons, were randomly selected as a training polygon, and the remaining 16 polygons were used as test polygons. All the 49 distribution polygons were aggregated and rasterized at 30 m resolution. No > 100 pixels or 50% of the total pixels were systematically sampled from each of the 49 distribution polygons. In total, 2,772 samples were obtained from the 33 training polygons, and 1,008 samples were obtained from the test polygons as test data.

A MAXENT model was built with the 2,772 presence samples and 2,000 background samples based on the 67 climate variables with the default parameters setting in the R package dismo (Hijmans et al. 2016) (Text Appendix II). As a result, we obtained a climatic suitability layer of the study area. Four levels of climatic suitability were quantified according to relevant parameters of the MAXENT model for identifying core, medium, low quality and none habitat (Table 8.2).

Table 8.2 Habitat quality levels description of black snub-nosed monkeys.

Quality	Predicted value	Description
Core	> 0.50	Equal training sensitivity and specificity (0.50)
Medium	0.41-0.50	Maximum training sensitivity plus specificity (0.41)
Edge	0.04-0.41	Minimum training presence data omission (0.04)
Non-habitat	< 0.04	Less than 0.1% of training presence data predicted (0.04)

The black snub-nosed monkey is arboreal and inhabits mainly subtropical broad-leafed forest and mixed temperate forest. Forests provide shelter and ample food for them; non-woodland areas could, therefore, be regarded as non-habitat. We obtained habitat maps in both stages (2000s and 2015s) by masking out non-woodland areas from the climatic suitability layer using forest cover maps. Habitat changing from 2000 to 2015 was estimated by simply comparison of the habitat maps in 2000 and 2015 (Figure Appendix II).

The AUC (area under curve) of ROC (receiver operating characteristic curves) was employed to measure the accuracies of the habitat maps. The AUC values were 0.964 and 0.957, which indicated that our habitat evaluation models were accurate (Araújo et al., 2005)

All the above procedures were performed using R Statistics software (version 3. 3. 1, R Core Team, 2016) with some key packages, such as raster (Hijmans, 2016) and dismo (Hijmans et al. 2016).

8.3 Results

8.3.1 Habitat distribution for the black snub-nosed monkey

Based on results generated from the hierarchical model, the current geographical distribution for *R. strykeri* is predicted to be in the range of E98°20'-98°50', N25°40'- 26°50', with a total area of approximately 3,575 km^2, of which the core habitat is 1,420 km^2, medium quality habitat is 750 km^2, and low quality habitats are 1,405 km^2 (Fig 8.2). Among these suitable habitats, 2,444 km^2 (68.4%) are in Myanmar, and 1,131 km^2 (31.6%) in China. Only 10.9% of the total habitat is located in existing protected areas of China. The largest patch of core habitat harbouring most *R. strykeri* populations covered 1,280 km^2 and crossed the China-Myanmar border within a range of 50 km, including 450 km^2 (35.2%) in China and 830 km^2 (64.8%) in Myanmar. The vertical range of core habitat is estimated to be 2,330 to 3,240 m in both countries.

8.3.2 Habitat alterations in the past fifteen years

In 2000 there was an estimated 3,670 km^2 of habitat in the Sino-Myanmar border region, but by 2015 this figure had declined by 95 km^2 (a destruction of 2.6% over time or a mean of 0.17% yr^{-1}). Correspondingly, the core, medium and low quality habitats decreased by 50, 20, 25 km^2. Of the total habitat loss, 96% was in northern Myanmar. The loss of habitat within Myanmar from 2000 to 2015 was estimated to be 97 km^2, or 3.8% of the entire habitat of the species in Myanmar for the year 2000 (Fig. 8.3), while in China, forest cover loss (4 km^2)

was lower than the amount of forest rehabilitation (6 km^2); the destruction occurred mostly outside the Gaoligong Mountains National Nature Reserve.

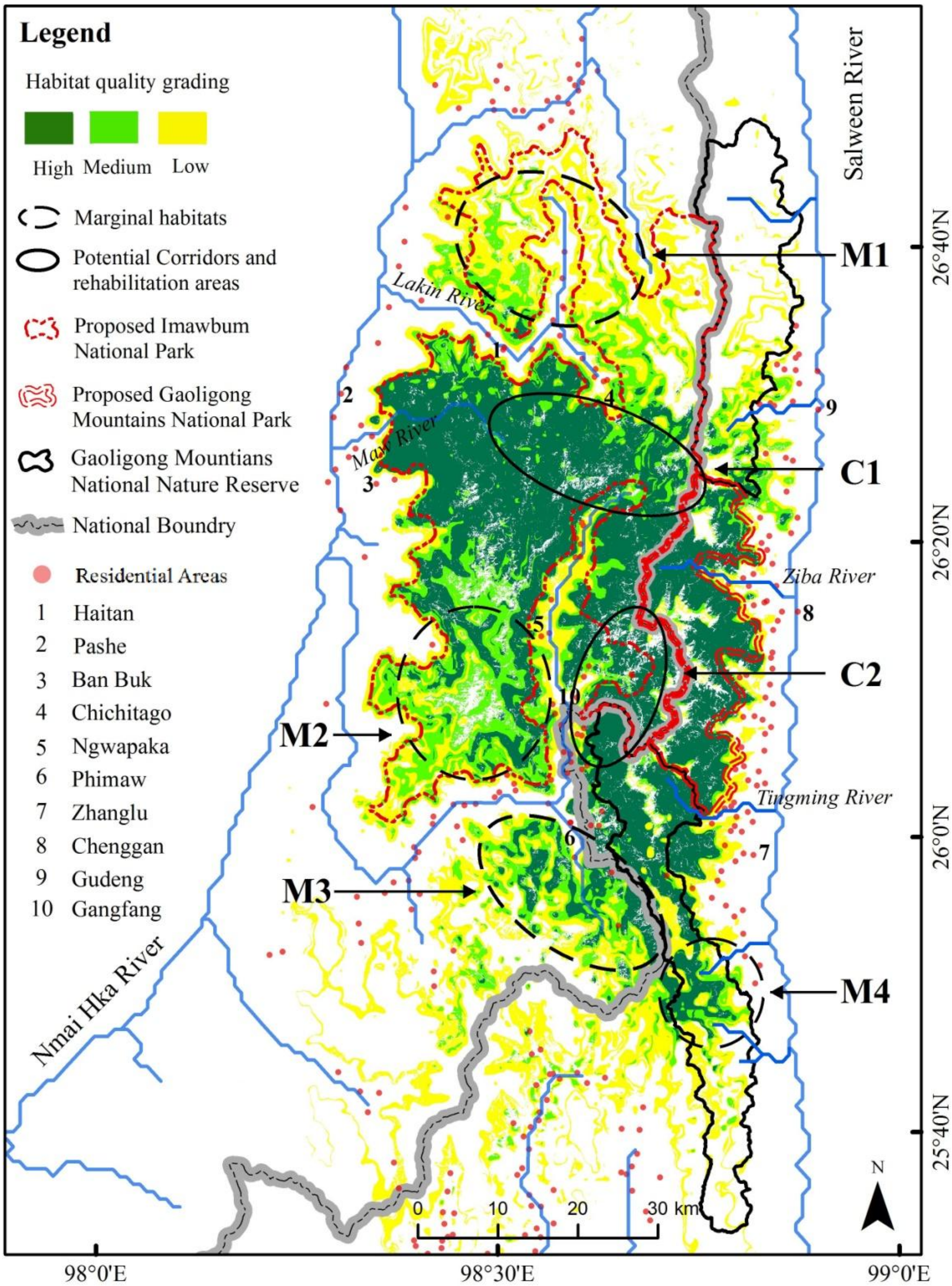

Figure 8.2 Predictive habitat distribution and quality map with proposed conservation and management areas for *Rhinopithecus strykeri* in the Sino-Myanmar border region according to climate niche analysis, forest cover in 2015 and MAXENT.

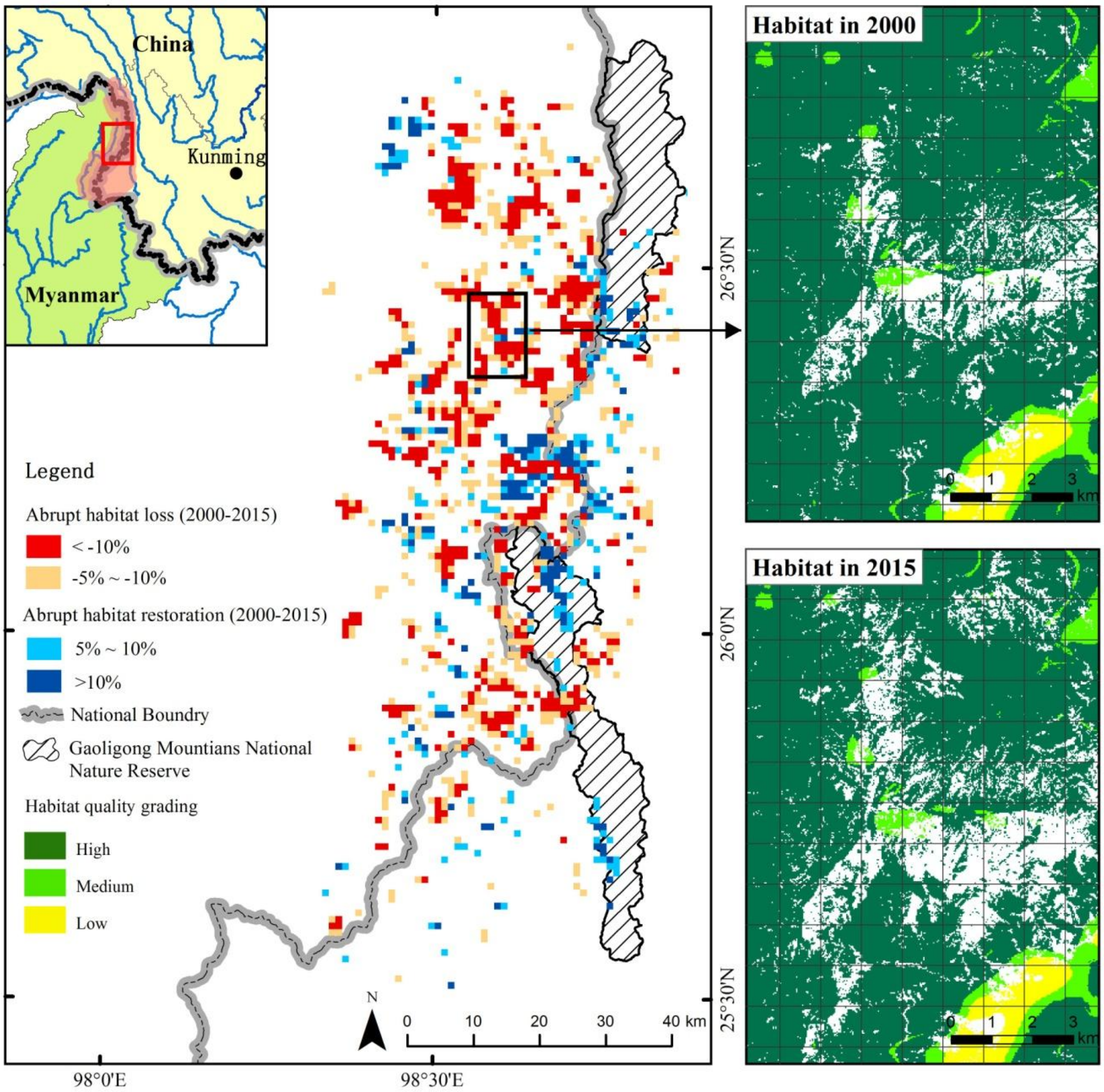

Figure 8.3 Habitat alterations (2000–2015) in the Sino-Myanmar border region, measured as the percentage of habitat cover change within 1 × 1 km^2 grid cells. The locations of habitat loss are marked with red colours and rehabilitation with blue colours. The right-hand images show an example of habitat loss, fragmentation and insularization in the study area during 2000 and 2015.

8.4. Discussion

8.4.1 Urgent conservation needs and applications

In general, the entire habitat area was reduced between 2000 and 2015, although the overall speed of loss and recovery was irregular (Fig. 8.3). In Myanmar, *R. strykeri* faces intense pressure from habitat loss and hunting, as there are no conservation measures to date. Habitat loss is due not only to shifting cultivation near human settlements in the valley, but also extends along criss-crossing roads because of commercial logging, dam construction and mining into the central core habitat area, and even peaks (Momberg et al., 2010; Geissmann et al., 2012), which may have affected population continuity because of habitat fragmentation and disruption of habitat connectivity (see Fig. 8.3, right images). In addition, hunting of this species is still very severe, even after the species' discovery in 2010 (Meyer et al., 2015); combined with habitat loss it is projected that this may lead to a species decline of > 80% in the next fifteen years in Myanmar (Geissmann et al., 2012). In China, more than half of the habitat (753 km^2, 66.6%) is located outside the reserve and there has been depletion of forest resources caused by human population growth. Occasionally hunting for bush meat and traditional medicine has provided further pressure (Ma et al., 2014; Figure Appendix III. F). Considering all these factors, we propose the three following major conservation procedures:

First, conservation gaps should be filled by the establishment of new protected areas (Fig.8. 2). The habitat of *R. strykeri* slightly recovered in the Gaoligong Mountains National Nature Reserve, and this indicates that the creation of new protected areas might be an efficient method to maintain the habitat in this region. Long et al. (2012) recommended Sino-Myanmar cooperation in delineating a trans-boundary wildlife sanctuary to protect this species. Support by the relevant institutions, the Myanmar Ministry of Environmental Conservation and Forestry and the National Democratic Army of Kachin (a pro-government militia), has already led to agreement to create a protected area, named Imawbum National Park, in Sawlaw Township (pers. comm. Fauna & Flora International, 2016). We propose that the range of Imawbum National Park should border the Gaoligong Mountains National Nature Reserve in China (see Fig. 8.2), which covers an area of 1,934 km^2. At present, the proclamation of new national parks in China is proceeding rapidly. The Program for National Park Development in Yunnan Province 2009-2020 under the Yunnan Provincial Government proposed to create a national park in Nujiang Autonomous Prefecture. We hence suggest setting up the range of Gaoligong Mountains National Park (328 km^2) to link the southern and central segments of the Giaoligong Mountains National Nature Reserve in Lushui County (Fig. 8.2).

Second, management plans need to consider the linkages between the protected areas on both sides of the border and the importance of maintaining habitat corridors that presently connect them. Based on the habitat distribution pattern, two wildlife corridors with management measures are proposed here to contribute habitat connection and connectivity maintenance. In particular, the most important trans-boundary habitat corridor should be located in the northern edge of the fragmented habitats to connect the two largest core habitats in Myanmar (Fig. 8.2, black circled area C1) in order to enable population dispersal and gene flow and increase the probability of the species' persistence. Timber extraction, shifting cultivation and road construction are imminent threats to this corridor. Any further deforestation should be prohibited there, and the lost habitats must be speedily restored by natural rehabilitation. We subsequently propose that the remaining vegetation in Myanmar must be strictly protected where the area borders the habitat at Gangfang Village of Pianma Township in China (Fig. 8.2, black circled area C2). The habitat there was dramatically disjointed by commercial harvesting, shifting cultivation, road construction and mining; a small cross-border population, however, may inhabit this general area (Ma et al., 2014; a newly dead adult female specimen was collected from a hunter in Gangfang of Myanmar in November, 2015). Natural recovery assisted by artificial restoration of trees in this area will quickly restore a habitat corridor on the Sino-Myanmar border and increase the chances for survival for the threatened population.

Third, we urgently recommend a Sino-Myanmar joint-patrol law enforcement team for implementation of the trans-boundary zoning and management plan to combat hunting and illegal trans-boundary logging activities. Thus, we can ensure that effective National Park management will be in place and that both forest departments can enforce the law against hunting and illegal logging in the border area, and to avoid a “paper park” becoming established like many others in Myanmar (see Rao et al., 2002; Aung, 2007). As the Sino-Myanmar border is a relatively poor and backward remote area, hunting wildlife has become a source of protein supplementation and income for local minorities; it is therefore important to China and Myanmar to set up an intergovernmental investment fund for payments for ecosystem services, developing sustainable alternative livelihoods, and community co-management projects, improving financial support to the protected areas for their effective operation and capacity building, and enhancing environmental education for local people and park staffs.

As a supplementary note we also suggest that the distribution maps produced here should be superimposed with existing and future land-use maps for both the development departments of China and Myanmar to prevent future damage of the stretches of habitat that are key for the conservation of *R. strykeri*.

8.4.2 Determining new investigation areas

Another contribution of our study is to determine where the monkeys may occur for future field survey. Further population investigations should be conducted in the core habitats which have not been investigated and monitored. According to MAXENT, for example, field census or interview based surveys should be conducted in the core habitat between the proposed corridors of C1 and C2 (Fig. 8.2) which has a great potential to sustain some *R. strykeri* populations. Besides, the peripheral population in marginal habitats are probably more subject to habitat loss and more vulnerable to extinction before conservationists can find them. Accordingly, the northernmost habitat (Fig. 8.2, black circled area NO. M1), the Imawbum Mountain (black circled area NO. M2), and the southernmost habitat (black circled area NO. M4) need immediate field censuses to confirm the potential existence of peripheral populations.

8.4.3 Approach caveats

There probably are some prediction errors resulting from insufficient information on habitat. The elevational range of the core habitats is strongly associated with the mid-montane moist evergreen broad-leaved forest and coniferous broadleaved mixed forest (Li et al., 2000). We found that *R. strykeri* prefers the abovementioned forest types but avoids secondary deciduous broad-leaved forest, secondary bamboo forest, artificial pine forest and coppice lands. These non-preferred forests have mostly been degraded and fragmented by large scale logging, road creation and burning or Lisu and Law Waw peoples' forest-crop rotation on these lands (eg. Fig. 8.2, black circled area NO. M3). We did not consider any of these forests in this study due to the limitation of ground-truth data for forest mapping. Additionally, the woodland being used in forest mapping includes shrub land, thus contains some non-habitats for the arboreal *R. strykeri*. Therefore, the entire habitat range could eventually be overrated and the habitat loss rate underestimated. A fine-scale vegetation type map of the distribution of *R. strykeri*, where possible, should be considered. Moreover, very informative materials are also essential to catch the typically patchy distribution of intense anthropogenic impacts, which in our study case include dam construction, mining sites, road opening and human settlements.

8.5 Conclusion

As the black snub-nosed monkey is endemic to the junction of two biodiversity hotspot regions, Indo-Burma and Mountains of Southwest China (Marchese, 2015), we believe that, to protect this Critically Endangered species, all of its actual and potential distribution area should be strictly protected by banning hunting, logging, agricultural reclamation and other improper activities. As an umbrella and global flagship species, the effective conservation of *R. strykeri* will also protect other sympatrically threatened biota (see Table Appendix XIII). Results from this work highlighted that strengthening current conservation practices should ensure the long survival of *R. strykeri* in China while establishment of cross-border conservation and the rapid elimination of the threat in Myanmar are necessary for re-emergence of this species in that country and across the border. The conservation framework proposed in this paper therefore can serve as a basic prop for conservation department to formulate more specific strategies to protect *R. strykeri*.

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Chapter 9 Identifying Transboundary Conservation Priorities in a Biodiversity Hotspot of China and Myanmar: Implications for Data Poor Mountainous Regions

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9.1 Introduction

Over the past 20 years, biodiversity conservation priorities have largely focused on species-based approaches at large spatial scales rather than on more targeted and smaller-scale analyses at the local level to determine the specific sites where species or habitat protection is required (Brooks et al., 2006; Hoffmann et al., 2008; Pimm et al., 2018). This may reflect the fact that many conservation assessments rely heavily on species occurrence data from open source databases (i.e. IUCN Red List of Threatened Species and Birdlife International) to inform the location of priority conservation areas (Beresford et al., 2011; Dorji et al., 2018; Pimm et al., 2018; Santarem et al., 2019). However, in many cases, these distributional databases contain geographic biases, are only updated every 5-10 years, and provide very general information on species distributions, limiting their value in developing fine-scale conservation priorities (Rondinini et al., 2006; Hoffmann et al., 2008; Di Minin and Moilanen., 2014; Brooks et al., 2019). For proper zoning and management planning at smaller scales, it is essential to have dimensionally accurate and high-resolution information of the geographical range of target species. This is especially challenging for species inhabiting mountainous or steeply sloped terrain, or for rare or cryptic species (Turvey et al., 2015; Ren et al., 2017).

An effective approach for obtaining cost-efficient and relatively accurate data in areas characterized by poor accessibility uses local ecological knowledge (LEK) to assess the distribution and conservation status of rare species (Ma et al., 2014, 2015; Turvey et al., 2015, 2017; Cui et al., 2016). These studies have shown that LEK can provide more accurate information than open source databases resulting in fact-based conservation policies at local levels (Groombridge et al., 2004).

9.1.1 The Gaoligong Mountains: Conservation Priority

The Gaoligong Mountains (GLGMs, 23°48-26°23′N, 97°38′-98°16′E) are located across the northern section of the ChinaMyanmar border at the convergence of three globally important biodiversity hotspots: the Himalaya, Indo-Burma, and the Mountains of Southwest China (Meyers et al., 2000). The GLGMs run north-south for more than five latitudes across an area of some 500 km. The main ridge of these mountains forms part of the China-Myanmar border. The GLGMs also represent the divide between two major river basins the Irrawaddy to the west and the Salween (Nu Jiang in Chinese) to the east. .

With abundant rainfall, an elevational drop of 6,135 m, diverse microclimates, complex topography, and multiple latitudinal belts, the GLGMs are rich in vegetation types ranging from moist tropical forest to alpine meadows and contain a highly diverse set of flora and fauna (Li et al., 2000; Chaplin, 2005). According to incomplete accounts, the area is home to 486 avian and 117 mammalian taxa (Xue, 1995; Dumbacher et al., 2011), many of which have been recently described, including two flagship primate species: the black snub-nosed monkey (*Rhinopithecus strykeri*) and the Skywalker hoolock gibbon (*Hoolock tianxing*) (Geissmann et al., 2011; Fan et al., 2017). The area is of high priority for global conservation and has been listed by the Alliance for Zero Extinction (AZE) as an important site (Ricketts et al., 2005). The Chinese part of the GLGMs is recognized as a core component of the Three

Parallel Rivers World Natural Heritage site (WNH) (UNESCO Ref: 1083bis, 2003). Given that the GLGMs are divided by the boundary of two nations, cooperative transboundary management and conservation are needed from both China and Myanmar.

To conserve the biodiversity of this region, the Chinese Government established the 4,055 km^2 Gaoligongshan National Nature Reserve (GLGMNNR) in 1983 (Fig. 9.1). On the Myanmar side, the area partially falls into the Northern Forest Complex (25,800 km^2 of intact mountain forest along the international border with China and India), which is one of 15 national Priority Corridors/Priority Sites for protected areas (PA) and conservation investment (National Biodiversity Strategy and Action Plan Myanmar, 2011). At present, only 22.8% of the Northern Forest Complex has been placed under conservation protection (Fig.1). With the support of Fauna & Flora International (FFI), the Government of Myanmar (MoNREC) is working to establish a new national park in Imawbum, Sawlaw Township, Northeastern Kachin State to protect the habitats exploited by the black snub-nosed monkey (Meyer et al., 2017). Once established, this new national park will connect to the GLGMNNR (China) at the Pianma Township of Lushui County, Yunnan. Despite this initiative, a gap in conservation protection still exists, which extends from the east bank of N' Mai Kha River to the main ridge of Gaoligongshan Mountains, on the Myanmar side of the border (Fig. 9.1). Forests and endangered species within this lacuna face the imminent threat of wildlife trade (Meyer et al., 2017; Zhang et al., 2017) and deforestation for slash-and-burn agriculture, timber, mining, and infrastructure development. Internal conflicts and insurgencies in the area also have contributed to illegal logging and the wildlife trade (Geissmann et al., 2011; EIA, 2015). Thus, both the Chinese and Myanmar governments, as well as international organisations are committed to strengthening the conservation protection of existing reserves in the GLGMs, and establishing new PAs along their northern border. However, to move these conservation management priorities forward, an accurate assessment of biodiversity distribution across the GLGMs is needed (Basnet et al., 2017).

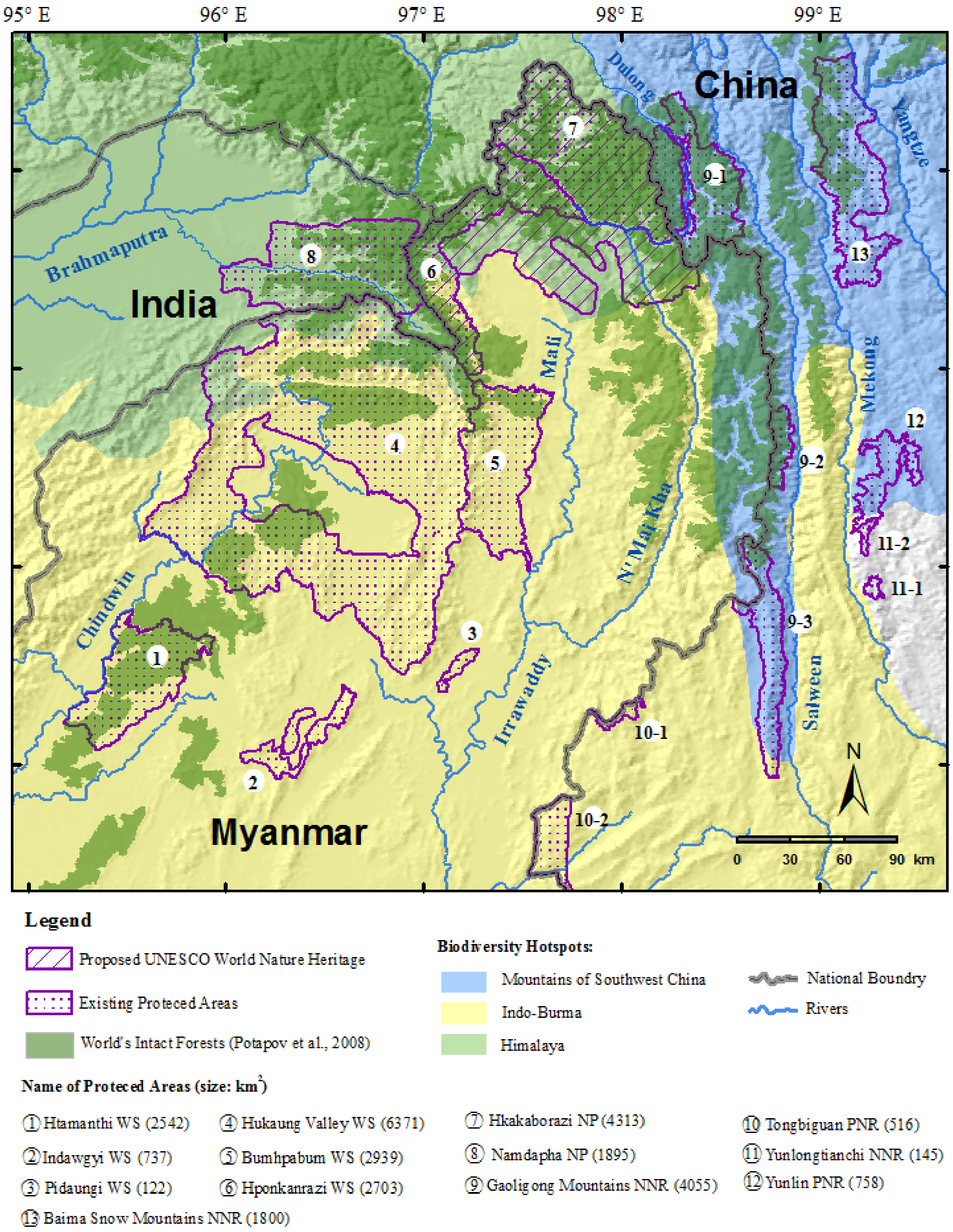

Figure 9.1 Conservation areas in Northern Myanmar and along the Northern Sino-Myanmar Border (NP = National Park; WS = Wildlife Sanctuary; NNR = National Nature Reserve; PNR = Provincial Nature Reserve).

In this study, we use community interview data along with 3D vision maps to obtain the LEK needed to develop a transboundary conservation priority plan for the GLGMs in the area of the Sino-Myanmar border. Our specific goals are (1) to select five charismatic and threatened flagship species as surrogates in order to model habitat distribution and use multicriteria decision analysis (MCDA) of habitat suitability to prioritize transboundary conservation areas within this poorlysurveyed but highly biodiverse region; (2) to promote transboundary cooperation between China and Myanmar; and (3) to present a practical, rapid and integrative method to identify and prioritize effective small-scale conservation projects that can be used in mountainous regions.

9.2 Materials and Methods

9.2.1 Vegetation types in the study area

Along the elevational gradients of the GLGMs, natural vegetation types include tropical monsoon rain forest (< 1,000 m a.s.l.), monsoon evergreen broad-leaved forest (1,100-1,800 m a.s.l.), semi-moist evergreen broad-leaved forest (1,700-2,500 m a.s.l.), mid-montane moist evergreen broad-leaved forest (1,900-2,800 m a.s.l.), coniferous broadleaved mixed forest (2,700-3,500 m a.s.l.), and alpine bushes or meadows (> 3,400 m a.s.l.) (Li et al., 2000).

9.2.2 Data collection

Community interviews

Interview-based surveys provide useful and low-cost information about both the distribution and abundance of animal taxa (Meijaard et al., 2011; Ma et al., 2015; Turvey et al. 2015, 2017). We initially selected 24 pheasant species, one bovid, the Mishmi takin (*Budorcas taxicolor*), and seven primate species as conservation targets for our surveys. These species were selected because they are relatively abundant and easily identified by local people (Ma et al., 1995). We interviewed local residents along the entire range of the GLGMs in Nujiang Autonomous Prefecture between July 2010 and July 2013 and made a specific effort to visit villages near areas of existing forests, and to interview older hunters or villagers who have made frequent visits into the forest. Semi-structured interviews were carried out by trained investigators (Bunce et al., 2000) in which local residents were shown species' photographs from a photo pool and asked if any of the species existed in the area. If participants answered yes, then a prepared questionnaire was administered that asked about the animals' group size, the encounter time, date and year, forest types inhabited, as well as the animals' and the interviewees activities in the forests. Then the interviewee was asked to identify and draw the approximate location (polygon) of species' sightings using Google Earth 3D maps that were highlighted with recognizable locations and key environmental features (see also Ma et al., 2015).

Species selection

Following past protocols (Tulloch et al., 2011; Di Minin and Moilanen., 2014; Santarem et al., 2019), we selected our five target or flagships from our initial survey of 32 species using the following criteria: (1) the species' should be of relatively large body length (70 cm) and easily identifiable; (2) the species' should have medium or large home ranges (100 ha); (3) the species' should be relatively abundant in the study area; (4) the species' should be locally recognized as a flagship and or umbrella species and endemic to the Eastern Himalaya Region (Dorji et al., 2018); (5) the species' should inhabit more than one forest type and their distributional ranges should at least partially overlap with other flagship species; (6) the species is facing population decline and habitat destruction and thus requires a detailed population and habitat distribution assessment for protection; and (7) both legislation and public awareness require conservation funds must be available to protect these species (i.e. *Yunnan Provincial Conservation Program of Extremely Small Populations, 2010-2020*).

Based on the above criteria, we identified five flagship species, including: Sclater's monal (*Lophophorus sclateri*), the Mishmi takin (*B. taxicolor*), Shortridge's langur (*T. shortridgei*), the Skywalker hoolock gibbon (*Hoolock tianxing*) and the black snub-nosed monkey (*Rhinopithecus strykeri*) as surrogates for determining the location of critical habitat and conservation zones for species protection (Figure Appendix V, Text Appendix IV).

We conducted field investigations of these five flagship species from 2013 to 2017 in areas of GLGMs that were relatively accessible and had a high probability of encounter with wild populations. A high probability of encounter was based on analyzing historical records of the distribution of each of the five species. Along with interview-based surveys, we collected the five species' presence data from recent published records. The field team of Fauna & Flora International (FFI) also investigated evidence of *H. tianxing* and *T. shortridgei* in the Imawbam Mountains in Myanmar. We combined the information from China and Myanmar into a single data set in our modeling analyses.

9.2.3. Environment data acquisition and vegetation mapping

As climatic conditions and vegetation types are two major factors limiting the distribution of wild animals (e.g. Kissling et al., 2010), we used both to predict the distribution of the selected species. For climate data we included 19 bioclimatic variables derived from 48 climate factors including maximum, minimum, and mean monthly temperature, and mean monthly precipitation (Table Appendix XII) obtained from WorldClim (30 arcsec spatial resolution, equal to 0.00833, http://www.worldclim.org, see Hijmans et al., 2005). We downloaded 32 Landsat ETM+/OLI images from USGS Earth Explorer (http://

earthexplorer.usgs.gov/) to map vegetation cover and vegetation type across the study area (Table Appendix XIV). As with WorldClim, we transformed the Landsat images to a Universal Transverse Mercator coordinate system (UTM, WGS84 datum, Zone 47 N) and spatially interpolated to 30 m resolution via bilinear interpolation.

We obtained “Ground-truth data” by systematically sampling 510 sets of 30 m ×30 m squares (9 squares per set) using visual interpretation of high-resolution images from Google Earth (Figure Appendix I). We used R Statistics software to generate square sample plots every 23.7 km along four angles from the first plot. Given the constraints of the software and the resolution of these images (based on Landsat TM/ETM+/OLI images (resolution = 30 m), we used a random forest algorithm with the “groundtruth data” (Breiman, 2001) to divide the study area into three habitat types: woodland (open forest and closed forest), shrub and alpine grassland, and non-vegetated areas (such as buildings and permanent glaciers). If the tree cover was higher than 40% in a square, it was considered woodland; otherwise, it was considered as non woodland, such as, shrub or grassland, or built areas such as roads and villages (Table Appendix XV). We employed R Statistics software (http://www.r-project.org/) with the Random Forest package (Liaw and Wiener, 2002) and Raster package for this analysis (Hijmans, 2015b).

9.2.4 Species distribution modeling

In recent years, MAXENT ecological niche modeling has proven to be a robust predictor of species' distribution and effective in identifying priority conservation areas, especially for rare and cryptic species (Thorn et al., 2009; Ren et al., 2017). As data on species occurrence and environmental variables are required to set up a MAXENT model, we randomly selected 10,000 pixels (30 m resolution, based on the 67 climate variables) from the study area as background data. Among these pixels, we employed 2,000 samples as pseudo-absence data for the MAXENT model and used the remaining 8,000 samples for validation. Each of the distribution polygons for *T. shortridgei*, *H. tianxing*, *L. sclateri*, and *B. taxicolor* were aggregated and rasterized at a resolution of 30 m. We randomly and separately selected 2,000 pixels from the distribution polygons for the four species. Seventy percent of the 2,000 pixels of each species were systematically sampled as presence data and the remainder were used as test data. The MAXENT model for each species was built using the 1,400 presence samples and the 2,000 background samples based on 67 climate variables assessed by the Jackknife variable contribution test (see Table Appendix XII for a list of these 67 climate variables, Pearson, 2007), with the default parameters setting in the R package, dismo (Hijmans et al., 2015b). As a result, climatic suitability layers for each of the four species in the study area were obtained.

For appraising core, medium, low quality and unsuitable habitat for each species, four levels of climate suitability data were quantified according to corresponding parameters of the MAXENT model (Table Appendix XVI). We found that the habitats occupied by the four species differed in terms of vegetation types. Therefore, according to the different habitat requirements of each species, we masked out non-habitat areas from the climatic suitability layer by using the data collected on vegetation cover in 2015. For example, *H. tianxing* inhabit principally intact subtropical broad-leafed forests, therefore, areas such as open forest, shrub and grass lands, farmland, and buildings were regarded as non-habitat for this species and omitted.

For ***R. strykeri***, we used data and results from recent studies by Ma et al. (2014) and Ren et al. (2017), based on LEK and MAXENT modeling, to predict their current range and habitat preferences.

We evaluated the models' predictive accuracy for each species using the area under the curve (AUC) of the receiver operating characteristics (ROC) plot. The ROC curve adopted the sensitivity and specificity as a reference for judging the predictive accuracy of the model. The AUC area of the ROC curve varies from 0 to 1, with values > 0.9 signifying strong predictive value (Araújo et al., 2005). A map of habitat suitability for all of the species was then plotted in ArcGIS 10.0.

9.2.5 Multi-criteria decision analysis

Prioritization using Multi-Criteria Decision Analysis (MCDA) has been widely used in natural resource management and conservation planning (Adem Esmail and Geneletti, 2018). The MCDA makes decisions by considering conservation goals and economic costs, as well as by exploring the trade-offs between different alternative decisions across multi criteria to achieve a particular goal (Adem Esmail and Geneletti, 2018). By weighing different considerations, five criteria including habitat suitability, species' conservation status, endemism, landscape levels (the presence of intact forests), and costs (distance to settlements) were used to map the suitability of local habitats to prioritize protected conservation areas. The evaluation column in Table 9.1 presents the reasons used to evaluate each criterion. The aggregate score for each pixel (30 m × 30 m) in the study areas was calculated using the equation:

$$CA = \sum H \times (I+E) + IF - R$$

Where H is the value of habitat suitability, I is the IUCN Red List Category level, E is the level of a species' endemism, IF is the score of Intact Forest Landscape area (see Table 2 and Potapov et al., 2008), R represents the distance to human settlement areas or residential areas, and CA is the aggregate score of a pixel (30 m × 30 m). We then used the Quantile

Classification Scheme to order the values into 3 categories (priority 40%, 60% and 100%) according to each score's accumulated contribution rates and input the score into ArcGIS 10.0 to map prioritization areas for conservation. A pixel with a CA value > 0 was designated as a conservation priority area. In addition, any pixel with a CA value 8, indicated that the area within the pixel contained edge habitat suitable for at least two flagship species' and was considered as high priority conservation area. Each of the above procedures was conducted in R Statistics software (version 3.0.3, R Core Team, 2016) along with additional packages such as raster (Hijmans et al., 2015a) and dismo (Hijmans et al., 2015b).

Table 9.1 Five criteria used in prioritization.

<table>
<tr><th>Layer 1</th><th colspan="2">Core habitat</th><th>Medium habitat</th><th colspan="2">Edge habitat</th><th>None habitat</th></tr>
<tr><td>MAXENT</td><td colspan="2">3</td><td>2</td><td colspan="2">1</td><td>0</td></tr>
<tr><th>Layer 2</th><th><i>T. shortridgei</i></th><th><i>H. tianxing</i></th><th><i>R. strykeri</i></th><th><i>L. sclateri</i></th><th><i>B. taxicolor</i></th><th>Value Identification</th></tr>
<tr><td>IUCN Red List Category</td><td>3</td><td>4</td><td>4</td><td>2</td><td>2</td><td>4: CR, 3: EN, 2: VU, 1: NT</td></tr>
<tr><td>Endemism</td><td>3</td><td>3</td><td>4</td><td>2</td><td>2</td><td>4: endemic to study area, 3: endemic to GLGMs; 2: endemic to Eastern Himalaya Mountains</td></tr>
<tr><th>Layer 3</th><th>Intact (2016)</th><th>Degraded (2000-2016)</th><th>None</th><th colspan="3">Value Identification</th></tr>
<tr><td>Intact Forest Landscape (IFL);</td><td>4</td><td>2</td><td>0</td><td colspan="3">IFL, which also contains naturally treeless areas such as grasslands, wetlands, and alpine areas, as defined by Potapov et al. (2008). The IFL map for the year 2016 was complicated by multiple joint research institutes (see http://www.intactforests.org/news.html). Value 4 was assigned to IFL, Value 2 to degraded IFL between 2000 and 2016, and a Value of 0 to areas with no intact forest.</td></tr>
<tr><th>Layer 4</th><th>< 1 km</th><th>1-2 km</th><th colspan="4">Value Identification</th></tr>
<tr><td>Distance to resident zone (1:250000)</td><td>-4</td><td>-2</td><td colspan="4">As forests surrounding human settlement are frequently impacted by human disturbance and degradation, maintenance and restoration of these habitats require large investments of time and funding to convert to a protected area. Therefore, for limiting conservation cost, if a patch was located within 1 km of a resident zone, it was assigned a value of -4; a patch within 1-2 km from the resident zone was assigned a value of -2.</td></tr>
</table>

9.3 Results

9.3.1 Distribution data of selected flagship species

We identified 38 localities with the possible presence of ***T. shortridgei***. We estimated that there were 250-370 Shortridge langurs residing in 19 groups located in the Dulong River Valley and possibly one group in the lower forests in Pianma Township. Subsequent field surveys confirmed two groups in the Dulong River Valley. In addition, data on eight distribution areas of *T. shortridgei* in the Imawbum Mountains in Myanmar were obtained by camera traps from 2010 to 2017. After merging all overlapping records, we obtained a distribution for *T. shortridgei* of 43 non-overlapped polygons ranging from 0.2 to 96 ha (Table 9.2, Fig. 9.2).

We obtained 15 distribution records for ***H. tianxing***, (11 of which were pre-1985 and the others from 1985 to 2000) from interviews with local villagers. This information suggested the existence of gibbons in Shangjiang and Pianma townships of Lushui county in the southern part of Nujiang Prefecture. Data for this species' southernmost distribution in the GLGMs were obtained from Fan et al. (2011) and Chan et al. (2017). We identified 17 separate forests in nine townships over three counties. For Myanmar, using data obtained by Fauna & Flora International (FFI) (covering an area of 3,000 km^2 to the east of the N'mai Kha River) we identified 12 records of gibbon sightings or calls along with three historical records of gibbons located in the Myitkyina and Imawbum Mountains east of N'mai Kha River from an altitude of 380 m to 2,620 m above sea level. Overall, *H. tianxing* was found in 73 independent polygons ranging in area from 3.6 to 64.6 ha (Fig. 9.2).

Table 9.2 Data from interview-based surveys and field surveys.

Species	Data type	Date of investigation	Number of people & villages visited	Investigated Areas	Independent distributed ranges (polygons)	Published Reference
T. shortridgei	Interview survey	Jul. 2010 - Aug. 2012	715 residents from 118 villages	Lushui, Fugong, Gongshan & Dulong County	36	Cui et al., 2016
	Field survey	2010 - 2017	—	Mt.Imawbom in Myanmar	7	
R. strykeri	Interview survey	Apr. 2011 - Dec. 2012	358 residents from 68 villages	Lushui, Fugong, & Gongshan County	46	Ma et al., 2014
	Interview survey	Feb. - Mar., 2010	65 residents from 33 villages	Maw River areas of Myanmar	3	Geissmann et al, 2011
H. tianxing	Interview survey	Apr. 2011 - Dec. 2012	358 residents from 68 villages	Lushui, Fugong, & Gongshan County	13	
	Field survey	Mar.- Aug. 2009	—	Baoshan, Tengchong & Yinjiang Countys	45	Fan et al., 2011
		Apr. - Nov. 2016	—	Tengchong		Chan et al., 2017
		2004 -2017	—	Mt. Imawbom in Myanmar	15	
L. sclateri	Interview survey	May. 2012 - Aug. 2012	370 residents in 78 residential areas of 68 villages	Lushui, Fugong, & Gongshan County	49	
	Field survey	Oct. 1999 - Aug. 2003	—	Baoshan, Tengchong, Lushui, Fugong, & Gongshan County	15	Han et al., 2004
B. taxicolor	Interview survey	May. 2012 - Aug. 2012	370 residents in 78 residential areas of 68 villages	Lushui, Fugong, & Gongshan county	153	Pan et al., 2019
	Interview survey	Dec. 2012 - Jul. 2013	25 villages	Dulong County	30	Zhang et al., 2014

Notes: The green highlight shows the interview survey completed by authors in this study.

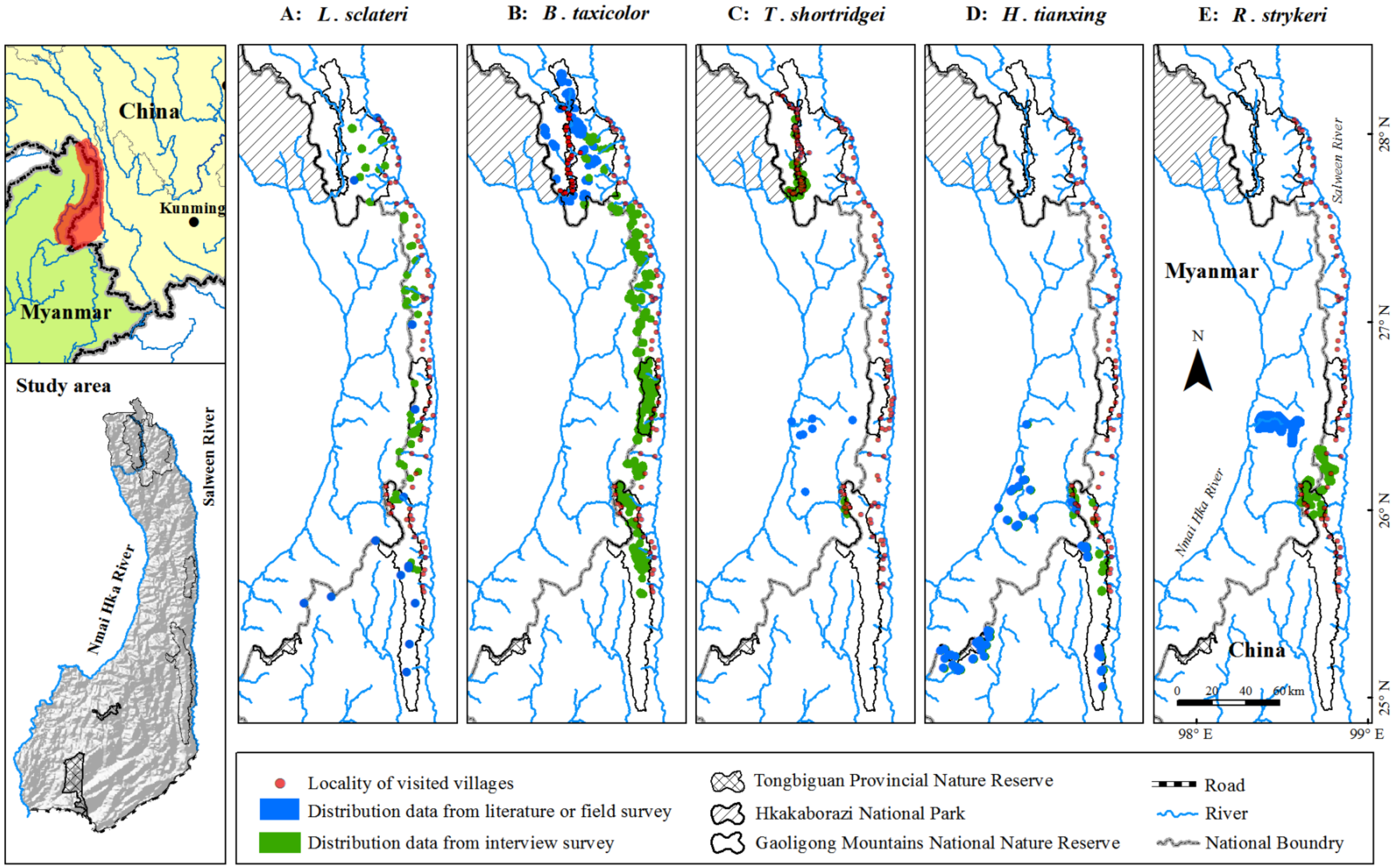

Figure 9.2 Map of the study area and distribution of five selected flagship species based on interviews and field surveys in the Sino-Myanmar border region.

Based on interviews, we obtained 129 encounter records for ***L. sclateri*** (Table 2). We observed *L. sclateri* at four localities. Using data from Han et al. (2004), who documented *L. sclateri* at 15 sites located in the area from 25°50'-28°50'N to 98°33'-98°33'E. This species was found to be present in 64 non-overlapped polygons ranging in size from 2.6 to 109 ha (Fig. 9.2).

Our results indicate 254 distribution records for ***B. taxicolor*** covering an area of 4,200 km^2 within the Nujiang Prefecture of the GLGMs. Adding Zhang et al.‘s (2014) results of 30 takin distribution ranges obtained during 2012-2013 in 25 villages in the Dulong River Basin (Table 2) to our 30 living traces (feces and footprints), three encounters with wild groups (group size ranged from 10 to 58 individuals), and camera trap records of takin groups or individuals at six interviewee-reported sites (including one group estimated to contain 65 individuals), we identified 183 polygons ranging in area from 0.7 to 3,181 ha inhabitated by this species (Fig. 2).

9.3.2 Habitat distribution of selected flagship species

The AUC values for the *T. shortridgei*, *H. tianxing*, *R. strykeri*, *L. sclateri* and *B. taxicolor* were 0.914, 0.969, 0.957 (Ren et al., 2017), 0.975 and 0.978, respectively, demonstrating that our habitat evaluation models had high efficiency in predicating the distributions of these five species' in the study area.

The predicted habitat distributional range and habitat suitability of each of the five flagship species within the study area differed markedly (Table 9.3). The habitat predicted for *T. shortridgei* covered 8,691 km^2 at an elevation of 1,100-2,900 m. Most of the highly suitable habitat for this species (85.3%) was found on the western slope of the GLGMs in Myanmar, and only about 16.9% of the highly suitable habitat (core + medium) was located in

Hkakaborazi National Park in Myanmar and the GLGMNNR in China. *R. strykeri* and H. tianxing were distributed in the mid and southern most segments of the GLGMs, covering an area of 3,575 km^2 (see Ren et al., 2017 and Fig. 9.3) and 6,424 km^2 respectively. Currently, only about 13.9% of *R. strykeri*'s highly suitable habitat is present in the GLGMNNR. In the case of *H. tianxing*, China and Myanmar each shared almost half of the remaining highly suitable habitat at an elevation between 1,000 and 2,700 m. Only 25.1% of the remaining highly suitable habitat for *H. tianxing*'s is located in protected reserves. These are the Tongbiguan Provincial Nature Reserve and the GLGMNNR, in China.

L. sclateri was distributed principally along the main ridge of the GLGMs [55.4% highly suitable habitat (Table 9.3) in China and 44.6% in Myanmar], covering 9,277 km^2 of the study area, principally at the elevational range of 2,700-3,900 m. Only 35.5% of the highly suitable habitat of this species was located in currently PAs. *B. taxicolor* had the broadest distribution of all five species, covering 9,932 km2. This species' main habitat was predicted to be between 2,700 and 4,600 m, with 46.9% of the highly suitable habitat located in PAs. For *B. taxicolor*, 63.8% of highly suitable habitat was located in China and 36.2% in Myanmar.

9.3.3 High priority areas

Fig. 9.4A shows the results of the prioritization areas produced by the MCDA. The CA value ranged from 2 to 61. The area of pixels with a CA value 16 accounted for 40% (8,299.5 km^2, Fig. 4B, dark green parts) of the priority area indicating that these areas are the most valuable for conservation as they include medium suitable habitat for at least two species or the core habitat of at least one species within the intact forests. Pixels with a CA value 12 accounted for 60% (12,464 km^2, dark green parts + light green parts) of the priority area and include the most critical habitats of each of the five flagship species along with other fauna in the

GLGMs. In total, thus 60% priority area (CA > 12) accounted for only 28.7% of the total study area. This likely reflects that fact that the high biodiversity areas in the GLGMs are distributed in relatively small and fragmented locations and that approximately 10,398.7 km^2 (83.4%) of the high priority area occurred outside of the existing system of PAs.

Table 9.3 Summary of different categories of suitable habitats and elevational range predicted for each of the selected flagship species.

Species	**Core habitat (km^2)**	**Medium habitat (km^2)**	**Edge habitat (km^2)**	**Elevation (m)**	**China (core + medium, %)**	**Myanmar (core + medium, %)**	**Highly suitable habitat in reserves (core + medium, %)**	**Note:**
T. shortridgei	1,525.6	3,426.2	3,739.2	1,105-2,986	14.7	85.3	16.9	
H. tianxing	1,128.8	1,916	3,379.5	1,091-2,695	48.3	51.7	25.1	
R. strykeri	1,420	750	1,405	2,330-3,240	29.3	70.7	13.9	Ren et al., 2017
L. sclateri	1,477	3,299	4,451	2,710-3,824	55.4	44.6	35.5	
B. taxicolor	2,052.5	2,742.7	4,536.6	2,307-3,770	63.8	36.2	46.9	

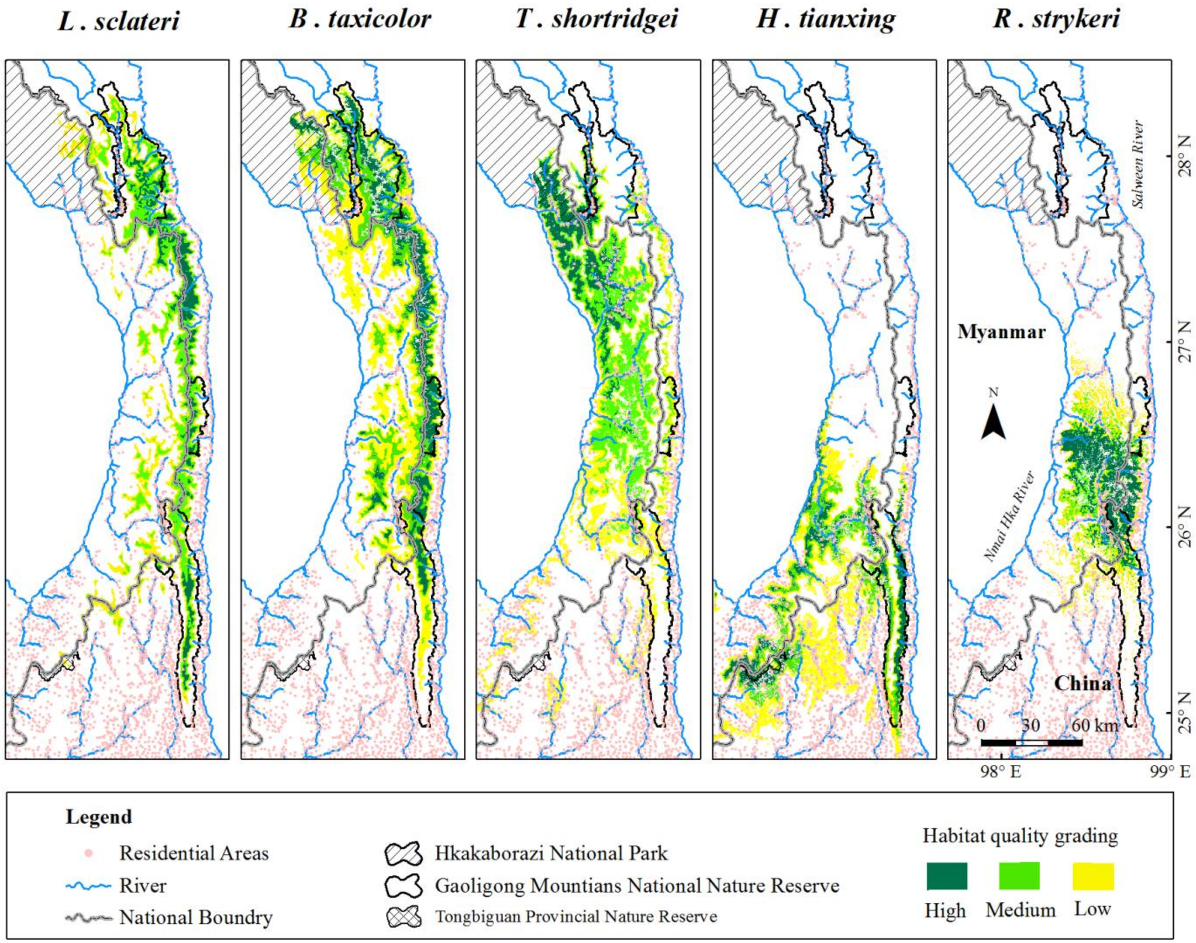

Figure 9.3 Predictive habitat distribution and habitat suitability of five selected flagship species based on MAXENT modelling in the Sino-Myanmar border region.

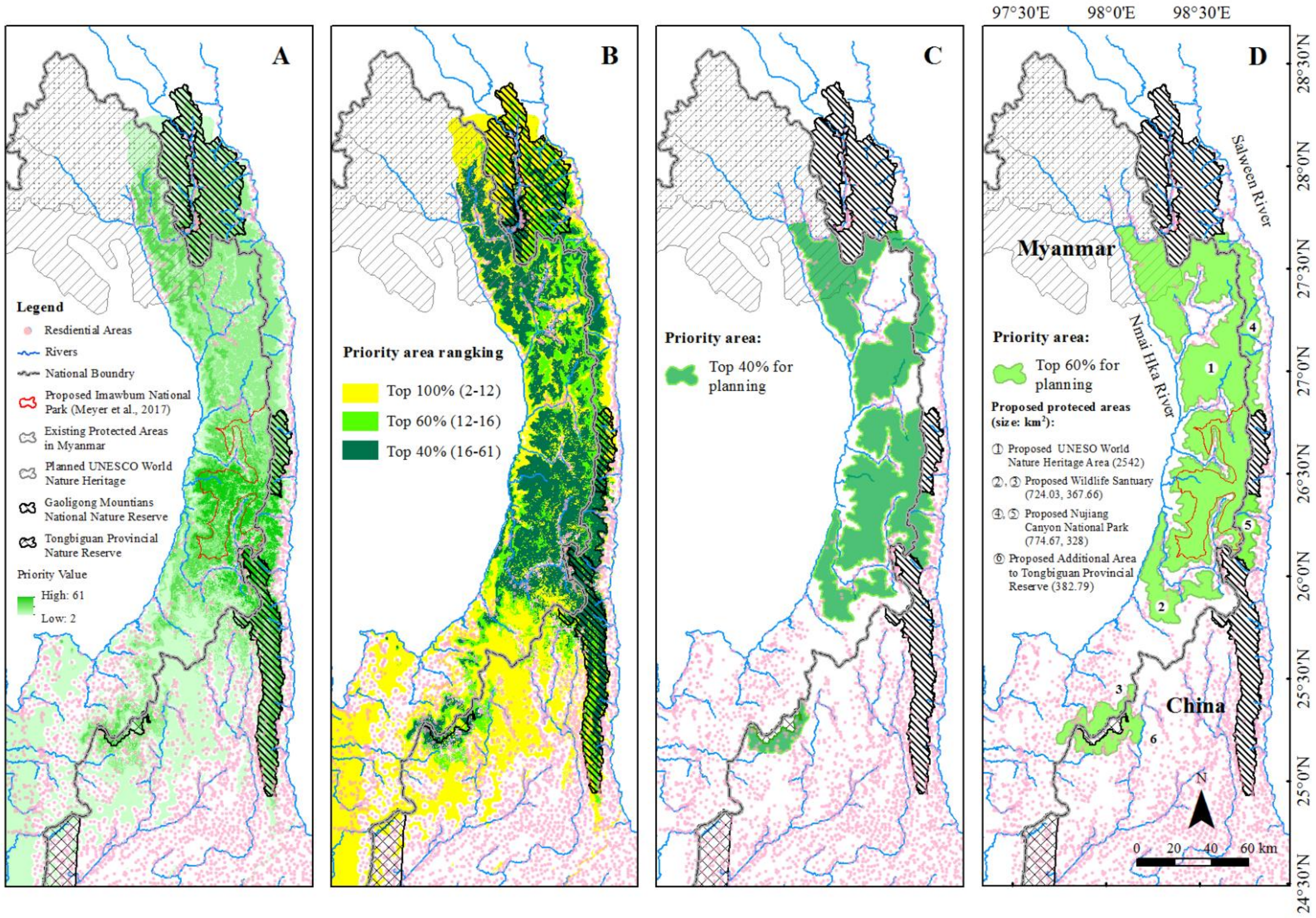

Figure 9.4 Priority areas for transboundary conservation along the north Sino-Myanmar border based on the remaining habitat distribution area for the five selected flagship species and multi-criteria decision analysis. C: Top 40% priority area; D; 60% priority area.

9.4 Discussion

The availability of sufficient and robust data are key to fine-scale conservation planning and evidence-based conservation management. However, difficult terrain, dense forests, low species encounter rates, and logistical challenges in remote mountainous areas make traditional animal survey methods such as linear transects time-consuming, labor-intensive and impractical. This is the situation in the GLGMs (e.g. Yang et al., 2018; Yang et al., 2019). Moreover, data on species occurrences and distributions obtained from out-of-date or incomplete open source materials can generate large inaccuracies or omission errors (Figure Appendix VI and Figure Appendix VII). Our results show that when used together, large-scale community interviews assisted by 3D visual maps, selective field surveys, and species distribution modeling can represent an effective set of research tools providing reliable data from which to predict population size, distribution, and availability of suitable habitat, as well as the conservation status of difficult to study species (cf. Ma et al., 2014; Turvey et al., 2015; Turvey et al., 2017; Ren et al., 2017). In combining data on species-specific habitat distribution patterns with the location and distance to roads, residential areas, agricultural fields and mining sites, and conservation prioritization methods (e.g. MCDA in this study) or software (e.g. Zonation, Marxan and C-Plan), researchers can effectively identify the location of critical conservation areas, travel or dispersal corridors, and target new areas for habitat restoration as part of a finer scale - lower cost conservation strategy than based on traditional methods (see also Fig. 5).

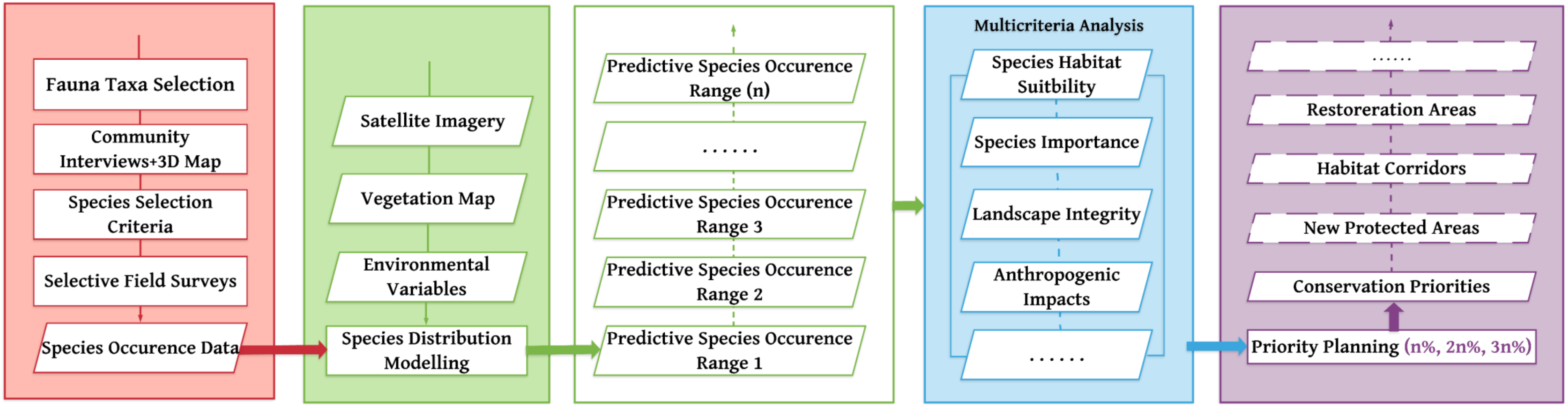

Figure 9.5 A recommended conservation prioritization framework for relatively quick and reliable conservation planning in poorly-studied areas.

The results indicate that over 80% of the priority conservation areas for the five flagship and threatened species examined in this study currently occur outside of the existing protected area system. Thus, urgent rethinking of the existing conservation policies and planning and management of the protected area network in the GLGMs are required. We propose the following three primary recommendations:

9.4.1. Upscaling the methodology for regional information collection and monitoring

The vast biologically-rich region from the GLGMs to northeastern India and northern Myanmar (the Northern Myanmar Forest Complex) represents a mountainous geographical zone that presents significant challenges regarding the acquisition of conservation information. These challenges include extremely poor accessibility, highly rugged topography, dense forests, low human population density, a severe shortage of information on the behavior and ecology of threatened species, prohibitively high costs of data acquisition, and limited government budgets for intensive field surveys. Adopting the approaches and methodologies introduced in this paper can provide park rangers and conservation managers with the data required to make critical conservation policies at an affordable cost. Moreover, using a similar region-wide methodology serves to generate quality and comparable landscape level information and promote meaningful regional collaboration for conservation. Many international organisations that work on landscape conservation such as the International Centre for Integrated Mountain Development (ICIMOD), FFI, and Wildlife Conservation Society (WCS) advocate the use of standardized approaches for collecting conservation data.

9.4.2. Coordinating conservation policies and planning between Myanmar and China for the GLGMs

It is essential that China and Myanmar strengthen transboundary collaboration in their conservation planning and implementation in the GLGMs in order to effectively protect vulnerable and endangered animal and plant species, whose populations or habitats are distributed on both sides of these national borders.

In the current study, we identified five flagship species that are geographically restricted to the GLGMs or easternHimalaya region, and are facing a high risk of extinction (Text Appendix IV). Landscape integrity and connectivity (see Fig. 4C and D) are thought to be essential for maintaining animal populations, thus we recommend that the top 60% of priority areas identified in our MCDA model (Fig. 9.4D) be set aside and protected for conservation. This would require Myanmar to add approximately 7,055 km^2 into their current PA system, including three large patches (Fig. 4D, No. 1: 5,693 km^2, No. 2: 724.03 km^2, and No. 3: 367.66 km^2) that are fragmented by rivers and human inhabitation. In China, the priority areas are mainly located on the eastern slopes of the GLGMs, comprising two unprotected areas along the main ridge (Fig. 4D, No. 4: 774.63 km^2 and No. 5: 328 km^2) and an additional area adjacent to the Tongbiguan Provincial Nature Reserve (Fig. 4D, No. 6: 382.79 km^2, bordering with No.3). Given that most of the southern part of the study area is densely settled by humans, its value as a conservation area is limited.

While the Government of Myanmar (MoNREC) is planning to create the Imawbum National Park in Sawlaw Township to protect its remaining population of some 200 *R. strykeri* (Fig. 4A, Meyer et al., 2017), the proposed area may be too small to protect other threatened species such as the *H. tianxing* and *T. shortridgei*. In both China and Myanmar, less than 300 individuals of each species exist in PAs. Additionally, MoNREC proposed the Hkakabo Razi

Landscape as Myanmar's first candidate for a UNESCO WNH site in 2014. The proposal recommended that the heritage area extend to the east side of the N'Mai Kha River and be connected with the GLGMNNR of China in order to strengthen its natural and conservation values (UNESCO Ref.: 5871, 2014). We, therefore, propose that that priority area No.1 (Fig. 4D) be added to the Hkakabo Razi Landscape as a WNH area (Text Appendix V) and that priority areas No. 2 and No. 3 be designated as wildlife sanctuaries to protect the remaining populations and habitats of *H. tianxing* and *T. shortridgei*, as well as other threatened and endangered primates such as Phayre's langur (*T. phayrei*) and the northern slow loris (*Nycticebus bengalensis*) (Momberg et al., 2010). Habitat No. 3 (Fig. 4D) has undergone extensive banana cultivation (Zhang Lixiang, pers. comm.) and thus requires conservation measures such as "enclosing the mountain for natural afforestation" and an active program of reforestation to ensure wildlife protection. Moreover, area No. 6 should be added to the Tongbiguan Provincial Nature Reserve in order to link the No. 3 wildlife sanctuary with the transboundary conservation area for protecting *H. tianxing*. Furthermore, the mid- and southern parts of the Nujiang Grand Canyon National Park, Area No. 5 (Fig. 4D), which contains the remaining major population of *R. strykeri* should be added to the Chinese National Park system (see The Program for National Park Development in Yunnan Province, 2009-2020). Finally, the proposed Nujiang Grand Canyon National Park (Fig. 4D, Areas No. 4, and 5) in China and the WNH area in Myanmar are expected to cover all or most of the remaining habitat of *B. taxicolor* and *L. sclateri* (an area of 7,443 km^2), along with other endangered fauna such as the grey-bellied tragopan (*Tragopan blythii*), the red panda (*Ailurus fulgens*), and the dhole (*Cuon alpinus*).

These measures would result in the creation of a large and continuous belt of protected conservation areas extending from China to India through Myanmar, resulting in one of the world's largest transboundary conservation networks, covering 13 PAs over 30,067 km^2, four

Alliance for Zero Extinction sites, four globally important ecoregions, and eleven key biodiversity areas. In addition the successful inclusion of the Hkakaborazi Landscape (including the proposed addition of area No 1) into the WNH list will provide a framework of opportunity for China (the Three Parallel Rivers WNH area) and Myanmar to develop and promote the transboundary management of a joint WNH site. The two countries also could work jointly to create an international peace park for biodiversity conservation, and ecotourism along the northern Sino-Myanmar border (see Vasilijevic et al., 2015).

Increasing coordination between China and Myanmar in conservation policy-making and planning can help to identify conservation gaps, the locations of critical biological corridors, and common conservation priorities.

9.4.3. Exploring possibilities for joint or transboundary protected areas along the northern China-Myanmar border

Considering the transboundary nature of the socio-ecological systems, in particular, the cross border distribution of many small ethnically diverse human populations and highly endangered flagship wildlife species and their habitats in the GLGMs, effective conservation policies require that decision-makers, managers and scientists of China and Myanmar work together to study the possibilities of joint or transboundary protected area management, especially in the northern sector. This is especially relevant given that some of the currently protected areas created by China and Myanmar are geographically connected.

Transboundary PAs are needed to secure and promote the ecological integrity and long-term survival of resident and migratory species, while transboundary conservation can generate socio-cultural and economic benefits to local human communities (Vasilijevic et al., 2015). In the past decade, transboundary collaboration for conservation and development in the

Hindu Kush-Karakoram-Himalaya region has made considerable headway in protecting transboundary landscapes of high biodiversity value through initiatives such as the Kangchenjunga Landscape Initiative (Gurung et al., 2019) and the Landscape Initiative for Far-Eastern Himalayas (HI-LIFE) (Basnet et al., 2018). This collaboration has been facilitated by the ICIMOD. HILIFE, which aims to foster long-term conservation partnerships between China, India and Myanmar. The initiative has facilitated much dialogue among the key stakeholders including policy makers from both sides (Basnet et al. 2017, 2018). The on-going efforts by the Government of Myanmar to inscribe the Hkakaborazi Landscape (including the proposed addition of area No 1) into the WNH list, if successful, will provide additional opportunities for China (the Three Parallel Rivers WNH area) and Myanmar to develop and promote transboundary management of this WNH site. We therefore recommend that further steps be taken to explore the possibilities for joint or transboundary protected area management between China and Myanmar in the GLGMs. To achieve this goal, the following initial steps can be taken:

- Establish regular mechanisms for policy makers, PA managers, conservation scientists and members of concerned local communities to share biodiversity data and management information, jointly identify common conservation priorities and gaps in the existing PA network, and exchange experiences, best practices, and management programs and policies;

- Develop joint research projects to investigate and monitor key threatened species and their habitats and study landscape level changes in deforestation and the affects of climate change species survivorship using common methodologies;

- Jointly monitor and control cross-border poaching, wildlife trade, logging, forest fires, non-timber forest product harvesting and trading, and respond to natural disasters; and

- Promote the transparent, equitable and sustainable cross-border trading of biological resources.

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Chapter 10 Conclusion and Recommendations

10.1 Introduction

This dissertation research is the first long-term and systematic study on the feeding ecology and conservation biology of the black snub-nosed monkey (*Rhinopithecus strykeri*) in Sino-Myanmar border areas. Some of the major achievements of this research were 1) the discovery of a second population of this species with 70 individuals in the Luoma area thus confirming their existence in Salween basin in China and providing a description of the environment phenology of the study site; 2) obtaining the first data on the feeding ecology and nutritional ecology of *R. strykeri* and using this to create a comparison of feeding ecology and ranging behaviour of all snub nosed monkey species; 3) assessing the habitat distribution, range and suitability of existing habitat for the survival of this species through an evaluation of current threats and its natural habitat; and 4) providing a conservation framework that can be used to prioritize transboundary conservation networks not only for this species but also for other four flagship species (Sclater's monal *Lophophorus sclateri*, Mishmi takin *B. taxicolor*, Shortridge’s langur *T. shortridgei*, and Skywalker hoolock gibbon *Hoolock tianxing*) in Gaoligong Mountains in Sino-Myanmar region. This chapter will summarize key findings and provide recommendations for future research and conservation.

10.2 Summary and Recommendations

1. Based on two years of surveying the remote and little disturbed forest of the Gaoligong Mountains National Nature Reserve in China, with outline transect sampling and infrared camera monitoring, a breeding population comprising > 70 individuals was found on the eastern slopes of the Gaoligong Mountains in the Salween River Basin. Given the Critically Endangered status of this primate (a total of < 400 individuals are estimated to remain in the

wild, Meyer et al., 2017), efforts to protect the relatively undisturbed habitat of this newly discovered population and to prevent hunting are essential for the long-term survival of this species. Conducting uninterrupted patrolling, monitoring and long-term research on this population' ecology and behaviour (i.e. feeding habits, ranging behaviours, habitat utilization pattern etc.) will promote such efforts because frequent presence of researchers and rangers at study areas may facilitate law enforcement, lower illegal activities in the area, introduce monitoring technology to local rangers, and promote public awareness of the improtance of primate conservation and local biodiversity. Also, feasible reserch outcomes of conservation needs can provide scientific guidance for the allocation of funding for relevant institutions. Long-term research and monitoring requires sufficient funding support, therefore, the reserve administration should actively cooperate with relevant research institutions and NGOs for funding application and planing the use of funds. Many NGOs (e.g. FFI, Kadoorie Farm and Botanic Garden, and the Nature Conservancy), universities (e.g. Southwest Forestry University and Sun Yat-sen University) and research institutes (e.g. IEHBR and ICIMOD) have common interests in biodiversity study and protection in Gaoligong Mountains; thus, the reserve administration should establish a consultative committee and involve related experts from these conservation and research organisations for advices on conservation programs and related funding applications.

2. Based on field observation of the Luoma population in the wild and cafeteria-style feeding trials on two captive *R. strykeri* with 600 plant species collected in their habitat, I found *R. strykeri* can consume diverse foods and food species, including young and mature leaves, fruits/seeds, buds, flowers, twigs, and bark from 170 different species of trees, bushes, and herbs representing 76 genera and 41 plant families, as well as 15 species of lichen. They also extract foods from ground and prey insects. Food plants consumed by *R. strykeri* found principally in intact subtropical evergreen broadleaf forests and hemlock-broadleaf mixed

forests at an altitude of 2,200–3,000 m. Strict enforcement of habitat protection and access to resources across this elevation zone appear to be essential for the conservation and survivorship of this Critically Endangered primate. To prevent the severely illegal logging of high-value tree species (including food tree species) *R. strykeri* monkey habitat, setting up a red list and legislation to strictly prohibit the trading of these primary forest trees in the local wood markets in Nujiang Prefecture are indispensable; and replacing the local timber-processing industry with other green industries is imperative to eliminate the driving factors of illegal logging. For some local poor residents, illegal logging is an important and easy source of income. Therefore, developing other alternative income-generating activities (e.g. traditional medicine cultivation, see setails in sect. 10.3.2) for local poor families also is important to address illegal logging.

By conducting cafeteria-style feeding trials to two captive *R. strykeri* and comparing the nutrient content of 100 leaf samples the monkeys selectively consumed (n = 70 plant species) with the nutrient content of 54 leaf samples (n = 48 plant species) the monkeys' avoided, I found that in spring and autumn, *R. strykeri* selected leaves that were high in moisture, crude protein, total nonstructural carbohydrates and phosphorus and low in neutral detergent fibre content. Also, foods selected in autumn were characterized by a higher amount of metabolisable energy than those rejected. Although my sample was limited to the only two captive individuals in the world, considering that our results are consisted with nutritional profiles of other species of wild snub-nosed monkeys, I propose that cafeteria-style feeding trials provide valuable insights into the feeding habits of rare and difficult to study primate species. In addition, the nutritional analysis of these food items has highlighted some key nutrients of importance for the species that could be used to determine key food species in the wild. For example, 32 food tree species had higher protein-to-fibre (Neutral Detergent Fibre) ratios than all others. Among these species, 18 species also had greater contents of moisture, crude protein, total nonstructural carbohydrates, and phosphorus and may thus be considered as high priority species that can use for forest rehabilitation in destroyed and degraded *R. strykeri* habitat or identified corridors resulting by moor-burning, commercial logging, and mining. For example, *Acer wardii*, *Betula cylindrostachya*, *Lithocarpus pachyphyllus*, and *Skimmia arborescens* can be used for lower evetional habitat or corridor restoriton for *R. strykeri*.

3. In order to identify the most urgent gaps in the conservation of *R. strykeri*, I employed MAXENT and interview-based survey data to predict the distribution and alterations in its habitat over the past 15 years. *R. strykeri* appeared to inhabit a range from E98°20′–98°50′, N25°40′–26°50′, including high quality habitat at 1,420 km^2, medium quality habitat at 750 km^2, and low quality habitats at 1,410 km^2. Approximately 2.6% of the entire habitat has been lost in the past 15 years, 96% of which has been in Myanmar. To save this species from extinction, it is urgent to establish trans-boundary conservation and management networks to address the loss of habitat, and to locate and preserve key wildlife corridors to link fragmented habitats between Myanmar and China. Accordingly, two national park areas, Gaoligong Mountains National Park (328 km^2, now named Nujiang Grand Canyon National Park by Yunnan Provincial Government) in Luzhang township in China and Imawbum National Park (1,934 km^2) (Fig. 8.2) in Sawlaw Township in Myanmar are proposed for transboundary conservation of this species. More future works should concentrated on locating more *R. strykeri* populations and clarifying their population status in predicted habitats and establishing a joint patrol team to combat illegal wildlife hunting and illegal logging in two parks (see sect. 10.3).

4. To address challenges of data collection in Gaoligong Mountains (GLGMs), interview-based surveys assisted by Google earth 3D maps were applied to locate the distributional

range of several native animal taxa in the GLGMs. Based on selection criteria I set, five flagship species (*L. sclateri*, *B. taxicolor*, *T. shortridgei*, and *H. tianxing*) in different niches were nominated form investigated fauna species as surrogates of biodiversity. Selective field surveys of each species were conducted to determine their presence/absence to confirm the reliability of species distribution data obtained from interview-based surveys. Multicriteria decision analysis techniques were then used with data on habitat suitability (MAXENT Models) to prioritize transboundary conservation areas. Only about 13.9%, 35.5%, 46.9%, 16.9%, and 25.1% of *R. strykeri*, *L. sclateri*, *B. taxicolor*, *T. shortridgei*,, and *H. tianxing*'s highly suitable habitat (core + medium) are located in existing PAs, respectively. There are approximately 10,398.7 km^2 of remaining habitat with high conservation value for each of the five flagship species is unprotected. This includes six large zones separated by rivers and human settlements that should be designated as the transboundary World Nature Heritage, National Parks, or Wildlife sanctuaries along the northern Sino-Myanmar border (see Fig. 9.4 D). The method I used in this study presents a reliable, rapid and integrative data collection and planning procedure for identifying areas of conservation prioritization in data poor areas. This method can be applied successfully to assess conservation priorities in other difficult to access and mountainous regions.

10.3 Further Recommendations for Conservation

The conservation of *R. strykeri* in Sino-Myanmar border area will be a challenge, especially as this relatively pristine area is increasing its exposure to the outside world in the context of development for the eradication of poverty. This creates an increase list of threats to the species (listed in Chapter 4 of this dissertation and Section 3 in Meyer et al. 2017). Such increased threats will require new strategic action plans be put into place to ensure protection of the species and their habitat. I would suggest the following recommendations be incorporated into such plans in order to optimise their chance of success.

10.3.1 Locating remaining populations

Knowing populations of this species and the size of the habitat of all populations will better enable us to predict the possible size and distribution of the entire species, to get a better sense of their vulnerability. For example, I currently don't have a reliable population density for *R. strykeri*, thus I am forced to use the density data of *R. strykeri* to estimate the total population size of *R. strykeri*. With a total of 1,700 individuals (Jin & Long, 2010) and the entire habitat estimate of 4,169 km^2 (Xiao et al., 2003), the density of the black-and-white snub-nosed monkey (*R. bieti*) is estimated to be 0.41 individuals per km^2, from which we roughly calculate a total of approximately 900 individuals of *R. strykeri* in existence, given 2,105 km^2 of high and medium quality habitat in Myanmar and China. Including the low quality habitat of 1405 km^2, the total population may be as much as 1,460 individuals. High resolution images on Google Earth, however, show that the forest in some marginal habitats (e.g. Fig. 8.2, black circled area NO. M3) have been highly degraded and fragmented by large scale logging, road creation and burning; making them unsuitable as viable habitat options for *R. strykeri* populations in the long term and should thus be excluded from the population prediction. Accordingly, the top population size of *R. strykeri* is probably around 1,200 individuals.

Considering that interview-based population censers covered only 270 km^2 (11%) of the habitat in Myanmar, the population size should be higher than 330 individuals in Myanmar and some population in China have been eradicated (Geissmann et al., 2011; Ma et al., 2014; Meyer et al., 2017), so highlighting 900 - 1,200 individuals may be a more convincing estimate for the total population size in the whole border area. However, after eight years

(2010-2018) extensive field surveys in Sino-Myanmar border areas, I and my colleagues only confirmed five sub-populations with approximately 400 individuals existing in the Sino-Myanmar border areas, due to the difficulties (steep terrain, long-term rainfall, hard logistics, ethnic conflicts etc.) hindered the survey process.

A lack of available and robust data on their total population size and distribution status is one critical issue faced by management for the conservation of *R. strykeri* and its habitat in the border area. Therefore, conducting additional surveys and locating remaining populations of *R. strykeri* in this area will give a baseline for creating prioritizes for formulating conservation actions. Further population investigations should be conducted in the core habitats predicted (see Chapter 7) which have not been investigated and monitored. However, it is difficult to accomplish this daunting task solely by the power of conservationists because they often lack sufficient funds and human resources to sustain long-term field searching in such the remote region. The high investment of field surveys and low returns of available data also stop other biologists inform joining in. The success of such an endeavor, would therefore also require the involvement of government departments of the two countries of interest who would be best able to charging national surveys to clarify the population size and distribution of *R. strykeri* like that the National Giant Panda (*Ailuropoda melanoleuca*) Survey Chinese Grovenment did. The relevant government departments (i.e. MoNREC, NFGA) are able to provide sufficient survey funding, charge earlier survey plan laying and designing, coordinate and organize joint survey teams and expert supports form two countries, and supplely logistic helps (cf. Wei et al., 2018). Moreover, the government departments should create a special funding program to subsidize zoologists and conservationists to continue the relevant (or transboundary joint) research programs in long-term in this region.

10.3.2 Mitigating poaching and logging effects

Nujiang Prefecture still remains one of only 14 areas of China with prevailing poverty and is the key target area for China’s drive for poverty alleviation. However, the announced *Yunnan Red Lines for Ecological Conservation* (2018) has placed 70% of the land area of the Nujiang Autonomous Prefecture into protected zones. This highlights the contradiction between the importance of the area for ecological conservation and the need for economic development for local human communities, and thus the need to create policies of sustainable development that include benefits both for local people and for the environment.

In GLGMs, poverty stands as the main driver of poaching as it an easy way to increase income and sustenance to ethic households. Although poaching events that target *R. strykeri* have been recorded less in recent years, poaching is still being recognized as a major threat to the monkey populations in Sino-Myanmar border areas. Moreover, increasing economic development in the region may stimulate subsistence hunting to transform into commercial hunting in response to market demands from outside and the arrival of tourists and workers (e.g. game demands from wealthier communities, Challender & MacMillan, 2014). Similarly, the livelihoods of illegal loggers have been considerably sustained through revenue generated from forest products sales. Projects aiming to conserve threated species populations and their habitat by alleviating poaching and illegal logging have depended upon income-based poverty reduction and prompt local residents to suspend poaching and illegal logging activities (Dudley, 2004; Knapp et al., 2017). These projects, however, have mixed results and often do not provide people with a consistent alternate income. Therefore, enhancing the benefits of these programs to local people in the transboundary areas is essential and could be reached using the following recommendations:

(a) jointly implement sustainable community development projects at the border area to reduce poverty and decrease the pressure on biodiversity resources by local communities. This could be achieved by providing alternative livelihood projects (e.g. locally traditional medicine cultivation and beekeeping), related technologies and necessary assistance to poor households, and conducted project tracking process by nature reserve officials to ensure the project success (Fig. 10.1); For example, compared with *Euodia rutaecarpa* which can be grown under three years' relatively extensive management for fruiting (Fig. 10.1), while cultivation of *Paris* spp. needs five years' intensive managment for fruiting. However, the current market price per kg of *Euodia rutaecarpa* dry seeds is ranged of 250-350 yuan depending on its quality, while the price for *Paris* is > 2,000 yuan per kg for seeds and > 800 yuan per kg for dry roots depending on the species cultivated and their quality. Therefore, the proper development project should be selected in accordance with the willingness, actual economic situation and endurance of the local communities.

Figure 10.1 Small-scale planting trials of *Euodia rutaecarpa* in Pianma.

(b) secure and provide more job opportunities like forest rangers or park rangers to local people; train local rangers to participant in wildlife monitoring (i.e. camera trapping, walking transect investigation) or sustainable development projects. Many local hunters have richer local ecological knowledge in terms of plants and animals identification, distribution, and explaining qualitative population trends. Recruiting local hunters as rangers not only can reduce the pressure of poaching but also may help PAs to adjust strategies and resources toward more proper conservation measures by recognising hunters' LEK, skills, and social economic context. Currently, the basic salary for forest rangers in Nujiang Autonomous Prefecture is far lower than average level of Yunnan Province (700 vs. 1,200 yuan per mo). Therefore, this recommendation will only be effective if the wage far exceeding the hunting earnings is guaranteed, which needs the financial support from local and provincial government .

(c) operate plantation programs (i.e. local fast growing species: *Betula alnoides, Betula luminifera*) in rehabilitation areas surrounded PAs to provide alternative sources for replacing highly desirable resources to decrease the pressure on demands of wood, non-timber forest products and firewood from PAs;

(d) pursue positive traditional cultural practices linked to wildlife and engage local communities in conservation programs to take advantage by using their local knowledge and inspire a sense of pride for the iconic species (i.e. GLGMs' big five: *R. strykeri*, *B. taxicolor, H. tianxing, T. shortridgei, L. sclateri*) surrounding their living areas;

(e) carry out targeted awareness raising and conservation education programs in prioritized villages within and surrounding the protected area and in primary and secondary schools in townships;

(f) religious propaganda: most Lisu and Nu people (> 50 %) believe in Christianity (Gao, 2017). By a chance conversation with two villagers, we learned that the parish priest once preached that the poaching is not a life-respecting activity, resulting that the villagers decided not to participant in this activities anymore (Pu Sancai & Liu Pu, pers. comm., 2016). Therefore, conservation related sermon may be efficient way that can be applied to mitigate poaching;

(g) suppress the local timber-processing industry (promulgating a decree or taking administrative measures by local government) thus to eliminate the driver of illegal logging activities; develop nature-based tourism or ecotourism (e.g. bird watching, primate ecotourism), agritourism or ethnic tourism with essential training in business skills, guiding and production of ethnic goods and handicraft to reconstruct regional pillar industry. Local peoples' everyday life may be outsiders' adventure. PAs should not only are able to ‘sell’ a limited range of wildlife watching and landscape experiences to tourists but also ensure some products (including handicrafts or cultural performances) can be sold by local ethnic people for diversifying their livelihoods and raising household incomes.

(h) secure funding and provide technical assistance to establish transboundary management planning and start-up of operations for the Nujiang Canyon National Park in China and Imawbum National Park in Myanmar, establish and operate collaborative patrolling law enforcement systems between China and Myanmar, and foster trans-boundary collaboration on law enforcement between China and Myanmar. These are top-down steps and require coordination between relevant departments of the two governments;

(i) monitor and control cross-border poaching, wildlife trade, logging, forest fires, non-timber forest product harvesting and trading, and other natural disasters; and promote the transparent, equitable and sustainable cross-border trading of biological resources;

(j) strength cooperation and communication with conservation/wildlife management agencies and NGOs (i.e. FFI, ICIMOD, Could Mountain Conservation etc. with primate conservation funding) for available projects or supports.

All in all, assembling all recommendations from Section 10.2 and 10.3 provide long-term solutions for addressing the threats of the *R. strykeri* and protecting other sympatric flora and fauna species in Gaoligong Mountains along Sino-Myanmar border areas. Reaching an agreement of management objective and conservation measures in all stakeholders in the transboundary areas is very important to ensure that these measures ban be materialized. The last point, hence, is that unless many of these recommendations are become actions, the long-term survival of the *R. strykeri* would not be promised. This is a huge challenge for the future.

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Table Appendix I. Band size, habitat information and diet category for snub-nosed monkeys

Species	**Study Site**	**Mean Elevation (m)**	**Mean Latitude (N)**	**Annual Temperature (℃)**	**Annual Rainfall (mm)**	**Mean Band Size**	**Number of Food Species**	**Lichen**	**Leaves**	**Fruit & Seeds**	**Flowers**	**Buds**	**Bark**	**Reference**
R. avunculus	Khau Ca	950	22.83	23.3	2,300	51	50	0	53.1	32.2	12.2	–	0	Dong, 2012
R. brelichi	Fanjingshan	1,550	27.92	9.2	1,433	250	107	0	47.3	21.6	9.4	15.3	–	Xiang et al., 2012
R. roxellana	Baihe	2,575	33.23	7.8	811.4	280	117	5.9	54.6	19.1	3.9	3.9	3.9	Li, 2016
R. roxellana	Shennongjia	2,065	32.5	7.5	1,800	236	133	38.4	22.4	30.3	1.3	5.8	0.2	Liu et al., 2013;Li et al., 2014
R. roxellana	Zhouzhi East	2,150	33.8	6.4	980	112	84	29	24	24	–	4.2	11.1	Guo et al., 2007
R. roxellana	Zhouzhi West	2,000	33.78	9.2	690	188	123	1.3	35.92	21.42	7.52	18.44	8.07	Huang, 2015
R. roxellana	Qingmuchuan	1,493	31.37	12.9	972	110	–	0	25	25.94	–	10.65	10.65	Li et al., 2010
R. roxellana	Guanyingshan	1,900	33.67	11.5	924	70	53	22	20	25	–	11	15	Zhao et al., 2015
R. bieti	Wuyapiya	3,950	28.5	0.9	740	175	27	86	6	0.1	–	–	–	Kirkpatrick, 1996
R. bieti	Xiaochangdu	3,875	29.25	4.7	936	210	25	82.1	6.05	1.1	1.1	6.05	4.2	Xiang et al., 2007

R. bieti	Samage	3,300	27.57	7.5	1,004	410	94	67	16.5	11.4	0.2	3.6	0.5	Grueter et al., 2009b
R. bieti	Xiangguqing	3,350	27.6	9.8	1,370	480	105	50.6	24.7	10.5	1.9	3	0.8	Li et al., 2011a
R. bieti	Tacheng	3,200	27.6	7.5	1,370	360	55	60	–	–	–	–	–	Ding & Zhao, 2004
R. bieti	Longma	3,023	26.23	8.8	1,501	80	98	–	–	–	–	–	–	Huo et al., 2005

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Table Appendix II. Band sizes, daily range lengths, home rang sizes, time budgets and feeding efforts of temperate snub-nosed monkeys (*Rhinopithecus bieti & R. roxellana*)

Species	**SS**	**ME (m)**	**MT**	**MGS**	**MD (m)**	**SpD**	**SuD**	**AD**	**WD**	**MH km^2**	**SpH**	**SuH**	**AH**	**WH**	**F**	**T**	**R**	**S**	**FE**	**Reference**
R. roxellana	Baihe	2,575	7.8	280	-	-	-	-	-	22.13	8.94	10.25	6	8.13	-	-	-	-	-	Li, 2016
R. roxellana	Shennongjia	2,165	5	129	1,113	1,325	1,320	1,930	1,035	-	-	-	-	-	-	-	-	-	-	Li et al., 2005
R. roxellana	Shennongjia	2,165	5	104	-	1,575	2,375	1,795	1,150	-	-	-	-	-	-	-	-	-	-	Li, 2002
R. roxellana	Shennongjia	2,085	7.5	236	-	-	-	-	-	-	-	-	-	-	28.1	39.2	27.2	5.4	2.47	Liu, 2013a,b
R. roxellana	Shennongjia	2,085	7.5	236	-	-	-	-	-	22.5	18.6	14.5	19.4	12.3	-	-	-	-	-	Fan et al., 2019
R. roxellana	Shennongjia	2,085	7.5	62	-	-	-	-	-	12.4	12	11.7	-	6	-	-	-	-	-	Fan et al., 2019
R. roxellana	Zhouzhi East	2,150	6.4	112	2,100	2,200	2,600	1,900	1,600	18.3	11.9	5	2.9	7.1	35.8	22.9	36.2	5.1	1.62	Guo et al., 2007; Tan et al., 2007;
R. roxellana	Zhouzhi West	2,000	9.2	90	-	-	-	-	-	22.5	14.1	9.5	12.1	12.3	35	19	27	12	2.84	Li et al., 2000

R. roxellana	Qingmuchuan	1943	12.9	110	-	-	1,020	-	676	-	-	8.09	-	7.43	-	-	-	-	-	Li et al., 2010
R. bieti	Wuyapiya	3,950	0.9	175	1,311	1,465	1,601	1,059	1,183	25.25	3.88	3.25	3.08	2.5	32	36	22	10	3.09	Kirkpatrick et al., 1998
R. bieti	Xiaochangdu	3,875	4.7	210	765	-	-	-	-	21.2	-	16.75	-	10.5	49.1	20.4	17.8	12.7	3.9	Xiang et al.,2007; 2010; 2013
R. bieti	Samage	3,300	7.5	410	1,514	1,721	1,516	1,877	985	32	17.8	18.6	9.3	18.2	38.5	19.1	28.5	13.9	2.02	Grueter et al., 2008; Grueter et al., 2013
R. bieti	Xiangguqing	3,350	9.8	450	-	-	-	-	-	14.79	-	-	-	-	38.8	27.4	20.9	12.9	3.17	Li et al., 2011a
R. bieti	Tacheng	3,200	7.5	360	-	-	-	-	-	-	-	-	-	-	35	15	33	17	1.52	Ding & Zhao, 2004
R. bieti	Fuhe	3,100	11.1	80	801	1,013.8	1,013.8	601.6	601.6	10.7	8.4	8.4	4.9	4.9	30	15	41	16	1.01	Liu et al., 2004
R. bieti	Lasha	3,200	4.7	100	1,210	1,032	1,826	1,142	839	8.2	5.3	4.5	4.9	2.7	41.7	22.6	27.5	8.4	2.34	Wang, 2016, Huang et al., 2017
R. bieti	Longma	3,150	8.8	80	-	-	-	-	-	9.6	-	-	-	-	-	-	-	-	-	Huo, 2005
R.bieti	Jinsichang	3,400	6.5	180	909	870	1,023	940	814	23.3	7	7.3	6	5.1	-	-	-	-		Ren et al., 2009a, b

SS = study site; ME = mean elevation; MT = mean annual tempreture; MBS = mean Band size; MD = mean daily range length; SpD = spring daily range length; Sud = summer daily range length; AD = Autumn daily range length; WD = Winter daily range length; MH = mean home range; SpH = spring home range; SuH = summer home range; AH = autumn home range; WH = winter home range; F = feeding time; T = travalling time; R = resting time; S = socialling time; FE = feeding effort

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Table Appendix III. The comparison results for prediction 1, 2, 3 by using Pearson correlation

Independent Variable	Dependent Variable	Disk Free (df)	Coefficient of Determination (R^2)	p-value of statistical test	Note
Prediction 1					
Mean Elevation	No. of food species	11	0.12	0.254	
		9	0.58	0.07	Excluded sites of Khau Ca and Guanyinshan
Mean annual temperature		11	0	0.897	
		9	0.61	0.004	Excluded sites of Khau Ca and Guanyinshan
Annual rainfall		11	0.01	0.796	
		9	0.12	0.29	Excluded sites of Khau Ca and Guanyinshan
Mean latitude		11	0.12	0.254	
		9	0.15	0.274	Excluded sites of Khau Ca and Guanyinshan
Mean Elevation	Feeding time of lichen	11	0.83	< 0.001	
Mean annual temperature		11	0.53	0.007	
Annual rainfall		11	0.05	0.48	
Mean latitude		11	0.05	0.46	
Mean Elevation	Feeding time of leaves	10	0.47	0.014	
Mean annual temperature		10	0.38	0.033	

Annual rainfall		10	0.17	0.187	
Mean latitude		10	0.02	0.687	
Mean Elevation	Feeding time of fruits/seeds	10	0.88	< 0.001	
Mean annual temperature		10	0.48	0.012	
Annual rainfall		10	0.28	0.077	
Mean latitude		10	0.03	0.589	
Feeding time of leaves	Feeding time of lichen	10	0.69	0.001	
Feeding time of fruits/seeds		10	0.68	0.001	
Feeding time of flowers		10	0.69	0.001	
Feeding time of buds		10	0.4	0.005	
Feeding time of bark		8	0.12	0.325	
Prediction 2					
Mean annual temperature	Feeding efforts	8	0.19	0.206	
Mean elevation		8	0.12	0.324	
Mean home range		7	0.07	0.498	
Mean daily range length		7	0.07	0.498	
Mean band size		8	0.02	0.678	
Mean annual temperature	Mean daily range length	6	0.02	0.74	

	Mean home range	11	0.09	0.315	
Mean band size	Mean daily range length	6	0.02	0.762	
	Spring daily range length	6	0.06	0.558	
	Summer daily range length	7	0.01	0.783	
	Autumn daily range length	6	0.1	0.444	
	Winter daily range length	7	0	0.875	
	Mean home range size	11	0.27	0.068	
	Spring home range size	8	0.18	0.226	
	Summer home range size	10	0.4	0.028	
	Autumn home range size	7	0.08	0.455	
	Winter home range size	10	0.45	0.017	
Spring daily range length	Spring home range size	4	0.33	0.236	
Summer daily range length	Summer home range size	5	-0.261	0.572	
Autumn daily range length	Autumn home range size	4	0.237	0.651	
Winter daily range	winter home range size	5	0.069	0.882	

length					
Prediction 3					
Spring daily range length	Summer daily range length	6	0.769	0.026	
Summer daily range length	Autumn daily range length	6	0.599	0.117	
Autumn daily range length	Winter daily range length	6	0.679	0.064	
Winter daily range length	Spring daily range length	6	0.891	0.003	
Summer daily range length	Winter daily range length	7	0.841	0.005	
Spring daily range length	Autumn daily range length	6	0.745	0.034	
Spring home range	Summer home range	8	0.845	0.002	
Summer home range	Autumn home range	7	0.679	0.044	
Autumn home range	Winter home range	7	0.649	0.058	
Winter home range	Spring home range	8	0.899	< 0.001	
Summer home range	Winter home range	10	0.839	0.007	
Spring home range	Autumn home range	7	0.782	0.013	

Table Appendix IV. Cranial and Dental Measurements of a male and femal Specimens of *Rhinopithecus strykeri*

Variable of cranium (mm)	Holotype, male, AIMZ 15504.a	Paratype , female, KZISTRYKEI.b	Variable of cranium (mm)	Holotype, male, AIMZ 15504.a	Paratype , female, KZISTRYKEI.b
Cranial length nasion—opisthocranion	101.5	98.41	Length of M^3 (lower)	-	12.03
Greatest skull length prosthion—opisthocranion	132.8	125.34	Breadth of M^3 (lower)	-	7.44
Bizygomatic breadth	105.7	82.21	Length of M^2 (lower)	-	8.16
Cranial breadth across vault (bieuryonic)	79.1	69.16	Breadth of M^2 (lower)	-	7.13
Cranial breadth across supramastoid crests	89.6	80.41	Length of M^1 (lower)	-	7.54
Postorbital breadth	55.1	53.64	Breadth of M^1 (lower)	-	6.35
Cranial height, basion—bregma	59.8	55.58	Basal length (basion to prosthion)	96	85.31
Nasion to basion	-	70.59	Biorbital breadth, inner	77.1	70.14
Maximal length of foramen zygomaticum	41.85	30.41	Biorbital breadth, outer	93.7	78.36
Palate length, prosthion to staphylion	51.9	49.53	Orbit breadth	31.2	27.2
Palate breadth, across (a) M^3–	42.2	37.31	Orbit height	27.35	26.99

M^3					
Palate breadth, across (b) canines	42.1	31.94	Face height, prosthion to nasion	53.1	45.33
Maximal length of postcanine toothrow, maxilla	36.1	34.45	Mandible length infradentale (a) to gonion	85.1	77.63
Maximal breadth of postcanine toothrow, maxilla	9.1	39.14	Mandible length infradentale (b) to condyle	10.075	89.51
Canine length (mesiodistal)	10	9.99	Bimental breadth	28.33	24.76
Canine breadth (labiolingual)	8	6.28	Minimal corpus height between M_1 and M_2	28.55	2.38
Length of M^3 (upper)	9.3	8.18	Mandibular width, bicondylar	84.33	73.23
Breadth of M^3 (upper)	8.75	8.78	Ascending ramus height, perpendicular from tip of coronoid	58.78	53.54
Length of M^2 (upper)	8.1	7.91	Maximal ascending ramus height, gonion to centre of condyle surface	47.3	44.65
Breadth of M^2 (upper)	9	8.8	Breadth incisura mandibulae	23.7	16.16
Length of M^1 (upper)	-	7.7	Coronoid to gonion	55	45.25
Breadth of M^1 (upper)	-	7.47			
a. data published by Geissmann et al. (2011); b. data masured from a female specieman stored in Kunming Institute of Zoology. CAS.					

Table Appendix V. List of the *Rhinopithecus strykeri*'s food resources

ⅰ. Sp = Spring; Su = Summer; Au = Autumn; Win = Winter;

ⅱ. OS = Occasionally selected; F = Consistently selected; P = Preference;

ⅲ. 1 = Subtropical Evergreen Broadleaf Community Forest; 2 = Disturbed Subtropical Evergreen Broadleaf Forest (by Slight Agriculture or Slight Felling) or Secondary Subtropical Evergreen Broadleaf Forest Community (Mixed Evergreen - Deciduous Broadleaf Forest); 3 = Deciduous Broadleaf Forest (3-1 = Primary Forest; Degraded Forest, disturbed by felling, clearing or wildfire: 3-2 = dominant by *Lithocarpus*, *Cylobalanopsis* and *Ilex*, *Magnolia* or regenerating *Tsuga* at high elevation; 3-3 = dominant by *Alnus nepalensis*, *Betula spp*, *Populus yunnanensis* or *Pinus yunanensis* at low elevation); 4 = Mixed Broadleaf - Hemlock Forest or Hemlock – Rhododendron Forest; 5 = Hemlock - Bamboo Thicket Forest; 6 = Bamboo Thicket and Alpine Scrub.

Latin Name	Family	Part Consumed							Biotype	Altitude (m)	Habitat Environment (T)	Note
		Bud	Young leaf	Mature leaf	Flower/ Inflorescence	Fruit/ Seed	Young Twig	Bark				
Larix speciosa	Pinaceae	CS Sp①	CS Sp①						Deciduous Tree	2,130 - 3,600	1, 4	Field recorded
Pinus armandii						CS Au, Win			Evergreen tree	2,700 - 3,000	1, 3-2, 4	
Sabina recurva var. *coxii*	Cupressaceae					CS Sp①, Su, Au①,			Evergreen Tree	2,600 - 3,600	1, 4	Field recorded

						Win①						
Magnolia rostrata	Magnoliaceae			CS$^{\text{Su, Au}}$	CS$^{\text{Su}}$				Deciduous Tree	2,000 - 2,700	1, 2	
Magnolia campbelii		CS$^{\text{Sp}}$	P$^{\text{Sp}}$	CS$^{\text{Su, Au}}$	CS$^{\text{Sp, Win}}$		CS$^{\text{Sp}}$		Deciduous Tree	2,510 - 3,200	1, 2, 4	
Manglietia insignis		CS$^{\text{Sp}}$	CS$^{\text{Sp}}$	OS$^{\text{Au,}}$ Win		CS Su, Au			Evergreen Tree	1,900 - 2,800	1, 2, 4	
Michelia doltsopa		CS Sp, Win	P$^{\text{Sp}}$	P$^{\text{Su, Au,}}$ Win	CS$^{\text{Sp, Win}}$	CS Su, Au	CS$^{\text{Sp,}}$ Win		Evergreen Tree	1,870 - 2,700	1, 2, 3-2	
Michelia Floribunda		CS Sp, Win,	P$^{\text{Sp}}$	P$^{\text{Su, Au,}}$ Win	CS$^{\text{Sp, Win}}$	CS Su, Au	CS$^{\text{Sp,}}$ Win		Evergreen Tree	1,900 - 2,450	1, 2	
Michelia taronensis		CS Sp, Win,	P$^{\text{Sp}}$	P$^{\text{Su, Au,}}$ Win	CS$^{\text{Sp, Win}}$	CS Su, Au	CS$^{\text{Sp,}}$ Win	,	Evergreen Tree	2,000 - 2,500	1	
Schisandra grandiflora	Schisandraceae	CS$^{\text{Sp}}$	CS$^{\text{Sp}}$	CS$^{\text{Su, Au}}$	CS$^{\text{Sp, Su}}$	P$^{\text{Su,}}$ Au			Climbing Shrub	2,649 - 3,300	1, 2, 3-2, 4	
Schisandra henryi		CS$^{\text{Sp}}$	CS$^{\text{Sp,}}$ Su	CS$^{\text{Su, Au}}$	CS$^{\text{Sp, Su}}$	P$^{\text{Su,}}$ Au			Climbing Shrub	1,600 - 2,900	1, 2, 3-2, 4	
Schisandra neglecta		CS$^{\text{Sp}}$	CS$^{\text{Sp}}$	CS$^{\text{Su, Au}}$	CS$^{\text{Sp, Su}}$	P$^{\text{Su,}}$ Au			Climbing Shrub	1,820 - 2,950	1, 2, 3-2, 3-3	
Schisandra rubriflora		CS$^{\text{Sp}}$	CS$^{\text{Sp}}$	CS$^{\text{Su, Au}}$	CS$^{\text{Sp, Su}}$	P$^{\text{Su,}}$ Au			Climbing Shrub	2,710 - 3,100	1, 2, 3-2, 4	
Tetracentron sinense	Tetracentraceae	CS Win, Sp	P$^{\text{Sp}}$	CS$^{\text{Su}}$				CS Sp, Win	Deciduous Tree	2,450 - 3,000	1, 2, 3-1	
Dodecadenia grandiflora var. *griffithii*	Lauraceae		CS$^{\text{Sp}}$			P Au①, ②			Evergreen Tree	2,500 - 2,900	1, 2	Field recorded
Lindera floribunda		CS$^{\text{Sp}}$	CS$^{\text{Sp}}$						Evergreen Tree	2,900 - 3,100	4	
Lindera			CS$^{\text{Sp}}$						Evergreen	1,640 -	1, 2, 3-2, 3-	

thomsonii									Tree	3,200	3; 4, 5	
Lindera thomsonii var. *vernayana*			CS^{Sp}						Evergreen Tree	2,600 - 3,000	1, 2, 4, 5	
Lindera obtusiloba var. *heterophylla*		CS^{Sp}	P^{Sp}	CS$^{Su, Au}$					Deciduous Tree	2,450 - 3,000	1, 4	
Litsea cubeba		CS^{Sp}	CS^{Sp}	CS$^{Su, Au}$	CS^{Sp}	CS^{Au}			Deciduous Tree	2,500 - 3,200	1, 2, 3-1, 4	
Litsea kingii		CS^{Sp}	CS^{Sp}	CS$^{Su, Au}$	CS^{Sp}	CS^{Au}			Deciduous Tree	2,253 - 3,204	1, 2, 3-1, 4, 6	
Litsea rubesceus		CS^{Sp}	CS^{Sp}	CS$^{Su, Au}$	CS^{Sp}	CS^{Au}			Deciduous Tree	1,820 - 3,000	1, 2, 3-2; 3-3, 4	
Litsea Sericea		CS^{Sp}	CS^{Sp}	CS$^{Su, Au}$		CS^{Au}			Deciduous Tree	2,810-3,400	1, 2, 3-1, 6	
Machilus rufipes		CS$^{Sp, Win}$	CS^{Sp}	CS$^{Sp, Su, Au, Win}$	CS^{Sp}	CS^{Au}	CS$^{Sp, Win}$	CS^{Sp,Win}	Evergreen Tree	2,250 - 2,710	1	
Machilus salicina		CS$^{Sp, Win}$	P^{Sp}	CS$^{Sp, Su, Au, Win}$	CS^{Sp}	CS^{Au}	CS$^{Sp, Win}$	CS$^{Sp, Win}$	Evergreen Tree	1,600 - 2,510	1, 2, 3-3	
Machilus shweliensis		CS$^{Sp, Win}$	P^{Sp}	CS$^{Sp, Su, Au, Win}$	CS^{Sp}	CS^{Au}	CS$^{Sp, Win}$	CS$^{Sp, Win}$	Evergreen Tree	2,400 - 24,50	1, 2	
Sinosasafras flavinervia				OS$^{Su, Au}$					Deciduous Tree	2,300 - 2,800	1, 2, 4	
Clematis kochiana	Ranunculaceae		P$^{Sp, Su}$	P$^{Sp, Su, Au, Win}$	OS$^{Au, Win}$				Climbing herb	2,400 - 3,063	1, 2, 4	
Clematis Montana			P$^{Sp, Su}$	P^{Su}	CS^{Sp}				Climbing Shrub	2,600 - 3,000	1, 2, 3-3, 4, 5	
Clematis Montana var. *longipes*			P$^{Sp, Su}$	P^{Su}	CS^{Sp}				Climbing Shrub	2,130 - 3,600	1, 2, 3-2, 4, 5, 6	
Clematis			P$^{Sp, Su}$	P^{Su}	CS^{Sp}				Climbing	2,370 -	1, 2, 3-3, 4	

Montana var. *trichogyma*									Shrub	2,810		
Holboellia latifolia	Lardizaballaceae	CS$^{\text{Sp}}$	P$^{\text{Sp}}$	P$^{\text{Su, Au, Win}}$	CS$^{\text{Sp}}$	P$^{\text{Su}}$			Climbing Shrub	1,780 - 3,050	1, 2, 3-2, 4	
Polygala fallax	Polygalaceae		P$^{\text{Sp}}$	P$^{\text{Sp, Su, Au}}$		CS$_{\text{Su, Au}}$			Deciduous Tree	1,820 - 3,060	1, 2, 3-2, 3-3, 4	
Eurya cavinervis	Theaceae	CS$^{\text{Sp}}$				CS$_{\text{Au}}$①			Evergreen Tree	2,500 - 3,000	1, 4	
Eurya handel-mazzettii						CS$_{\text{Au}}$②			Evergreen Tree	2,350 - 2,800	1, 2, 3-2	Field recorded
Eurya jintungensis						CS$^{\text{Au}}$			Evergreen Tree	1,830 - 2,720	1, 2, 3-2,	
Eurya tsaii		CS$^{\text{Sp}}$				CS$_{\text{Au}}$②			Evergreen Tree	1,950 - 2,800	1, 2, 3-2	Field recorded
Schima argentea				OS$^{\text{Au, Win}}$					Evergreen Tree	1,600 - 2,800	1, 2, 3-2, 3-3	
Actinidia glauco-callosa	Actinidiaceae	CS$^{\text{Sp}}$	CS$^{\text{Sp}}$			CS$^{\text{Au}}$			Climbing Shrub	1,820 - 2,650	1, 2, 3-3	
Actinidia kolomikta		CS$^{\text{Sp}}$	CS$^{\text{Sp}}$			CS$^{\text{Au}}$			Climbing Shrub	1,600 - 2,800	1, 2, 3-3	
Actinidia kungshanensis		P$^{\text{Sp, Su, Au}}$	P$^{\text{Sp, Su, Au}}$	CS$^{\text{Su}}$	CS$^{\text{Su}}$	P$^{\text{Au}}$	P$^{\text{Sp, Su, Au}}$	CS$^{\text{Sp, Win}}$	Climbing Shrub	2,150 - 2,800	1, 2	
Actinidia pilosula		P$^{\text{Sp, Su, Au}}$	P$^{\text{Sp, Su, Au}}$	CS$^{\text{Su}}$	CS$^{\text{Su}}$	P$^{\text{Au}}$	P$^{\text{Sp, Su, Au}}$	CS$^{\text{Sp, Win}}$	Climbing Shrub	2,000 - 3,300	1, 2, 3-2, 4	
Actinidia venosa		P$^{\text{Sp, Su, Au}}$	P$^{\text{Sp, Su, Au}}$	CS$^{\text{Su}}$	CS$^{\text{Su}}$	CS$^{\text{Au}}$	P$^{\text{Sp, Su, Au}}$		Climbing Shrub	2,400 - 3,100	1, 2, 3-2, 4, 5	
Hydrangea davidii	Hydrangeaceae		CS$^{\text{Sp}}$	OS$^{\text{Su}}$	CS$^{\text{Sp, Su}}$				Deciduous shrub	1,820 - 2,800	1, 2, 3-3, 4	
Hydrangea heteromalla			P$^{\text{Sp}}$	CS$^{\text{Su, Au}}$	CS$^{\text{Su}}$		P$^{\text{Sp}}$		Deciduous Tree	2,850 - 3,200	3-1, 4	
Hydrangea sp.			P$^{\text{Sp}}$	CS$^{\text{Su, Au}}$	CS$^{\text{Su}}$		P$^{\text{Sp}}$		Deciduous Tree	3,100 - 3,200	3-1, 4	

Amygdalus persica	Rosaceae			CS$^{\text{Su}}$		CS$^{\text{Su}}$			Deciduous Tree	1,600 - 2,510	1, 2, 3-3	
Armeniaca mume		CS$^{\text{Sp,}}_{\text{Win}}$	CS$^{\text{Sp}}$	OS$^{\text{Su}}$	P$^{\text{Win}}$	CS$_{\text{Sp, Su}}$			Deciduous Tree	1,950 - 2,510	1, 2, 3-2	
Cerasus caudata		CS$_{\text{Sp,}}$	P$^{\text{Sp}}$	P$^{\text{Su, Au}}$	P$^{\text{Sp}}$	P$^{\text{Su}}$	CS$^{\text{Sp,}}_{\text{Win}}$	CS$^{\text{Sp,}}$	Deciduous Tree	(1,600) 2,800 - 3,450	1, 2, 3-1, 4, 5	
Cerasus clarofolia		CS$^{\text{Sp}}$	P$^{\text{Sp}}$	CS$^{\text{Su, Au}}$	P$^{\text{Sp}}$	P$^{\text{Su}}$	CS$^{\text{Sp}}$	CS$^{\text{Sp}}$	Deciduous Tree	1,812 - 2,800	1, 2, 3-2	
Cerasus henryi		CS$^{\text{Sp}}$	P$^{\text{Sp}}$	CS$^{\text{Su, Au}}$	P$^{\text{Sp}}$	P$^{\text{Su}}$	CS$^{\text{Sp}}$	CS$^{\text{Sp}}$	Deciduous Tree	1,600 - 2,700	1, 2, 3-2, 3-3	
Cotoneaster chengkangensis						CS$_{\text{Su, Au}}$			Deciduous shurb	2,300 - 2,900	1, 2, 4	
Cotoneaster franchetii						CS$_{\text{Au}}$①			Evergreen Shrub	1,820 - 3,080	1, 2, 3-2, 3-3, 4	Field recorded
Cotoneaster nitidus						CS$_{\text{Au, Win}}$			Evergreen Shrub	2,700 - 3,200	1, 2, 4, 5, 6	
Cotoneaster verruculous						CS$_{\text{Au, Win}}$			Evergreen Shrub	2,800 - 3,250	1, 2, 4, 5, 6	
Laurocerasus dolichophylla			CS$^{\text{Sp}}$	CS$^{\text{Su, Au}}$	CS$^{\text{Sp}}$	CS$_{\text{Su, Au}}$	CS$^{\text{SP}}$		Evergreen Tree	2,310 - 3,050	1, 2, 4	
Malus sp.			CS$^{\text{Sp}}$						Deciduous Tree	2,400 - 2,600	1	
Padus napaulensis		CS$^{\text{Sp}}$	P$^{\text{Sp}}$	P$^{\text{Su}}$	CS$^{\text{Sp}}$	CS$^{\text{Su}}$	CS$^{\text{Sp}}$		Deciduous Tree	2,500 - 2,700	1, 2, 3-2	
Padus obtusata			P$^{\text{Sp}}$	CS$^{\text{Su}}$	CS$^{\text{Sp}}$	CS$_{\text{Su, Au}}$			Deciduous Tree	2,276 - 3,000	1, 2, 4	
Padus perulata		CS$^{\text{Sp}}$	CS$^{\text{Sp}}$	OS$^{\text{Su}}$	CS$^{\text{Sp}}$	CS			Deciduous	1,600 -	1, 2, 3-2	Field

						$^{Su,\ Au}$①			Tree	2,510		recorded
Photinia integrifolia		CS ^{Sp}	P $^{Sp,\ Su}$	OS $^{Sp,\ Su,\ Au,\ Win}$	CS ^{Su}	P ^{Au}②			Evergreen Tree or Epiphytic Shrub	1,860 - 2,800	1, 2	Field recorded
Prunus salicina		CS ^{Sp}	P ^{Sp}		CS ^{Sp}	CS ^{Sp}			Deciduous Tree	1,600 - 2,600	1, 2, 3-2, 3-3	
Sorbus aronioides			P ^{Sp}	CS ^{Su}	CS ^{Sp}	CS ^{Su}			Deciduous Tree	2,276 - 3,000	1, 4	
Sorbus Coronata			P ^{Sp}	CS ^{Su,}		CS ^{Au}			Deciduous Tree	2,350 - 2,970	1, 2, 3-2, 4	
Sorbus insignis		P ^{Sp}	P ^{Sp}	P ^{Su,}	CS ^{Su}①	P ^{Au}①	CS ^{Sp}		Deciduous Tree	2,700 - 3,100	1, 2, 3-1, 4	Field recorded
Sorbus oligodonta			P ^{Sp}	P ^{Su,}		CS ^{Au}			Deciduous Tree	2,650 - 3,600	1, 4, 5, 6	
Sorbus pteridophylla			P ^{Sp}	P ^{Su}	CS ^{Sp}	CS $^{Su,\ Au}$			Deciduous bush	2,181 - 3,100	1, 2, 3-2, 4	
Sorbus rhamnoides			P ^{Sp}	CS ^{Su}		CS ^{Au}			Deciduous Tree	1,793 - 3,100	1, 2, 3-1, 3-2, 4	
Sorbus thibetica			CS ^{Sp}	CS ^{Su,}		CS ^{Au}			Deciduous Tree	2,780 - 3,400	3-1, 3-2, 4, 5	
Sorbus vilmorinii			P ^{Sp}	P ^{Su}	CS ^{Sp}	CS $^{Su,\ Au}$	CS ^{Sp}		Deciduous bush	2,830 - 3,600	1, 2, 3-1, 3-2, 4, 6	
Dalbergia mimosoides	Papilionaceae		CS ^{Sp}						Deciduous Tree	1,600 - 2000	1, 2, 3-3	
Sarcococca wallichii	Buxaceae		CS ^{Sp}				CS ^{Sp}		Evergreen Shrub	1,823 - 2,700	1, 2, 3-2, 3-3	
Populus davidiana	Salicaceae		CS ^{Sp}	CS ^{Su}					Deciduous Tree	1,600 - 2,800	1, 2, 3-2, 3-3	
Populus rotundifolia var.			CS ^{Sp}	CS ^{Su}			CS ^{Sp}	CS $^{Sp\ ,\ Win}$	Deciduous Tree	2,530 - 2,900	1, 2, 3-2, 4	

duclouxiana												
Salix cathayana		CS ^{Sp}	CS ^{Sp}						Deciduous bush	2,800 - 3,300	1, 2, 4, 6	
Salix daliensis		CS ^{Sp}	CS ^{Sp}						Deciduous Tree	1,820 - 3,125	1, 2, 3-2, 3-3, 6	
Salix dibapha		CS ^{Sp}	CS ^{Sp}						Deciduous Tree	1,859 - 2,810	1, 2, 3-2; 3-3	
Salix wallichiana		CS ^{Sp}	CS ^{Sp}				CS ^{Sp}	CS $_{Sp, Win}$	Deciduous Tree	2,200 - 3,200	1, 2, 3-2, 4, 5, 6	
Betula alnoides	Betulaceae	CS ^{Sp}	CS ^{Sp}	CS $^{Sp, Su}$					Deciduous Tree	1,820 - 2,800	1, 2, 3-2, 3-3	
Betula cylindrostachya		CS ^{Sp}	CS ^{Sp}	CS $^{Sp, Su}$					Deciduous Tree	1,600 - 2,900	1, 2, 3-1, 3-2, 3-3	
Betula delavayi			P ^{Sp}	CS $^{Sp, Su}$					Deciduous Tree	2,500-2,650	1, 2, 3-1, 3-2, 4	
Betula luminifera		CS ^{Sp}	CS ^{Sp}	CS $^{Sp, Su}$					Deciduous Tree	2,700 - 2,900	1, 4	
Betula utilis		CS ^{Sp}	CS ^{Sp}	CS $^{Sp, Su}$					Deciduous Tree	2,530 - 3,000	1, 2, 3-1	
Carpinus monbeigiana			P ^{Sp}	CS ^{Su}					Deciduous Tree	1,800 - 2,600	1, 2, 3-3	
Corylus ferax		CS ^{Sp}	P ^{Sp}	CS ^{Su}	CS ^{Su}	P ^{Au,} $_{Win}$	CS ^{Sp}		Deciduous Tree	2,500 - 3,010	1, 2, 3-1, 3-2, 4	
Cyclobalanopsis gambleana	Fagaceae					CS ^{Au}			Evergreen Tree	1,600 - 2,718	1, 2, 3-2, 3-3	
Cyclobalanopsis glaucoides						CS ^{Au}			Evergreen Tree	1,990 - 2,750	1, 2, 3-2, 3-3	
Cyclobalanopsis lamellosa			OS ^{Sp}			CS $_{Au, Win}$			Evergreen Tree	2,250 - 2,600	1, 2, 3-2	
Lithocarpus hancei		CS ^{Sp}	CS ^{Sp}			CS ^{Au}			Evergreen Tree	2,310 - 2,800	1, 2, 3-2, 3-3	
Lithocarpus						CS			Evergreen	1,820 -	1, 2, 3-3	

hancei						$_{\text{Au, Win}}$			Tree	2,430		
Lithocarpus pachyphyllus		CS $^{\text{Sp}}$	CS $^{\text{Sp}}$			CS $_{\text{Au, Win}}$			Evergreen Tree	1,820 - 2,820	1, 2, 3-2	
Quercus engleriana						CS $^{\text{Au}}$			Evergreen Tree	2,700 - 2,800	1, 4	
Quercus griffithii						CS $^{\text{Au}}$			Evergreen Tree	1,600 - 2,600	1, 2, 3-3	
Ulmus microcarpus	Ulmaceae	CS $^{\text{Sp}}$	P $^{\text{Sp}}$	CS $^{\text{Su}}$					Deciduous Tree	2,500 - 2,800	1, 2	
Ilex micropyrena	Aquifoliaceae	CS $^{\text{Sp}}$	OS $^{\text{Sp}}$						Evergreen Shrub	2,450	1	
Ilex sikkimemsis		CS $_{\text{Sp①, Win}}$	P $^{\text{Sp①}}$	OS $^{\text{Sp, Su, Au, Win}}$	CS $^{\text{Su}}$	CS $_{\text{Au, Win}}$	CS $^{\text{Sp}}$		Evergreen Tree	2,600 - 3,000	1, 2, 4	Field recorded
Ilex yunnanensis		CS $^{\text{Sp}}$	CS $^{\text{Sp}}$	OS $^{\text{Sp, Su}}$	CS $^{\text{Sp}}$	CS $^{\text{Au}}$	CS $^{\text{Sp}}$		Evergreen Tree	2,500 - 3,000	1, 2, 3-2, 4	
Euonymus frigidus	Celastraceae		OS $^{\text{Sp}}$			CS $_{\text{Su, Au}}$			Deciduous Tree	1,600 - 3,450	1, 2, 3-1, 3-2, 4, 5, 6	
Dipentodon sinicus	Dipentodontaceae	CS $^{\text{Sp}}$	CS $^{\text{Sp}}$	CS $^{\text{Su}}$					Deciduous Tree	1,809 - 2,800	1, 2, 3-2	
Berchemia yunnanensis	Rhamnaceae		CS $^{\text{Sp}}$	CS $^{\text{Su}}$	CS $^{\text{Su}}$				Climbing Shrub	2,700 - 3,050	1, 4	
Skimmia arborescens	Rutaceae	CS $^{\text{Sp}}$	P $^{\text{Sp①}}$	P $^{\text{Sp①, Su, Au, Win}}$	CS $^{\text{Sp, Au, Win}}$	P $^{\text{Su, Au}}$	CS $^{\text{Sp, Win}}$	CS $^{\text{Sp, Win}}$	Evergreen Tree	2,300 - 3,100	1, 2, 3-1, 3-2, 4	Also fed petiole $^{\text{Sp, Su, Au, Win}}$; field recorded
Acer campbellii	Aceraceae	CS $^{\text{Sp}}$	CS $^{\text{Sp}}$			CS $^{\text{Su}}$			Deciduous Tree	2,200 - 3,000	1, 2, 4	
Acer davidii		CS $^{\text{Sp}}$	CS $^{\text{Sp}}$						Deciduous Tree	1,600 - 2,600	1, 2, 3-3	
Acer forrestii		CS $^{\text{Sp}}$	CS $^{\text{Sp}}$		CS $^{\text{Sp}}$				Deciduous	3,000 -	1, 4, 5, 6	

									Tree	3,500		
Acer hookeri		CS ^{Sp}	OS ^{Sp}			CS ^{Su}			Deciduous Tree	2,350 - 2,600	1, 2, 3-2	
Acer oliverianum		CS ^{Sp}	CS ^{Sp}						Deciduous Tree	2,300 - 2,800	1, 2	
Acer pectinatun		CS ^{Sp}	CS ^{Sp}						Deciduous Tree	2,700 - 3,400	1, 4	
Acer sikkimense var. *serrulatum*		CS ^{Sp}	CS ^{Sp}		CS ^{Sp}				Deciduous Tree	2,080 - 2,600	1, 2	
Acer taronense		CS ^{Sp}	CS ^{Sp}		CS ^{Sp}				Evergreen Tree	2,282 - 2,800	1, 2	
Acer wardii		CS ^{Sp}	P ^{Sp}	OS ^{Su}		CS ^{Su}	CS ^{Sp}		Deciduous Tree	2,500 - 3,000	1, 2, 4	
Acer sp.		CS ^{Sp}	CS ^{Sp}						Deciduous Tree	2,510 - 2,850	1, 2	
Acer sp.			CS ^{Sp}						Deciduous Tree	2,900	4	
Sabia parviflora	Sabiaceae	P ^{Sp}	P ^{Sp}	P $^{Su\ Au}$	CS ^{Sp}	CS $^{Su,\ Au}$	CS ^{Sp}		Climbing Shrub	1,600 - 2,930	1, 2, 3-1, 3-2, 3-3, 4	
Sabia yunnanensis		P ^{Sp}	P ^{Sp}	P $^{Sp,\ Su\ Au,\ Win}$	CS ^{Sp}	CS $^{Su,\ Au}$	CS ^{Sp}		Climbing Shrub	1,823 - 3,197	1, 2, 3-1, 3-2, 3-3, 4, 6	
Helwingia himalaica	Helwingiaceae		P ^{Sp}	CS $^{Su,\ Au,\ Win}$	CS ^{Sp}	CS ^{Su}	CS ^{Sp}	CS $^{Sp,\ Win}$	Evergreen bush	1,600 - 3,000	1, 2, 3-1, 3-2, 3-3, 4	
Helwingia japonica			P ^{Sp}	CS $^{Su,\ Au,\ Win}$	CS ^{Sp}	CS ^{Su}	CS ^{Sp}	CS $^{Sp,\ Win}$	Evergreen bush	1,823 - 2,600	1, 2, 3-3	
Gamblea ciliata	Araliaceae	CS Sp①	P ^{Sp}①	P $^{Su,\ A}$①	CS ^{Su}	P Au①	CS ^{Sp}		Deciduous Tree	2,650 - 3,400	1, 2, 3-1, 4, 5	Also fed petiole $^{Sp,\ Su}$; field recorded
Merrilliopanax listeri		CS ^{Sp}	P ^{Sp}	CS $^{Su,\ Au,\ Win}$	CS ^{Su}		CS ^{Sp}	CS ^{Sp}	Evergreen Shurb	2,050 - 2,800	1, 2	Also fed petiole $^{Sp,\ Su,\ Au,\ Win}$;

												backup food
Pentapanax racemosus		P^{Sp}	P^{Sp}	P$^{Su, Au}$					Epiphytic Tree or Deciduous Tree	2,250 - 2,800	1, 2, 3-2	Also fed petiole$^{Sp, Su}$
Pentapanax yunnanensis		P^{Sp}	P^{Sp}	P$^{Su, Au}$					Deciduous Shrub	2,450	1	Also fed petiole$^{Sp, Su}$
Schefflera elliptica		CS^{Sp}	CS^{Sp}	CS$^{Au, Win}$			CS^{Sp}		Evergreen Tree	2,500 - 2,600	1	Also fed petiole$^{Sp, Win}$ and stem^{Win}
Schefflera glabrescens		CS^{Sp}	P^{Sp}	CS$^{Au, Win}$			P$^{Sp, Su, Au}$	CS^{Win}	Evergreen Tree	2,600 - 2,900	1, 4	Also fed petiole$^{Sp, Win}$ and stem^{Win}
Schefflera hoi				CS$^{Au, Win}$			CS^{Win}	CS^{Win}	Evergreen Tree	2,600 - 2,800	1, 2	Also fed petiole^{Win} and stem^{Win}
Schefflera shweliensis		CS^{Sp}	CS^{Sp}				P$^{Sp, Su, Au}$	CS^{Win}	Evergreen Tree	1,900 - 2,800	1, 2	Also fed petiole$^{Sp, Win}$ and stem^{Win}
Lyonia doyonensis	Ericaceae		CS^{Sp}						Deciduous Tree	2,510 - 3,125	1, 2, 3-1, 4, 6	
Lyonia ovalifolia			CS^{Sp}						Deciduous Tree	1,809 - 3,020	1, 2, 3-1, 3-2, 3-3, 4, 5	
Rhododendron arizelum					CS^{Sp}				Evergreen Tree	2,700 - 3,600	1, 3-1, 4, 6	
Rhododendron ciliipes					P^{Sp}				Epiphytic Evergreen	2,253-2,810	1, 2, 4	

									Shurb			
Rhododendron decorum			CS ^{Sp}		CS ^{Sp}				Evergreen Tree	1,950 - 2,850	1, 2, 3-2, 4	
Rhododendron edgeworthii					P ^{Sp}				Epiphytic Evergreen Shurb	2,737 - 3,100	1, 2, 4	
Rhododendron facetum					CS ^{Sp}				Evergreen Shrub	2,130 - 2,720	1, 2	
Rhododendron monanthum					CS $^{Sp, Su, Au}$				Epiphytic Evergreen Shurb	2,700 - 3,000	1, 4	
Rhododendron neriiflorum					CS ^{Sp}				Evergreen Shrub	(2,450) 2,800 - 3,200	1, 2, 4, 5, 6	
Rhododendron protistum var. *gigantewn*					CS ^{Win}				Evergreen Tree	2,450 - 2,652	1	
Rhododendron rubiginosum					P ^{Sp}				Evergreen Tree	2,770 - 3,160	1, 3-1, 4, 6	
Rhododendron sidereum					CS $^{Sp, Su}$				Evergreen Tree	2,700 - 3,310	1, 2, 4, 6	
Rhododendron sinogrande					CS ^{Sp}				Evergreen Tree	2,480 - 3,183	1, 3-1, 4, 6	
Rhododendron sulfureum					CS ^{Sp}				Epiphytic Evergreen Shurb	2,713 - 3,000	1, 4	
Rhododendron tephropeplum					CS ^{Sp}				Evergreen Shrub	2,700- 3,000	1, 4	
Rhododendron zaleucum			CS $^{Sp, Su}$	OS $^{Sp, Su}$	P ^{Sp}				Evergreen Tree	2,710 - 3,250	1, 2, 4, 6	
Agapetes mannii	Vacciniaceae		CS ^{Sp}			CS ^{Au}	CS ^{Sp}		Epiphytic Evergreen	2,600 - 2,800	1	

									Shurb			
Vaccinium delavayi			CS$^{\text{Sp}}$		CS$^{\text{Sp}}$	CS$^{\text{Au, Win}}$	CS$^{\text{Sp}}$		Epiphytic Evergreen Shurb	2,700 - 3,100	1, 4	
Vaccinium bulleyanum			CS$^{\text{Sp}}$		CS$^{\text{Sp}}$	CS$^{\text{Sp, Su}}$	CS$^{\text{Sp}}$		Epiphytic Evergreen Shurb	1,930 - 2,800	1, 2, 3-1, 3-2, 4	
Vaccinium dendrocharis			CS$^{\text{Sp}}$		CS$^{\text{Sp}}$	CS$^{\text{Au, Win}}$	CS$^{\text{Sp}}$		Epiphytic Evergreen Shurb	2,633 - 3,100	1, 3-1, 4	
Vaccinium leucobotrys			CS$^{\text{Sp}}$		CS$^{\text{Sp}}$	CS$^{\text{Sp, Su}}$	CS$^{\text{Sp}}$		Epiphytic Evergreen Shurb	2,100 - 2,800	1, 3-1, 4	
Styrax perkinsiae	Styracaceae		CS$^{\text{Sp}}$	CS$^{\text{Sp, Su,}}$	CS$^{\text{Sp}}$				Deciduous Tree	1,850-3,200	1, 4	
Symplocos dryophlia	Symplocacea	CS$^{\text{Sp, Win}}$	P$^{\text{Sp}}$	CS$^{\text{Sp, Su, Au, Win①}}$	CS$^{\text{Sp, Su}}$	P$^{\text{Au}}$	CS$^{\text{Sp}}$	CS$^{\text{Sp, Win①}}$	Evergreen Tree	2,200 - 3,127	1, 2, 3-2; 4, 5, 6	Field recorded
Symplocos theaefolia		CS$^{\text{Sp}}$	CS$^{\text{Sp}}$				CS$^{\text{Sp}}$		Evergreen Tree	2,600 - 3,000	1, 2, 4	
Ligustrum confusum	Oleaceae	P$^{\text{Sp}}$	P$^{\text{Sp}}$	CS$^{\text{Sp, Su, Au, Win}}$	CS$^{\text{Su}}$	P$^{\text{Au}}$			Evergreen Tree	1,809 - 2,746	1, 2, 3-2	
Ligustrum delavayanum		P$^{\text{Sp}}$	CS$^{\text{Sp}}$	OS$^{\text{Su}}$	CS$^{\text{Su}}$	P$^{\text{Au}}$		CS$^{\text{Sp, Win}}$	Evergreen Shrub	1,640-2,800	1, 2, 3-2, 3-3	
Ligustrum sinense		P$^{\text{Sp}}$	P$^{\text{Sp}}$	CS$^{\text{Sp, Su, Au, Win}}$	CS$^{\text{Su}}$	P$^{\text{Au}}$		CS$^{\text{Sp, Win}}$	Evergreen Tree	2,400 - 2,800	1, 2	
Leycesteria gracilis	Caprifoliaceae		CS$^{\text{Sp, Su}}$	CS$^{\text{Su, Au}}$	CS$^{\text{Au}}$	OS$^{\text{Su}}$			Deciduous bush	2,080 - 2,800	1, 2, 3-3	
Leycesteria formosa			CS$^{\text{Sp, Su}}$	CS$^{\text{Su, Au}}$					Deciduous bush	1,809 - 3,063	1, 2, 3-2, 3-3, 4	
Lonicera acuminata		CS$^{\text{Sp, Su}}$	CS$^{\text{Sp}}$	CS$^{\text{Su, Au}}$	CS$^{\text{Su}}$	CS$^{\text{Au}}$			Climbing Shrub	2,310 - 3,100	1, 2, 3-2, 4	
Lonicera nigra		CS$^{\text{Sp}}$	CS$^{\text{Sp}}$	CS$^{\text{Su}}$	CS$^{\text{Sp, Su}}$	CS			Deciduous	2,400 -	1, 2, 3-1, 3-	

						Su, Au			bush	3,400	2, 4, 5, 6	
Lonicera japonica		CS $^{Sp, Su}$	CS ^{Sp}	CS $^{Su, Au}$	CS ^{Su}	CS ^{Au}			Climbing Shrub	2,440 - 3,063	1, 2, 3-2, 4	
Senecio scandens	Asteraceae	CS $^{Sp, Su, Au}$	P $^{Sp, Su, Au}$	CS $^{Su, Au, Win}$	CS $^{Su, Au}$		CS $^{Sp, Su}$	CS ^{Win}	Perennial Herb	1,700 - 2,950	1, 2, 3-1, 3-2, 3-3, 4	
Polygonatum cathcartii	Liliaceae		CS ^{Sp}	CS $^{Su, Au}$	CS ^{Sp}	CS ^{Su}			Perennial Herb	2,400 - 3,210	1, 2, 4	
Polygonatum punctatum			CS ^{Sp}	CS $^{Su, Au}$	CS ^{Sp}	CS ^{Su}			Epiphytic Herb	2,170 - 2,650	1, 2	
Streptopus simplex			P ^{Sp,}	CS $^{Su, Au}$	CS ^{Su}	CS $^{Su, Au}$			Perennial Herb	2,700 - 3,600	1, 3-2, 4, 5, 6	
Smilax menispermoidea	Smilacaceae		CS ^{Sp}	CS $^{Sp, Su, Au}$	CS $^{Sp, Su}$	CS ^{Su}			Climbing Shrub	2,900 - 3,250	4, 5, 6	
Carex nitidiutriculata	Cyperaceae		CS ^{Sp}						Perennial Herb	1,859 - 3,007	1, 2, 3-1, 3-2, 3-3, 4	Also fed rootstock ^{Sp}
Dendrobium hookerianum	Orchidaceae				CS ^{Su}	CS $^{Su, Au}$			Epiphytic orchid	1,600 - 2,300	1, 2	
Coelogyne pianmaensis					CS ^{Sp}	CS $^{Su, Au}$			Epiphytic orchid	2,600 - 3,130	1, 4	Also fed pseudobulbs $^{Sp, Su, Au, Win}$
Coelogyne corymbosa					CS ^{Sp}	CS $^{Su, Au}$			Epiphytic orchid	1,600 - 3,100	1, 4	Also fed pseudobulbs $^{Sp, Su, Au, Win}$
Coelogyne occultata					CS ^{Sp}	CS ^{Au}			Epiphytic orchid	1,900 - 2,400	1	Also fed pseudobulbs $^{Sp, Su, Au, Win}$

An epiphytic monocot	Unidentified			CS Au①					Epiphytic Herb	2,800 - 2,900	4	Field recorded
Other type of food items												
Fargesia contracta	Bambusoideae	Bamboo Shoots (preferred food, occurred from late May to early June)							Perennial Herb	2,700 - 3,000	1, 2, 3-1, 3-2, 3-3, 4, 5	
Fargesia orbiculata		Bamboo Shoots (preferred food, occurred early to mid- June)							Perennial Herb	3,150 - 3,600	4, 5, 6	
Fargesia papyrifera		Bamboo Shoots (occurred from mid- May to early June)							Perennial Herb	2,500 - 3,180	1, 2, 3-1, 3-2, 4	
Usnea floria	Usneaceae	Entire fungi (preferred food, occurred year around)①							Epiphytic fungi	2,503 - 3,080	1, 2, 3-1, 3-2, 3-3, 4, 5, 6	Field recorded
Usnea cavernosa		Entire fungi (occurred year around)							Epiphytic fungi	2,600 - 3,166	1, 4	
Usnea comosa		Entire fungi (occurred year around)							Epiphytic fungi	2,732 - 3,000	1, 4, 6	
Usnea bismolliuscula		Entire fungi (occurred year around)							Epiphytic fungi	2,700	1	
Usnea himantodes		Entire fungi (occurred year around)							Epiphytic fungi	2,800	1	
Usnea sp.		Entire fungi (occurred year around)							Epiphytic fungi	2,715	1	
Ramalina pollinaria	Ramalinaceae	Entire fungi (preferred food, occurred year around)							Epiphytic fungi	2,650	1	
Ramalina conduplicans		Entire fungi (preferred food, occurred year around)①							Epiphytic fungi	2,503 - 3,063	1, 3-1, 3-2, 3-3, 4	Field recorded
Ramalina roesleri		Entire fungi (preferred food, occurred year around)							Epiphytic fungi	2,715	1	
Ramalina sinensis		Entire fungi (preferred food, occurred year around)①							Epiphytic fungi	2,300 - 2,800	1, 2, 3-3	
Dolichousnea longissima	Pameliaceae	Entire fungi (occurred year around)							Epiphytic fungi	2,800 - 3,200	4	Field recorded

Nephromopsis nephromoides		Entire fungi (preferred food, occurred year around)	Epiphytic fungi	2,710	1	
Nephromopsis ornata		Entire fungi (preferred food, occurred year around)	Epiphytic fungi	2,580 - 2,850	1, 4	
Nephromopsis pallescens		Entire fungi (preferred food, occurred year around)	Epiphytic fungi	2,500 - 2,600	1, 2	
Nephromopsis stracheyi		Entire fungi (preferred food, occurred year around)①②	Epiphytic fungi	2,400 - 2,900	1, 2, 4	Field fecorded
Cnipsus colorantis	Phasmatodea	Stick insect (can be found from May to August)	Arthropods	-	-	
Ramulus sp.		Stick insect (can be found from May to October)	Arthropods	-	-	
Mirollia sp.	Tettigoniidae	Katydid (can be found from April to October)	Arthropods	-	-	

Note: The Latin name order of seed plants is according to the Hutchinson systems (1973 revised edition).

①: observed in Luoma MLS on east slopes of the Gaoligong Mountain.

②: observed in Pianma MLS on west slopes of the Gaoligong Mountain (Chen Yixin, pers. commu.).

NS = None selected means that the captive snub-nosed monkeys tried to bite 1-3 times but then discarded the plant species or plant parts and never tried it after , or they never consumed the plant species or part. .

OS = Occasionally selected means that when the plant species or plant parts appeared, the captive snub-nosed monkeys were recorded to consume the food several times but very less amount.

CS = Consistently selected means that whenever the plant species or plant parts appeared, the captive snub-nosed monkeys tried to consume the food as possible as they can.

P = Preference means that when presented with two different plant species or plant parts, the captive snub-nosed monkeys dependably chose one over the other.

Table Appendix VI. A comparison of babitat characteristics and dietary pattern of *Rhinopithecus* spp. at different study sites

Rhinopithecus	Study site	MLS size	Altitude range (m)	Habitat type	Diversity of Seed Plant Flora/Tropical Elements (%)	Food Plant Diversity/Evergreen Plants in Food Species (%)	The proportion of diet composition (%)						Other food items	Food selection varied according to seasonality and local availability	Reference
							Young leaves / Mature leaves	Fruits/Seeds	Flowers	Buds	Bark	Lichens			
R. avunculus	Khau Ca	22-81	600-1,300	Tropical evergreen broadleaf forests	NA	50 species in 25 families/62	46.2/6.9 6.7:1	25/7.2	11.2	-	1.5	0	2	Young leaves (year-round); Flowers (February - May); Fruits & seeds (October - January)	Dong, 2012
R. brelichi	Fanjing Mountain	40-160 †	1,300-2,000	Subtropical evergreen broadleaf forest; Mixed evergreen & deciduous broadleaf forest	2,584 species in 163 families/58.38	107 species in 28 families/40	25.5/21.8 1.17:1	21.6	9.4	15.3	Recorded in winter	0	Bamboo shoots & invertebrates	Buds (September - January); Young leaves (February - May); Mature leaves (May - November); Flowers (March - April); Fruits & seeds (June - October)	Wu et al. 2016; Xiang et al. 2012
R. roxellana	Shennongjia	236 ± 38	1,550-2,663	Temperate deciduous broadleaf forests	2,439 species in 163 families/28.7	198 species in 43 families/20.7	13.5/8.9 1.51:1	9.5/20.8	1.3	5.8	0.2	38.4	Insects	Flowers (March - April); Young leaves (April - July); Mature leaves (May - September), Fruits (June - October); Seeds (September - March); Buds (December - April); Fruticose lichens (year-round)	Zheng et al. 1993; Liu et al. 2013; Li et al. 2015
	Zhouzhi East	112	1,400-2,900	Temperate deciduous broadleaf forest; Deciduous broadleaf & coniferous mixed forest	1,131 species in 109 families/14.4	84 species in 29 families/Not avaliable	24.0	29.4	–	4.2	11.1	29.0	unidentified	Lichen & seeds (winter); Lichen, barks, buds & young leaves (spring); Seeds & mature leaves (summer); Lichen & seeds (winter).	Yuan et al. 2007; Guo et al., 2007
	Zhouzhi West	232	1,500-3,400	Temperate deciduous broadleaf forest; Deciduous broadleaf & coniferous mixed forests	1,131 species in 109 families/14.4	119 species in 40 families/ 8.4	35.92	21.42	7.52	18.44	8.07	0.95	Invertebrates, fungi & clay	Young leaves & flowers (spring); Mature leaves (summer); Fruits (fall-winter); Buds & barks (winter)	Yuan et al. 2007; Huang 2015

	Qingmuchuan	100-120	800-2,054	Temperate evergreen & deciduous broadleaf forest; Deciduous broadleaf forest	1,467 species in 146 families/29.1	42 species in 23 families/16.7	25.0	25.94	–	21.3		0	–	Frugivorous (summer) - Folivorous (winter)	Jiang 2005; Li et al., 2010; Li et al., 2013
R. bieti	Xiaochangdu	210	3,500-4,250	Conifer forest	NA	25 species in 13 families/16.7	12.1	1.1	1.1	–[c]	4.2	82.1	Invertebrates, resin, & herbs	Lichens (year-round); Buds & leaves (March - August); Flowers (spring); Fruits (summer)	Xiang et al., 2007
	Wuyapiya	175	3,300-4,600	Conifer forest	NA	22 species in ≥ 12 families/13.6	6.0	0.1	Occasionally recorded	–	Occasionally recorded	85.0	–	Lichens (year-round); Buds & leaves (spring - summer); Fruits (summer)	Kirkpatric 1996
	Xiangguqing	480 ‡	2,600-4,100	Deciduous broadleaf & coniferous mixed forests	1,674 species of 135 families/30.3	92 species in 37 families/19.6	8.4/16.3 0.51:1	10.5	1.9	3.0	0.8	50.6	Bamboo shoots (7.9%), petiole, stem, fungi, bird and bird eggs & insects	Young leaves (spring); Bamboo shoot & mature leaves (summer); Fruits & seeds (fall); Lichens & buds (winter)	Li 2003; Li 2010
	Samage	410 ‡	2,500-4,000	Deciduous broadleaf & coniferous mixed forests	1,674 species of 135 families/30.3	94 species in 38 families/21.3	12.4/4.1 3.02:1	11.4	0.2	3.6	0.5	67	Petiole, Pith, bamboo shoots, mushrooms, tubers, bird eggs, &flying squirrel	Bamboo shoot (June - July); Lichen and leaves (year-round); Fruits (summer, fall, & winter); Buds (winter).	Li, 2003; Grueter et al. 2009a, b
	Longma Mt	80	2,700–3,200	Deciduous broadleaf & coniferous mixed forests	1,041 species in 148 families/34.2	97 species in 27 families/24.7	NA	NA	NA	NA	NA	NA	Bamboo shoots	Suggested buds, young leaves, flowers (spring); mature leaves (summer), leaves and fruits/seeds (fall-winter)	Huo 2005; Hua 2013
	Fuhe[§]	80	2,700-3,400	Deciduous broadleaf & coniferous mixed forests	NA	NA	50. 1	38.1 (summer-autumn)	–	–	–	5.5 (63% in feeding time)	Bamboo leaves & shoots (24.8 %, winter-	Unknown	Liu et al., 2004

													spring)		
R. strykeri	Gaoligong Mt.	70	1,700-3,200	Subtropical evergreen broadleaf forest; Coniferous & broadleaf mixed forest	4,303 Species in 210 families/50.8	Suggested 170 species in 41 families/43.5	NA	NA	NA	NA	NA	NA	Petiole, bamboo shoots, pseudobulbs, rootstocks & invertebrates	Suggested buds & young leaves (spring); Mature leaves (summer); Fruits/Seeds (summer - fall); Mature leaves (winter)	Li et al., 2000; this study

Note: NT = Not Available

†. *R. brelichi* usually maintained a MLS size of 40-160 individuals, although there is evidence that two MLS may temporarily merge into a larger social unit (Guo et al., 2017) .

‡. A recent study indicates that the large size of this MLS, > 400 individuals, represents two independent MLS with overlapped home ranges (Ren et al., 2016)

§. Using micro-histological analysis of feces, not recorded as part of feeding time.

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Table Appendix VII. Nutritional properties of spring and autumn food items and plant species consumed by captive *Rhinopithecus strykeri* in the Gaoligong Mountains Wildlife Rescue Centre.

Name	Item Position	Mo	AW	CP	CL	CA	CF	TNC	Ca	P	ADF	NDF	ADL	ME	Type
Acer oliverianum	BUD	78.8	8.17	26.75	1.77	5.3	25.26	37.67	0.05	0.56	18.93	28.51	2.62	1,379.55	T
Acer wardii	BUD	82.2	8.36	28.14	2.63	5.14	9.18	30.22	0.04	0.59	16.42	33.87	4.14	1,364.04	T
Betula luminifera	BUD	73.9	7.84	23.46	9.24	5.29	11.21	32.91	0.19	0.49	26	29.1	24.57	1,529.16	T
Dipentodon sinicus	BUD	79.8	10.9	27.52	1.59	4.76	6.89	55.01	0.17	0.55	17.04	11.12	3.31	1,506.91	T
Magnolia campbellii	BUD	83.3	8.52	29.78	3.05	5.41	34.66	22.6	0.17	0.64	42.61	39.16	32.22	1,333.15	T
Sabia japonica	BUD	87	9.02	34.9	1.38	6.83	11.73	16.72	0.07	0.7	22.61	40.17	4.54	1,273.41	L
Schisandra rubriflora	BUD	81	9.51	27.33	3.43	6.53	19.08	33.03	0.16	0.43	39.43	29.68	30.69	1,387.12	L
Sorbus coronata	BUD	75.9	8.75	20.63	4.4	5.72	59.4	40.97	0.18	0.47	26.18	28.28	24.28	1,423.55	T
Sorbus insignis	BUD	77	10.03	17.68	1.07	5.36	12.94	47.71	0.2	0.18	31.62	28.18	10.38	1,357.13	T
Tetracentron sinense	BUD	75	13.05	16.73	6.84	4.56	20.51	25.97	0.22	0.4	45.07	45.9	24.25	1,366.11	T
Ulmus microcarpa	BUD	83.5	10.33	27.09	1.4	7.38	35.71	25.31	0.22	0.62	45.2	38.82	33.4	1,266.67	T
Lithocarpus pachyphyllus	BUD	73	8.69	18.28	10.1	4.24	11.59	49.47	0.19	0.55	20.62	17.91	10.47	1,637.17	T
Manglietia insignis	BUD	83	10.14	23.64	0.97	4.69	11.65	31.33	0.21	0.28	52.85	39.37	20.04	1,293.38	T
Symplocos dryophila	BUD	82.2	8.86	19.45	0.95	7.81	29.41	36.36	0.19	0.47	43.02	35.43	28.15	1,266.43	T
Actinidia kungshanensis	ML	74.5	7.74	18.05	2.07	6.01	23.35	35.08	0.26	0.22	34.05	38.79	10.98	1,293.83	L
Actinidia pilosula	ML	65.9	8.26	16.54	5.37	5.63	51.66	44.23	0.28	0.14	19.72	28.23	10.92	1,441.62	L
Actinidia venosa	ML	65.9	9.45	13.78	3.73	8.72	7.65	10.51	0.61	0.06	23.57	63.26	9.71	1,109.77	L

Betula luminifera	ML	65.8	8.73	19.04	5.46	4.79	16.52	35.28	0.22	0.24	30.41	35.43	10.99	1,409.05	T
Clematis clarkeana var. stenophylla	ML	79.2	8.8	24.93	2.84	9.79	19.63	20.69	0.31	0.24	28.71	41.75	5.78	1,235.68	H
Gamblea ciliata	ML	77.6	7.62	14.39	4.73	5.2	57.04	32.28	0.71	0.83	28.31	43.4	7.73	1,326.72	T
Helwingia himalaica	ML	77.4	7.05	14.7	5.39	8.23	23.33	44.95	0.3	0.17	30.63	26.73	7.39	1,409.1	H
Ligustrum delavayanum	ML	64.2	6.94	19.22	2.19	4.04	11.52	29.92	0.29	0.18	28.83	44.63	15.93	1,288.16	T
Litsea cubeba	ML	65.5	5.64	13.38	14.17	4.41	19.09	32.01	0.22	0.34	18.81	36.03	11.13	1,588.18	T
Litsea rubescens	ML	68.4	6.99	24.39	5.33	6.41	16.26	26.18	0.29	0.23	29.94	37.69	13.87	1,369.93	T
Lonicera acuminata	ML	69.4	6.13	15.91	3.89	5.55	13.49	51.9	0.25	0.14	33.9	22.75	16.88	1,449.55	L
Lonicera maackii	ML	70.4	9.36	12.92	2.82	6.87	10.92	20.03	0.41	0.12	32.45	57.36	19.18	1,160.7	L
Magnolia campbellii	ML	78.1	8.04	11.05	4.06	7.99	35.54	33.3	0.2	0.15	32.63	43.6	9.62	1,263.27	T
Polygala fallax	ML	75.9	8.31	24.18	3.22	4.35	15.76	21.72	0.32	0.27	26.44	46.53	8.84	1,296.49	T
Sabia parviflora	ML	78	7.76	24.75	2.75	7.99	16.52	27.48	0.29	0.27	19.3	37.03	4.75	1,295.45	L
Schisandra neglecta	ML	79.6	8.07	21.27	5.82	7.64	27.89	35.43	0.26	0.27	19.55	29.84	4.61	1,412.37	L
Senecio scandens	ML	80.6	8.75	28.27	3.15	8.42	9.38	11.56	0.24	0.37	19.43	48.6	6.62	1,219.5	H
Sorbus insignis	ML	66.7	7.7	7.78	2.46	6.93	17.81	43.29	0.27	0.08	31.92	39.54	18.07	1,271.78	T
Sorbus oligodonta	ML	68.9	8.93	23.37	1.64	7.03	12.8	47.69	0.29	0.12	27.97	20.27	14.05	1,403.44	T
Helwingia japonica	ML	75.4	10.35	11.74	4.12	10.86	12.22	4.1	0.61	0.14	29.97	69.18	15.03	1,040.24	H
Holboellia fargesii	ML	70.2	9.06	10.5	2.4	7.76	15.71	42.46	0.35	0.135	31.1	36.88	9.11	1,278.58	L
Machilus rufipes	ML	61.4	8.07	9.15	1.47	3.79	37.64	37.06	0.29	0.09	48.53	61.09	24.56	1,352.71	T
Machilus salicina	ML	63.5	8.04	9.52	1.86	4.26	44.3	48.49	0.37	0.14	35.87	55.21	23.23	1,503.39	T
Merrilliopanax chinensis	ML	68.1	10.22	11.48	3.01	7.5	59.9	32.33	0.33	0.11	52.68	45.68	15.55	1,234.93	S
Schefflera actinophylla	ML	67.3	7.98	9.67	3.44	5.37	25.76	40.65	0.3	0.11	39.69	40.87	21.07	1,309.9	T

Skimmia arborescens	ML	64.9	7.38	11	4.62	8.28	35.61	37.94	0.43	0.13	33.6	38.16	13.28	1,308.42	T
Acer oliverianum	YL	77	9.21	18.23	2.07	3.93	52.39	60.27	0.19	0.26	19.77	15.5	12.44	1,490.32	T
Acer pectinatum	YL	78.1	9.87	15.67	1.3	4.76	23.31	48.09	0.21	0.25	24.47	30.18	13.23	1,355.55	T
Acer wardii	YL	84.5	8.88	30.31	1.95	6.68	40.58	41.79	0.19	0.57	23.28	19.27	5.57	1,429.49	T
Actinidia kolomikta	YL	77	8.82	23.72	2.07	5.47	23.78	29.91	0.18	0.54	18.74	38.83	17.11	1,307.86	L
Actinidia kungshanensis	YL	86.85	10.12	28.36	2.71	9.23	12.02	24.75	0.18	0.53	34.7	34.95	10.43	1,292.94	L
Actinidia pilosula	YL	82.8	11.45	33.49	2.6	9.38	33.78	31.87	0.23	0.86	31.05	22.66	10.45	1,379.39	L
Actinidia venosa	YL	87	9.49	28.31	3.02	8.53	11.49	23.73	0.12	0.57	24.78	36.41	3.61	1,300.38	L
Betula cylindrostachya	YL	74.8	8.99	29.73	4.87	3.39	40.5	35.74	0.2	0.55	33.21	26.27	21.17	1,493.87	T
Betula luminifera	YL	75.2	9.01	23.91	2.74	4.91	11.48	17.93	0.08	0.43	25.51	50.51	7.19	1,249.32	T
Carpinus monbeigiana	YL	77	9.58	21.61	2.89	5.63	44.03	39.07	0.21	0.3	50.54	30.8	19.2	1,376.62	T
Cerasus caudata	YL	78.3	9.19	30.73	4.05	6.81	9.57	22.65	0.1	0.61	19.98	35.76	8.49	1,356.5	T
Cerasus clarofolia	YL	78.6	8.36	21.98	2.38	7.8	11.22	46.23	0.23	0.45	24.26	21.61	33.1	1,395.89	T
Dipentodon sinicus	YL	77.7	8.98	22.62	2.31	4.14	10.1	34.47	0.07	0.27	15.26	36.46	3.09	1,349.67	T
Hydrangea sp.	YL	86.6	9.91	23.11	1.45	9.78	18.79	23.44	0.13	0.38	23.65	42.22	6.55	1,201.54	S
Laurocerasus dolichophylla	YL	69.5	9.25	17.88	2.56	4.24	13.36	38.35	0.08	0.22	22.42	36.97	6.43	1,345.01	T
Leycesteria gracilis	YL	76.2	9.36	23.35	2.43	5.59	18.48	48.99	0.19	0.51	27.89	19.64	7.65	1,447.46	S
Lindera obtusiloba var. heterophylla	YL	78.1	8.88	25.33	4.37	4.96	15.4	22.11	0.03	0.71	34.05	43.23	5.54	1,335.13	T
Litsea cubeba	YL	75.5	9.03	28.31	2.05	4.38	19.04	49.51	0.17	0.55	31.2	15.75	6.71	1,491.03	T
Litsea rubescens	YL	79.5	11.7	27.44	3.73	4.7	12.06	43.8	0.17	0.48	40.48	20.33	9.52	1,488.26	T
Lonicera maackii	YL	77.6	10	23.37	1.38	6.69	8.45	36.19	0.18	0.32	43.81	32.37	17.9	1,318.72	L

Lyonia doyonensis	YL	79.3	8.99	24.16	3.07	4.75	13.64	26.33	0.14	0.53	31.48	41.69	10.19	1,320.8	T
Magnolia campbellii	YL	84.5	8.59	30.86	3.94	5.71	14.62	14.39	0.03	0.5	27.13	45.1	8.93	1,306.54	T
Padus napaulensis	YL	77.1	9.33	27.23	3.26	5.19	45.33	39.79	0.2	0.4	39.69	24.53	17.12	1,440.94	T
Pentapanax leschenaultii var.forrestii	YL	85	9.29	20.96	3.02	7.75	25.7	45.38	0.22	0.62	30.88	22.89	6.32	1,400.15	T
Pentapanax yunnanensis	YL	85.4	11.2	13.25	1.81	6.49	8.03	55.87	0.28	0.21	40.26	22.58	9.3	1,389.36	T
Polygala fallax	YL	82.4	10.98	30.73	2.94	2.39	43.52	38.97	0.2	0.77	53.39	24.97	7.24	1,479.44	T
Populus davidiana	YL	81.3	8.58	25.17	3.5	4.88	23.41	35.49	0.17	0.41	39.97	30.96	23.52	1,403.87	T
Populus rotundifolia var. duclouxiana	YL	73.9	8.55	22.76	7.32	5.52	13.37	10.81	0.08	0.38	37.76	53.59	18.41	1,313.03	T
Prunus salicina	YL	76.7	7.93	28.81	2.54	5.9	8.82	48.99	0.21	0.59	18.44	13.76	7.08	1,491.43	T
Sabia japonica	YL	82.7	9.98	21.74	1.8	6.91	47.56	45.11	0.2	0.45	29.53	24.44	17.3	1,377.9	L
Sabia parviflora	YL	82.3	9.39	24.03	1.45	6.41	14.86	35.83	0.22	0.35	31.88	32.28	20.27	1,325.96	L
Salix balfouriana	YL	75.7	7.88	21.52	1.89	6.02	17.7	39.04	0.21	0.51	33.02	31.53	25.66	1,344.18	T
Salix daliensis	YL	83.3	9.01	29.62	3.94	8.24	13.52	24.39	0.13	0.56	28.78	33.81	5.3	1,343.85	T
Salix wallichiana	YL	77.7	7.78	23.58	1.87	5.72	19.3	17.95	0.04	0.42	36.22	50.88	9.85	1,215.17	T
Schisandra grandiflora	YL	86.5	9.98	17.75	3.71	7.31	12.61	20.28	0.07	0.52	32.44	50.95	8.95	1,222.05	L
Schisandra neglecta	YL	83.4	11.18	18.9	3.07	7.31	10.16	48.94	0.23	0.37	26.3	21.78	8.96	1,413.95	L
Senecio scandens	YL	86	10.84	28.42	3.49	9.06	10.17	27.53	0.15	0.48	15.27	31.5	3.03	1,336.25	H
Sorbus aronioides	YL	77.4	11.49	18.9	1.42	5.94	12.01	49.43	0.2	0.35	36.74	24.31	11.15	1,383.51	T
Sorbus coronata	YL	73.9	8.21	17.73	2.11	5.46	3.06	28.83	0.06	0.8	40.16	45.87	11.29	1,253.59	T
Sorbus insignis	YL	80.4	10.85	13.62	2.09	9.7	12.62	51.17	0.34	0.2	25.24	23.42	8.28	1,338.37	T
Sorbus rhamnoides	YL	79	6.4	25.14	2.64	7.76	18.27	36.75	0.22	0.55	30.46	27.71	16.51	1,362.45	T
Sorbus thibetica	YL	83	9.45	26.54	3.44	6	12.95	19.57	0.45	0.38	30.3	44.45	7.32	1,291.63	T

Sorbus vilmorinii	YL	77.6	9.99	18.44	2.26	8.54	12.03	46.79	0.23	0.47	27.6	23.97	8.31	1,361.47	T
Tetracentron sinense	YL	77.5	8.99	17.08	3.46	5.78	10.73	33.22	0.08	0.27	40.11	40.46	16.88	1,314.04	T
Ulmus microcarpa	YL	74.9	9.76	19.62	1.65	7.9	13.04	25.93	0.03	0.34	24.08	44.9	5.28	1,213.57	T
Helwingia japonica	YL	94.2	9.2	13.83	3.2	9.4	17.51	50.67	0.04	1.32	30.6	22.9	5.24	1,370.33	H
Holboellia fargesii	YL	82	10.04	20.29	1.39	8.07	14.93	52.1	0.22	0.56	27.9	18.15	6.26	1,393.31	L
Ilex sikkimensis	YL	76.2	7.49	16.49	8.93	4.29	29.11	38.37	0.18	0.33	34.37	31.92	19.8	1,511.94	T
Ligustrum delavayanum	YL	74.2	8.4	25.26	3.25	5.87	7.86	30.29	0.13	0.38	32.93	35.33	7.27	1,352.28	S
Lonicera nigra	YL	75.5	9.15	15.18	0.58	6.13	31.8	52.77	0.19	0.25	31.19	25.34	12.77	1,351.9	S
Machilus rufipes	YL	80.6	9.61	19.42	3.02	6.04	10.79	22.8	0.04	0.41	33.24	48.72	18.44	1,245.16	T
Merrilliopanax chinensis	YL	83	11.4	19.87	2.76	8.78	13.11	30.31	0.003	0.22	40.99	38.28	16.85	1,269.39	S
Michelia shiluensis	YL	84.2	9.48	27.96	3.76	6.1	8.88	22.99	0.06	0.52	26.1	39.19	3.6	1,334.99	T
Photinia integrifolia	YL	76.5	8.22	15.19	2.11	5.6	6.19	44.36	0.07	0.35	15.56	32.74	3.93	1,340.72	S
Rhododendron decorum	YL	81.6	9.49	18.1	2.63	5.46	17.68	33.18	0.06	0.35	29.26	40.63	12.6	1,301.56	T
Rhododendron zaleucum	YL	79	11.75	14.13	6.05	4.08	38.62	31.69	0.21	0.33	39.32	44.05	29.8	1,367.55	S
Skimmia arborescens	YL	84.8	9.41	27.77	4.43	6.5	11.54	36.01	0.06	0.52	22.44	25.29	4.11	1,439.77	T
Symplocos dryophila	YL	85	9.57	17.14	0.49	7.17	25.55	35.16	0.2	0.24	38.99	40.04	19.15	1,232.43	T
Vaccinium delavayi	YL	83.1	13.81	8.23	1.42	2.58	22.97	61.78	0.22	0.13	27.04	25.99	12.64	1,414.84	S
Hydrangea heteromalla	YL	84.7	10.99	37.58	2.42	10.91	15.78	25.55	0.33	0.54	27.03	23.54	8.12	1,349.06	S

Notes: ML= mature leaves, BUD = buds, YL = young leaves; Mo = Moisture, AW = Absorb Water, CP = Crude Protein, CL = Crude Lipids, CA = Crude Ash, TNC = Total Non-structural Carbohydrate, Ca = Calcium, P = Phosphorus, NDF = Neutral Detergent Fiber, ADF = Acid Detergent Fiber, ADL = Acid Detergent Lignin, ME = Metabolisable Energy; T = Tree, L = Liana, S = Shrub, H = Herb.

Table Appendix VIII. Nutritional properties of spring and autumn food items offered to but not consumed by captive *Rhinopithecus strykeri*

Name	Item Position	Mo	AW	CP	CL	CA	CF	TNC	Ca	P	ADF	NDF	ADL	ME	Type
Alnus nepalensis	BUD	74.2	8.35	30.84	6.8	4.57	44.17	30.76	0.17	0.62	27.8	27.03	19.63	1,511.31	T
Magnolia rostrata	BUD	79.1	12.61	23.17	1.35	4.41	16.56	25.86	0.14	0.5	63.53	45.21	15.28	1,264.49	T
Rhododendron araiophyllum	BUD	80.9	8.12	15	0.9	4.11	19.79	36.51	0.07	0.26	32.04	43.48	19.39	1,263.78	S
Salix phanera var. weixiensis	BUD	76.2	9.65	24.81	1.52	6.02	21.78	31.61	0.21	0.58	43.74	36.04	13.05	1,308.13	T
Viburnum cylindricum	BUD	70.1	9.07	29.63	15.71	5.97	14.54	11.77	0.22	0.46	24.18	36.92	19.59	1,605.58	T
Illicium simonsii	BUD	79.3	9.01	25.72	0.84	4.64	11.09	9.32	0.05	0.5	33.72	59.48	16.71	1,152.76	T
Pinus armandii	BUD	78.6	10.62	17.22	4.55	3.72	14.81	27.57	0.06	0.43	22.04	46.94	7.19	1,324.67	T
Rhododendron sidereum	BUD	76.4	10.64	30.86	1.84	4.16	23.59	11.23	0.22	0.19	45.37	51.91	27.02	1,239.57	T
Rhododendron sinogrande	BUD	80.4	9.95	13.7	2.21	4.13	19.73	39.54	0.21	0.29	42.32	40.42	19.8	1,311.09	T
Acer campbellii	ML	59.9	7.25	13.86	5.28	5.78	18.33	17.62	0.3	0.22	32.42	57.46	12.64	1,230.04	T
Betula luminifera	ML	65.8	8.51	18.17	3.69	5.31	14.35	12.78	1.11	0.12	34.14	60.05	17.78	1,190.35	T
Buddleja forrestii	ML	68.35	8.61	20.79	1.74	5.76	22.47	10.2	0.45	0.38	34.55	61.51	13.58	1,134.6	H
Cerasus clarofolia	ML	63.8	8.68	13.88	3.81	10.42	5.99	6.27	0.4	0.19	15.62	65.62	6.69	1,067.83	T

Prunus salicina	ML	60	6.93	16.44	3.13	7.86	20.51	23.26	0.42	0.13	23.09	49.31	11.72	1,211.24	T
Schisandra neglecta	ML	75.6	9.6	19.31	3.55	7.98	7.04	5.48	0.32	0.31	16.63	63.68	6.04	1,120.42	L
Tetracentron sinense	ML	67.7	9.37	10.39	3.36	6.95	9.95	0.25	0.36	0.15	37.35	79.05	21.62	1,016.4	T
Cyclobalanopsis glaucoides	ML	60.1	7.55	14.57	2.05	5.25	29.63	3.565	0.4	0.17	39.87	74.57	19.32	1,051.67	T
Dodecadenia var. gri	ML	58.5	7.42	13.81	4.23	5.12	29.56	9.17	0.39	0.13	41.87	67.67	20.68	1,147.03	T
Hedera nepalensis	ML	68.9	7.63	13.23	2.46	6.53	19.29	28.77	0.37	0.24	31.49	49.01	10.78	1,217.34	L
Ilex sikkimensis	ML	65.4	5.65	12.33	2.32	4.63	33.7	45.22	0.45	0.13	36.68	35.5	10.28	1,338.47	T
Ilex micropyrena	ML	65.4	9.42	4.4	2.99	4.99	23.48	46.54	0.22	0.03	41.77	41.08	19.44	1,299.79	S
Lindera thomsonii	ML	50	8.28	8.56	3.51	4.16	39.52	43.09	0.23	0.11	53.93	40.68	35.83	1,330.95	T
Lithocarpus pachyphyllus	ML	53	7.36	10.77	3.56	2.52	38.52	11.56	0.41	0.1	34.03	71.59	16.72	1,144.08	T
Lonicera nigra	ML	64.7	8.82	13.81	2.24	7.5	14.46	29.79	0.41	0.11	25.7	46.66	7.94	1,214.23	L
Rhododendron decorum	ML	61.6	7.22	15.37	4.05	3.81	58.4	32.74	0.27	0.08	32.78	44.03	8.52	1,331.25	T
Rhododendron sidereum	ML	61.6	9.61	5.15	3.67	4.37	23.45	42.46	0.31	0.04	35.24	44.35	13.05	1,301.85	T
Rhododendron sinogrande	ML	64.45	7.86	10.15	2.68	4.83	19.05	16.03	0.37	0.08	35.09	66.31	11.11	1,124.98	T
Sabina recurva	ML	61.6	8.27	10.9	6.18	3.38	29.57	23.65	0.51	0.27	31.82	55.89	14.39	1,295.37	T
Schefflera shweliensis	ML	62.4	8.14	10.08	1.4	5.33	13.01	35.04	0.33	0.09	22.37	48.15	13.18	1,217.15	T
Schima argentea	ML	63.1	7.65	13.99	2.5	3.92	27.96	15.91	0.51	0.07	30.99	63.68	14.29	1,158.01	T
Skimmia laureola	ML	66.3	9.36	14.61	5.51	6.26	16.93	33.44	0.38	0.08	28.2	40.18	14.52	1,348.9	T
Symplocos viridissima	ML	58.4	5.99	11.04	2.2	3.27	25.31	39.45	0.23	0.12	36.58	44.04	13.95	1,296.64	T
Vaccinium delavayi	ML	66.7	9.53	3.68	2.57	5.78	41.9	54.56	0.25	0.04	30.21	33.41	18.06	1,331.3	S
Alnus nepalensis	YL	73.6	9.53	29.24	3.9	4.62	18.01	12.82	0.06	0.32	33.79	49.42	27.81	1,291.28	T
Cotoneaster franchetii	YL	68	9.94	14.87	0.55	13.34	11.99	37.36	0.28	0.21	31.98	33.88	17.57	1,175.82	S

Enkianthus deflexus	YL	82.2	13.21	22.85	2.55	8.22	23.62	45.37	0.21	0.44	29.02	21.01	5.98	1,397.81	T
Juglans cathayensis	YL	78.5	9.34	22.42	2.28	6.65	13.56	21.95	0.14	0.34	28.78	46.7	8.68	1,237	T
Magnolia rostrata	YL	78.1	12.05	19.97	2.5	7.54	26.73	27.03	0.24	0.37	47.95	42.96	18.1	1,251.11	T
Polygonum molle	YL	81.5	9.5	21.04	2.82	6.64	15.06	28.63	0.05	0.52	22.1	40.87	5.13	1,287.93	H
Populus yunnanensis	YL	75.4	9.65	23.88	3.35	7.28	11.71	25.13	0.05	0.29	28.65	40.36	18.25	1,295.23	T
Pterocarya delavayi	YL	77.4	10.31	20.07	1.65	6.99	15.49	49.67	0.21	0.43	31.2	21.62	9.24	1,391.54	T
Rubus lineatus	YL	72.8	8.37	17.15	1.9	5.52	22.41	33.07	0.14	0.28	23.43	42.36	4.3	1,272.21	S
Salix phanera var. weixiensis	YL	76.8	9.88	24.24	3.4	10.67	13.2	29.47	0.27	0.52	39.07	32.22	16.08	1,299.38	T
Schefflera hoi	YL	80.3	9.34	13.72	2	6.3	10.56	22.58	0.07	0.25	36.06	55.4	6.33	1,167.12	T
Sinosassafras flavinervium	YL	80.7	9.47	25.73	7.05	5.08	11.77	32.26	0.04	0.3	16.27	29.88	5.18	1,483.34	T
Viburnum nudum	YL	72.6	8.59	13.03	4.2	6.05	22.56	30.51	0.12	0.21	36.3	46.21	13.56	1,280.96	S
Abies delavayi	YL	73.7	9.25	17.69	5.51	3.04	12.12	29.22	0.06	0.32	16.02	44.54	10.3	1,372.98	T
Eurya pseudocerasifera	YL	74	9.48	10.43	1.43	5.38	11.64	60.55	0.05	0.32	26.47	22.21	7.44	1,398.91	T
Gaultheria forrestii	YL	79.5	9.12	18.38	3.01	3.92	16.73	30.36	0.04	0.51	29.12	44.33	17.21	1,308.56	S
Litsea elongata	YL	75.1	9.58	22.26	2.84	5.2	13.61	3.83	0.05	0.53	52.59	65.87	30.94	1,137.61	T
Rhododendron edgeworthii	YL	76.5	9.74	11.18	2.03	5.67	67.88	26.58	0.22	0.33	54.99	54.54	27.5	1,181.31	S
Skimmia laureola	YL	66.3	7.95	16.32	5.86	4.39	21.34	45.37	0.34	0.13	24.51	28.06	12.83	1,472.72	T
Symplocos ramosissima	YL	76.6	9.15	18.9	0.55	8.4	11.42	38.97	0.09	0.31	21.36	33.18	6.9	1,263.79	T
Tsuga dumosa	YL	77	9.26	14.3	5.53	3.08	18	34.18	0.05	0.18	21.65	42.91	10.61	1,380.78	T

Notes: ML= mature leaves, BUD = buds, YL = young leaves; Mo = Moisture, AW = Absorb Water, CP = Crude Protein, CL = Crude Lipids, CA = Crude Ash, TNC = Total Non-structural Carbohydrate, Ca = Calcium, P = Phosphorus, NDF = Neutral Detergent Fiber, ADF = Acid Detergent Fiber, ADL = Acid Detergent Lignin, ME = Metabolisable Energy; T = Tree, L = Liana, S = Shrub, H = Herb.

Table Appendix IX. Pairwise comparison of the nutritional content (% dry matter) and metabolisable energy (kJ/100g) of food items consumed and avoided by captive *Rhinopithecus strykeri* in Mt. Gaoligong, China during Spring and Autumn. Data presented represent the mean nutritional content (%) ± SD.

			Moisture		CP		CL		TNC		Calcium	
			Statistic	P-value	Statistic	P-value	Statisti	P-value	Statistic	P-value	Statistic	P-value
FI	vs.	NFI	5.63	0	3.976	0	-0.193	0.847	3.423	0.001	-1.332	0.186
CYL	vs.	NCYL	3.842	0	2.702	0.01	-0.375	0.711	1.359	0.184	1.238	0.225
CML	vs.	NCML	4.943	0	2.469	0.017	1.104	0.276	2.164	0.036	-1.465	0.15
SFI	vs.	SNFI	4.153	0	2.152	0.036	-0.38	0.706	2.285	0.027	1.231	0.224
SFI	vs.	AFI	7.192	0	4.97	0	-2.249	0.03	1.16	0.254	-7.999	0

			Phosphorus		NDF		ADF		ADL		ME	
			Statistic	P-value	Statistic	P-value	Statistic	P-value	Statistic	P-value	Statistic	P-value
FI	vs.	NFI	4.223	0	-5.73	0	-1.208	0.23	-1.853	0.066	4.514	0
CYL	vs.	NCYL	3.109	0.003	-2.708	0.011	-0.066	0.947	-0.871	0.39	2.305	0.029
CML	vs.	NCML	2.068	0.044	-3.556	0.001	-0.975	0.335	-1.301	0.199	3.36	0.002
SFI	vs.	SNFI	2.913	0.005	-3.803	0	-0.786	0.436	-1.272	0.208	2.3	0.026
SFI	vs.	AFI	7.768	0	-3.813	0	0.199	0.843	-0.314	0.754	1.728	0.093

			CP/NDF		CP/ADF		Moisture/NDF		Calcium/Phosphorus	
			Statistic	P-value	Statistic	P-value	Statistic	P-value	Statistic	P-value
FI	vs.	NFI	6.227	0	3.561	0.001	6.576	0	-3.078	0.003
CYL	vs.	NCYL	3.828	0	1.56	0.128	3.337	0.002	-0.292	0.772
CML	vs.	NCML	4.204	0	2.129	0.038	4.733	0	-2.73	0.009
SFI	vs.	SNFI	4.287	0	1.95	0.056	4.519	0	-0.32	0.75
SFI	vs.	AFI	5.74	0	2.999	0.005	5.522	0	-7.966	0

Mean Proportion %	Moisture	CP	CL	TNC	Calcium
FI	77.68 ± 6.25	21.23 ± 6.46	3.22 ± 2.04	34.86 ± 11.76	0.21 ± 0.12
NFI	70.46 ± 7.99	16.81 ± 6.42	3.32 ± 2.29	21.44 ± 13.88	0.25 ± 0.18

SFI	80.07 ± 4.43	23.00 ± 5.65	2.97 ± 1.82	35.66 ± 11.51	0.16 ± 0.08
SNFI	76.39 ± 3.82	20.29 ± 5.72	3.35 ± 2.89	29.64 ± 12.08	0.14 ± 0.09
CML	70.88 ± 5.78	16.19 ± 5.85	3.92 ± 2.41	32.56 ± 11.97	0.33 ± 0.12
NCML	63.05±5.15	12.47 ± 4.21	3.28 ± 1.16	24.45 ± 15.38	0.39 ± 0.17
Mean Proportion %	Phosphorus	NDF	ADF	ADL	ME (kJ/100g)
FI	0.39 ± 0.21	34. 68 ± 11.25	30.68 ± 8.69	12.08 ± 7.55	1349.80 ± 94.71
NFI	0.27 ± 0.16	46.84 ± 13.47	32.75 ± 10.06	14.62 ± 6.80	1267.57 ± 114.05
SFI	0.46 ± 0.18	32.16 ± 9.8	30.82 ± 8.44	12.86 ± 8.12	1360.91 ± 82.24
SNFI	0.36 ± 0.13	40.87 ± 10.86	32.87 ± 11.39	14.59 ± 7.32	1310.96 ± 107.83
CML	0.20 ± 0.15	41.87 ± 11.85	30.31 ± 8.03	12.65 ± 5.41	1318.18 ± 116.79
NCML	0.14 ± 0.09	54.31 ± 12.67	32.60 ± 8.09	14.67 ± 6.09	1213.33 ± 97.15

Notes: FI - food items (n = 100); NFI – non-consumed food items (n = 54); CYL - consumed young leaves (n = 60); NCYL – non-consumed young leaves (n = 21); CML (=AFI) - consumed mature leaves (n = 26); NCML - non-consumed mature leaves (n = 24); SFI - spring food items (n = 69); SNFI - spring non-consumed food items (n = 30); AFI - autumn food items .

CP - crude protein; CL - crude lipids; TNC - total nonstructural carbohydrates; ME - metabolisable energy; NDF - neutral detergent fiber; ADF - acid

Table Appendix X. Evaluation of study methods and procedures in documenting primate feeding habits

Method	Advantages	Disadvantages	Reference or Examples
Field Observation	Reliable and accurate under good observation conditions.	① Heavy workload in the field; ② visibility and accessibility depending on environment conditions; ③ feasible only when observing a few easy-to-see behaviors.	Grueter et al., 2009; Xiang et al., 2007; Matthews et al., 2019
Collection of Discarded Food Present in the Forest	① Light workload in the field; identifying the species/parts consumed at a close distance; ③ as an auxiliary means.	Limited accuracy when there are sympatric species with similar feeding habits; reliable only when feeding remains are collected soon after an animal has vacated a patch.	Other non-human primates occurring in the study areas include *Trachypithecus phayrei*, *Macaca assamensis* and *Macaca arctoides* that consume some foods also eaten by *R. strykeri*; Stone, 2007
Cafeteria-style Feeding Trials	① Choice experiments under controlling ② identifying potential foods naturally consumed; ③ avoiding natural constraints (i.e. food availability, abundance, and distribution)	Omission of some foods that are consumed by wild individuals.	Molloy & Hart 2002; Hallgren & Hjältén, 2004; Yang et al., 2019; this study
Analysis of Stomach Contents	① Simple and easy operation; ② the volume of different food categories consumed can be assessed.	① Method does not permit one to visually observe foraging behaviors; ② only a single measure per animal is obtained; ③ entire dietary regime of a species cannot be accurately measured ④ hard to identify easily digested foods. ⑤ lethal method which should never be used for nonhuman primates.	Lahm, 1986; Nhat, 1994; Nekaris & Rasmussen, 2003

Fecal Analysis	① Favorable for the determination of food composition and food preferences; ② representative when samples collected at different locations in different seasons and among individuals of different age and sex classes.	① Differential digestibility of food items results in a proportional deviation of food composition because many easily digested foods quickly pass through the gastrointestinal tract; ② identifying foods items in feces demands substantial microscope training to accurately identify tissue and cell morphology and structures of different residue in feces. ③ requires a comprehensive collection and preparation of reference specimens; ④ impossible to determine how much time the primate species spends feeding on different food species/parts.	Yang & Zhao, 2001; Su et al., 2001
Fecal DNA Analysis	① Accurate species identification; ② not compromised by complexity of a diet.	① Not able to provide an accurate quantitative description of a diet due to variation in digestion rates.	Sheppard & Hardwood, 2005; Srivathsan et al., 2015; Lyke et al., 2019
Nutritional Chemistry Analysis	Reveals the nutritional content of a food item.	① Consumer selection and the degree to which the consume can process the food item are not necessarily linked; ② Only useful in conjunction with one of the above mentioned methods.	Huang et al., 2010; Liu et al., 2013; Hou et al., 2018; this study

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Table Appendix XI. Path, row numbers and acquisition dates of Landsat images used in forest dynamic analysis.

Landsat 7, 2000s			Landsat 8, 2015s		
path	row	date	path	row	date
132	40	2000360	132	40	2014022
132	41	2000360	132	41	2014022
132	42	2002013	132	42	2014022
132	43	2002013	132	43	2014022
133	40	2000047	133	40	2015016
133	41	2000047	133	41	2015016
133	42	2000047	133	42	2015016
133	43	2000047	133	43	2015048
132	40	2002333	132	40	2015105
132	41	2002333	132	41	2015105
132	42	2002333	132	42	2015073
132	43	2001330	132	43	2015073
133	40	2001337	133	40	2015080
133	41	1999300	133	41	2015080
133	42	1999316	133	42	2015080
133	43	1999316	133	43	2015080

Table Appendix XII. Bioclimatic variables used in MAXENT

BIO1	Annual mean temperature
BIO2	Mean diurnal temperature range [mean of monthly (max temp–min temp)]
BIO3	Isothermality (P2/P7) (×100)
BIO4	Temperature seasonality (standard deviation ×100)
BIO5	Max temperature of warmest month
BIO6	Min temperature of coldest month
BIO7	Temperature annual range (P5–P6)
BIO8	Mean temperature of wettest quarter
BIO9	Mean temperature of driest quarter
BIO10	Mean temperature of warmest quarter
BIO11	Mean temperature of coldest quarter
BIO12	Annual precipitation
BIO13	Precipitation of wettest month
BIO14	Precipitation of driest month
BIO15	Precipitation seasonality (coefficient of variation)
BIO16	Precipitation of wettest quarter
BIO17	Precipitation of driest quarter
BIO18	Precipitation of warmest quarter
BIO19	Precipitation of coldest quarter

Table Appendix XIII. List of threatened mammals monitored in *Rhinopithecus strykeri* habitat in the Gaoligong Mountains.

Location: MW, western slopes of Gaoligong Mountain, northeastern Kachin state, Myanmar; WC, western slopes of Gaoligong Mountain, Yunnan, China; EC: eastern slopes of Gaoligong Mountain, Yunnan, China. Habitat types: LE: low land subtropical evergreen rain forest; SM: semi moist evergreen broad-leaved forest; ME: mid-montane moist evergreen broad-leaved forest; TC, temperate coniferous forest; BB, bamboo bushes. Monitoring method: CT, camera trapping; FO, field observation; ST, specimens of skulls and tails collected from hunters; IT, interview. IUCN RED List: CR, Critically Endangered; EN, Endangered; VU: Vulnerable; NT, Near Threatened. CITES Appendix: APP. I, Appendix I; APP. II, Appendix II.

Species	Location	Elevation (m) / Habitat types	Monitoring method	Conservation status	Reference
Macaca arctoides	MW, WC, EC	2,500-3,100 / ME, TC, BB	CT, IT, FO	VU, APP. II	Momberg et al., 2010; Chen et al., 2016; Fig. A.2.D
Macaca assamensis	MW, WC, EC	2,500-3,100 / ME, TC, BB	CT; FO, ST	NT, APP. II	Momberg et al., 2010; Chen et al., 2016; Fig. A.2.C
Macaca leonina	MW	< 2,700/ LE, SM, ME	IT, FO	VU, APP. II	Momberg et al., 2010
Macaca mulatta	MW, EC	1,300-3,800/ LE, SM, ME, TC	IT, ST	APP. II	Momberg et al., 2010; Xiao et al., 2013a
Trachypithecus phayrei	MW, WC, EC	1,200-2,600 SM, ME	FO; IT	EN, APP.I	Meyer et al., 2015; Xiao et al,. 2013a
Trachypithecus shortridgei	MW, EC	500-2,000 / LE, SM, ME	FO, ST	EU, APP. I	Meyer et al., 2015; Momberg et al.,

					2010; Xiao et al,. 2013a
Hoolock tianxing	MW, WC	400-2,000 / LE, SM, ME	FO, ST, IT	EN, APP. I	Geissmann et al., 2013; Fan et al., 2016
Budorcas taxicolor	MW, WC, EC	2,700-3,200 / ME, TC, BB	CT, FO, IT, ST	VN, APP. II	Momberg et al., 2010; Xiao et al., 2013b
Elaphodus cephalophus	WC	2,500-3,300 / ME, TC, BB	CT, FO	NT	Momberg et al., 2010; Chen et al., 2016
Capricornis milneedwardsii	MW, WC, EC	2,500-3,100 / ME, TC, BB	CT, ST	NT, APP. II	Chen et al., 2016; Fig. A.2.E
Ailurus styani	MW, WC, EC	2,300-3,500 / ME, TC, BB	CT, FO	EN, APP. I	Momberg et al., 2010; Chen et al., 2016; Fig. A.2.B
Pardofelis cf. *marmorata*	EC	<2,800 / SE, ME	CT	NT, APP. I	Fig. A.2.A
Ursus thibetanus	MW, WC	<3,200 / ME, TC, BB	CT	VN, APP. I	Momberg et al., 2010; Chen et al., 2016
Helarctos malayanus	MW	< 2,700 m / SE, ME	FO, ST	VN, APP. I	Momberg et al., 2010
Cuon alpinus	MW	Uncertain	Iron Trap	EN, APP. II	Momberg et al., 2010

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Table Appendix XIV. Path, row numbers and acquisition dates of Landsat images used in modelling for selected flagship species.

	Landsat 8, 2015s	
path	row	date
132	40	2014022
132	41	2014022
132	42	2014022
132	43	2014022
133	40	2015016
133	41	2015016
133	42	2015016
133	43	2015048
132	40	2015105
132	41	2015105
132	42	2015073
132	43	2015073
133	40	2015080
133	41	2015080
133	42	2015080
133	43	2015080

Table Appendix XV. Hierarchical definition of forests

Name	Definition
Fs: forest + shurb	Tree crown cover degree > 40%，including forest and shrubs.
Fw: forest = Fc + Fo	Tree crown height > 3m
Fc: closed-forest	Tree crown cover degree > 70%
Fo: open-forest	40% < Tree crown cover degree < 70%
Sh: shrub	Plant cover degree > 40%, but not fulfilling the other conditions of forested habitats

Table Appendix XVI. Description of habitat quality levels for selected flagship species

Quality	Predicted value	Description
Core	> 0.50	Equal training sensitivity and specificity (0.50)
Medium	0.41-0.50	Maximum training sensitivity plus specificity (0.41)
Edge	0.04-0.41	Minimum training presence data omission (0.04)
Non-habitat	< 0.04	Less than 0.1% of training presence data predicted (0.04)

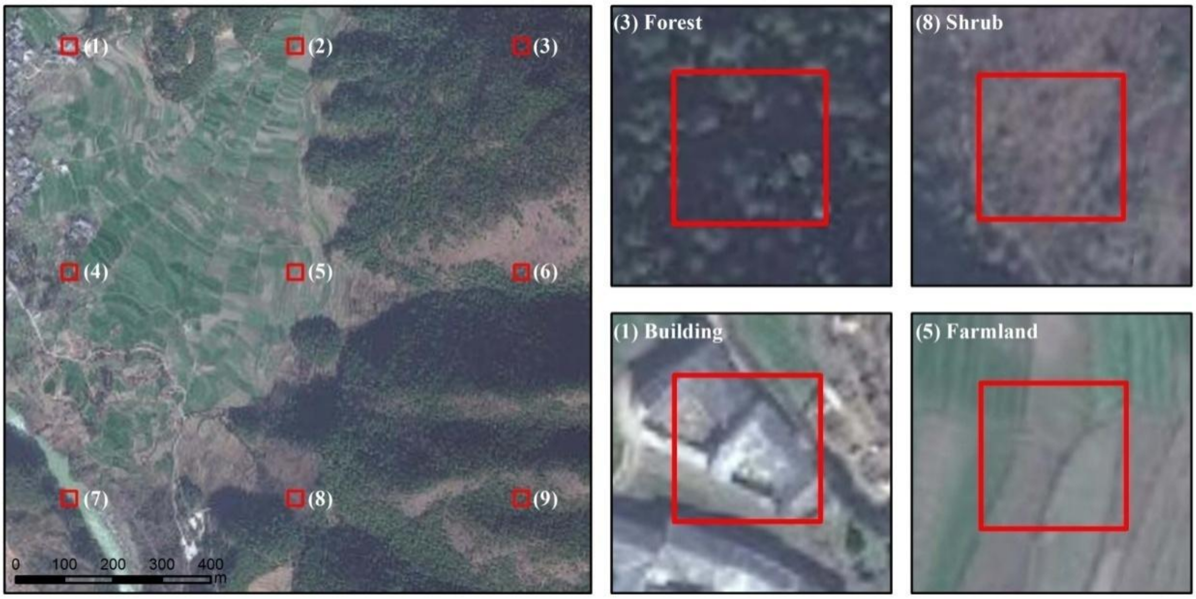
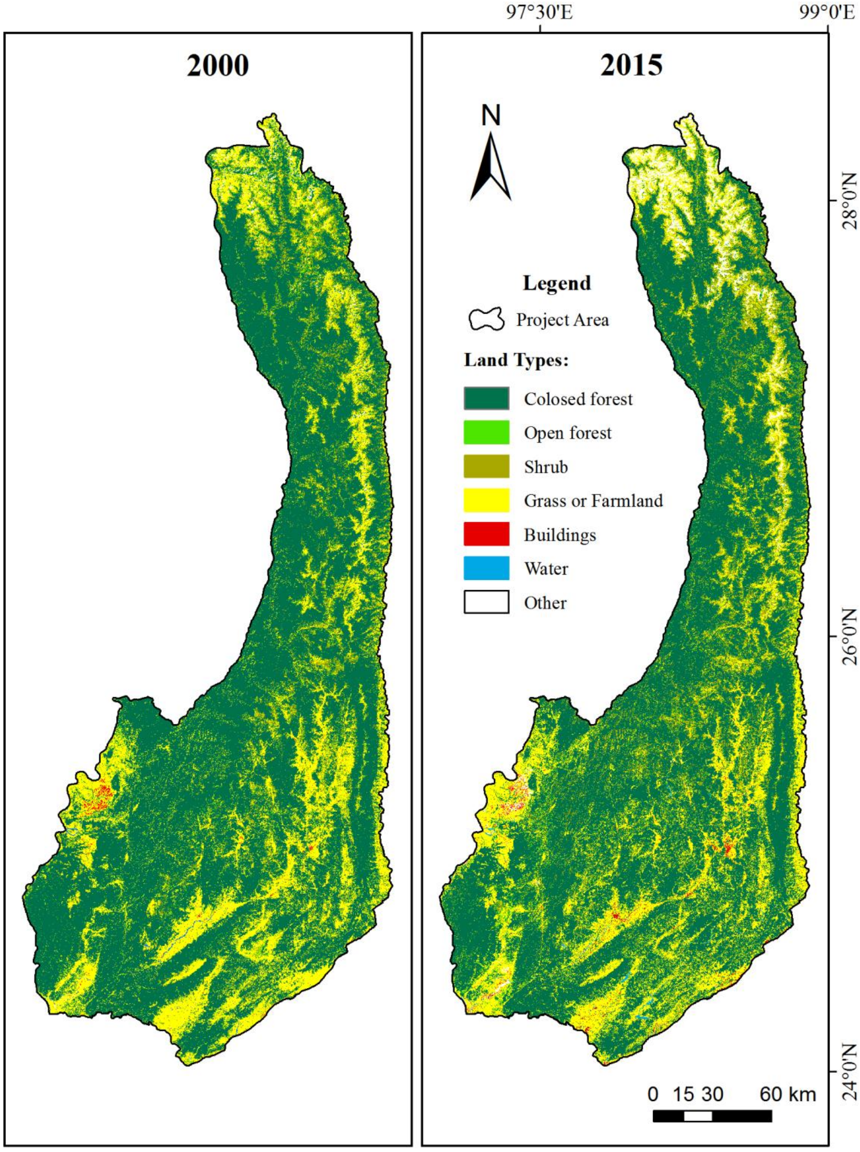


Figure Appendix I. High resolution images from Google Earth (Version 7.1), aggregated to 30 m resolution. The left-hand image has several land cover categories, including forest, shrub, buildings and farmland. Zoomed-in image 1 shows scattered buildings; images 3 and 8 show forest cover and dense shrubs respectively; farmland is shown in window 5.

Figure Appendix II. Land type changes in the study area in 2000 and in 2015.



2000		2015	
Land Type	Area Size (km^2)	Land Type	Area Size (km^2)
Closed Forest	28,979.0	Closed Forest	26,515.7
Open Forest	1,704	Open Forest	1,587.5
Shrub	3,286.2	Shrub	4,502.2
Grass+Framland	9,064.5	Grass+Framland	9,353.3
Buildings	104.3	Buildings	268.0
Water Area	101.7	Water Area	173.7
Other	251.5	Other	1,080.9

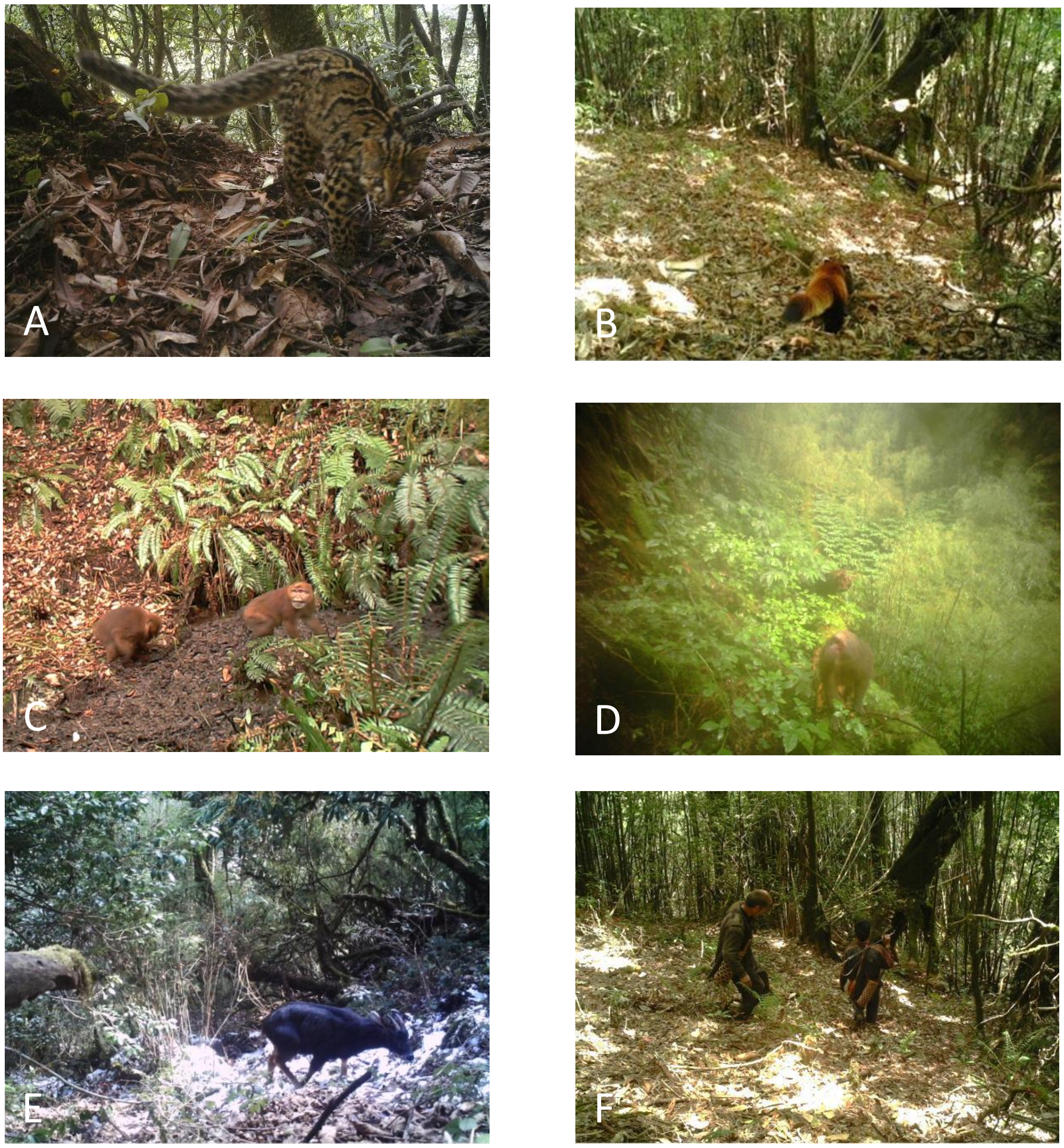


Figure Appendix III. Mammals and human threats were monitored by camera traps at *Rhinopithecus strykeri*'s habitats on the eastern slope of the Gaoligong Mountains, China from April 2015 to January 2017. A: *Pardofelis marmorata* (Credit to Wang Bin); B: *Ailurus styani*; C: *Macaca assamensis*; D: *Macaca arctoides*; E: *Capricornis milneedwardsii*; F: two hunters with a shotgun.

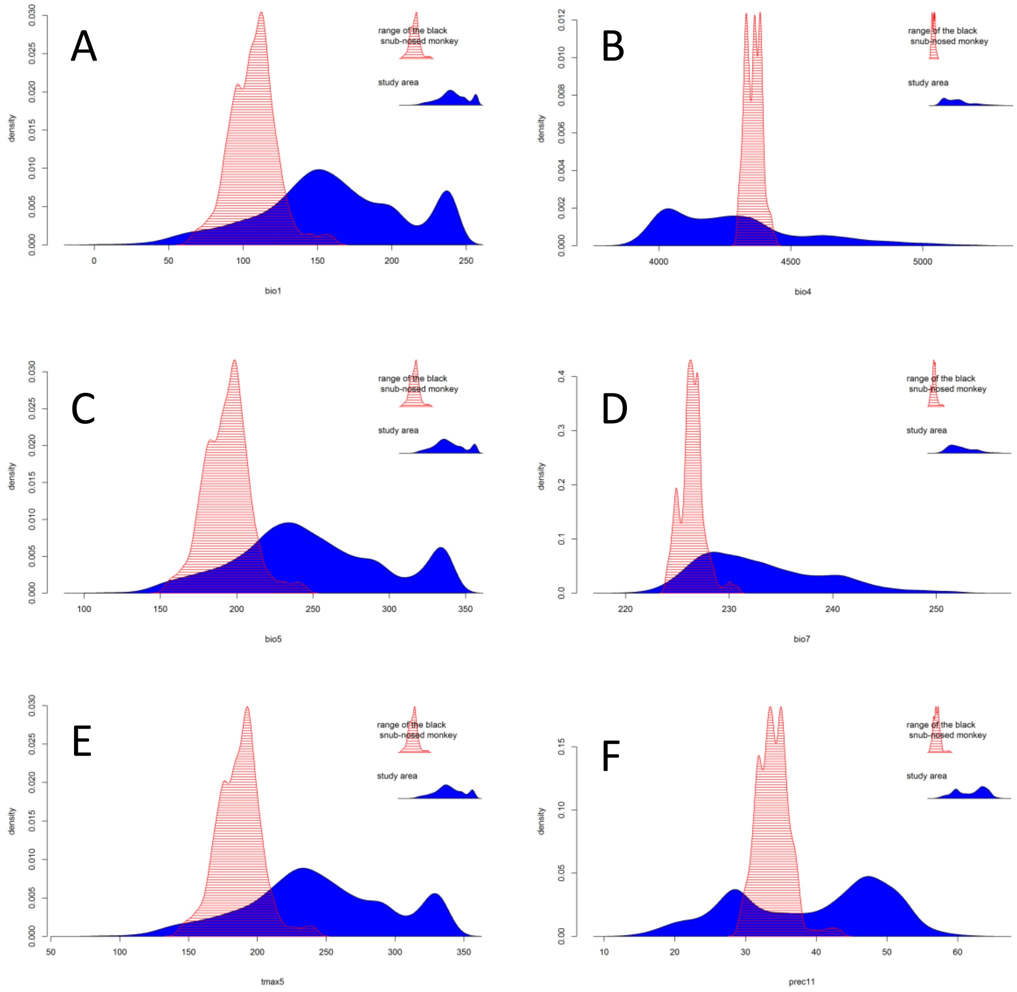


Figure Appendix IV. Probability distribution curves (six examples: A-F) show how climate variables affect the MAXENT prediction of habitat suitability for *Rhinopithecus strykeri* in the study area. Blue areas correspond to climate variables in the study area while red areas correspond to climate variables in the distribution range of *R. strykeri*. Among 67 climate variables, the seasonal variation of temperature (B: bio4), the temperature annual range (D: bio7), the average precipitation in November (F: pre11) and the maximum temperature in May (E: tmax5) contributed most to the MAXENT established in this study, reflecting a strong selectivity of *R. strykeri* for the above-mentioned variables. A: bio1, annual mean temperature; C: bio5, maximum temperature of the warmest month.

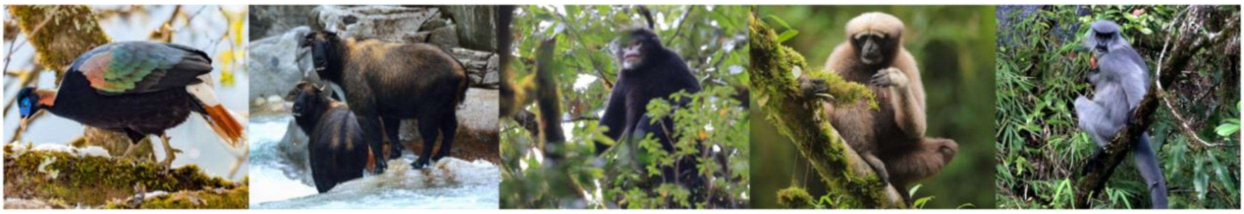


Figure Appendix V. The five flagship species: Sclater's monal (*Lophophorus sclateri*), Mishmi takin (*Budorcas taxicolor*), black snub-nosed monkey (*Rhinopithecus strykeri*), Skywalker Hoolock Gibbon (*Hoolock tianxing*), and Shortridge’s Langur (*Trachypithecus shortridgei*) (from left to right). Photo by Wan Bin, Zuo Lingren, Zhao Chao and Peng Jiansheng.

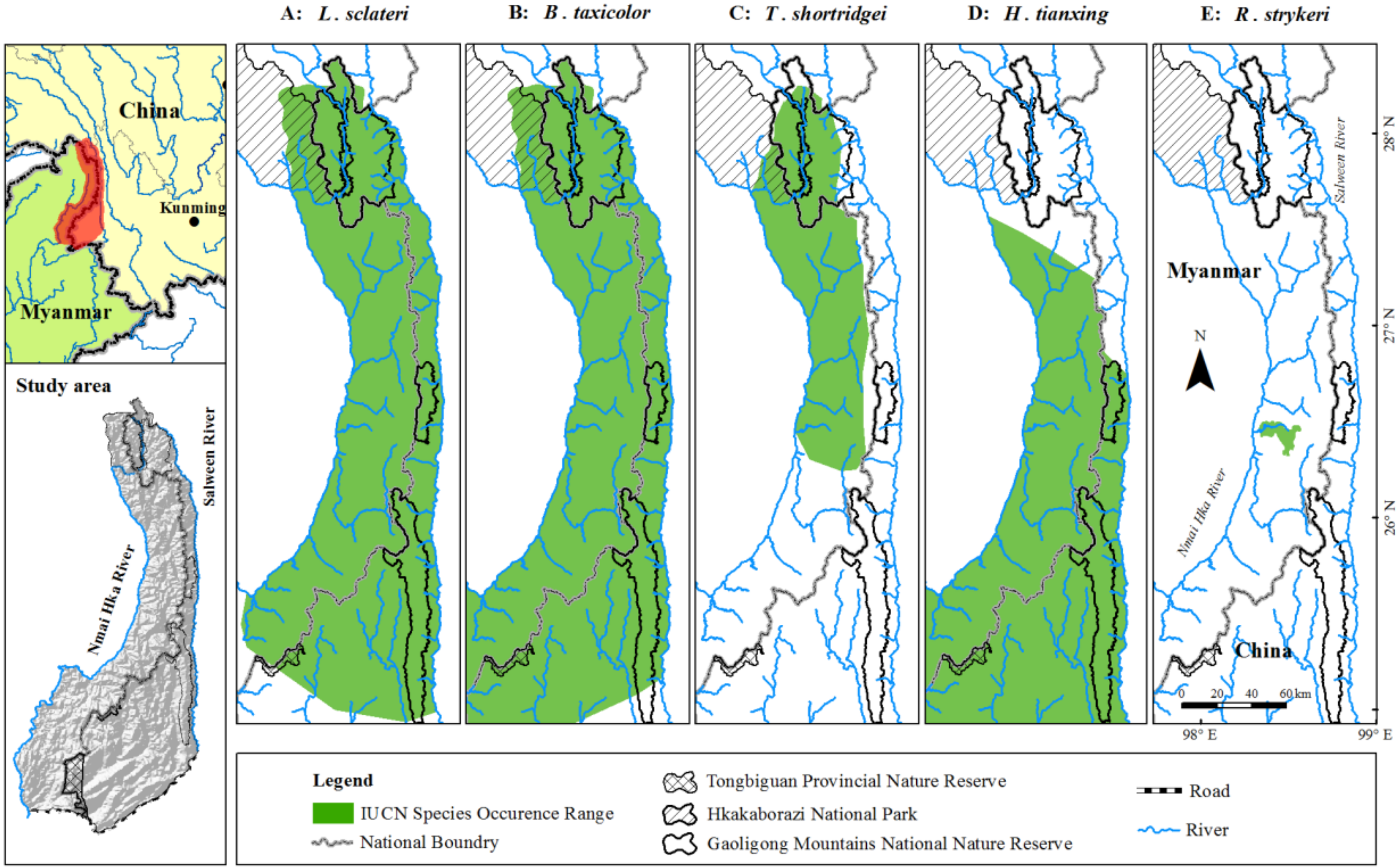


Figure Appendix VI. IUCN species distribution ranges of the five selected flagship species in the study area.

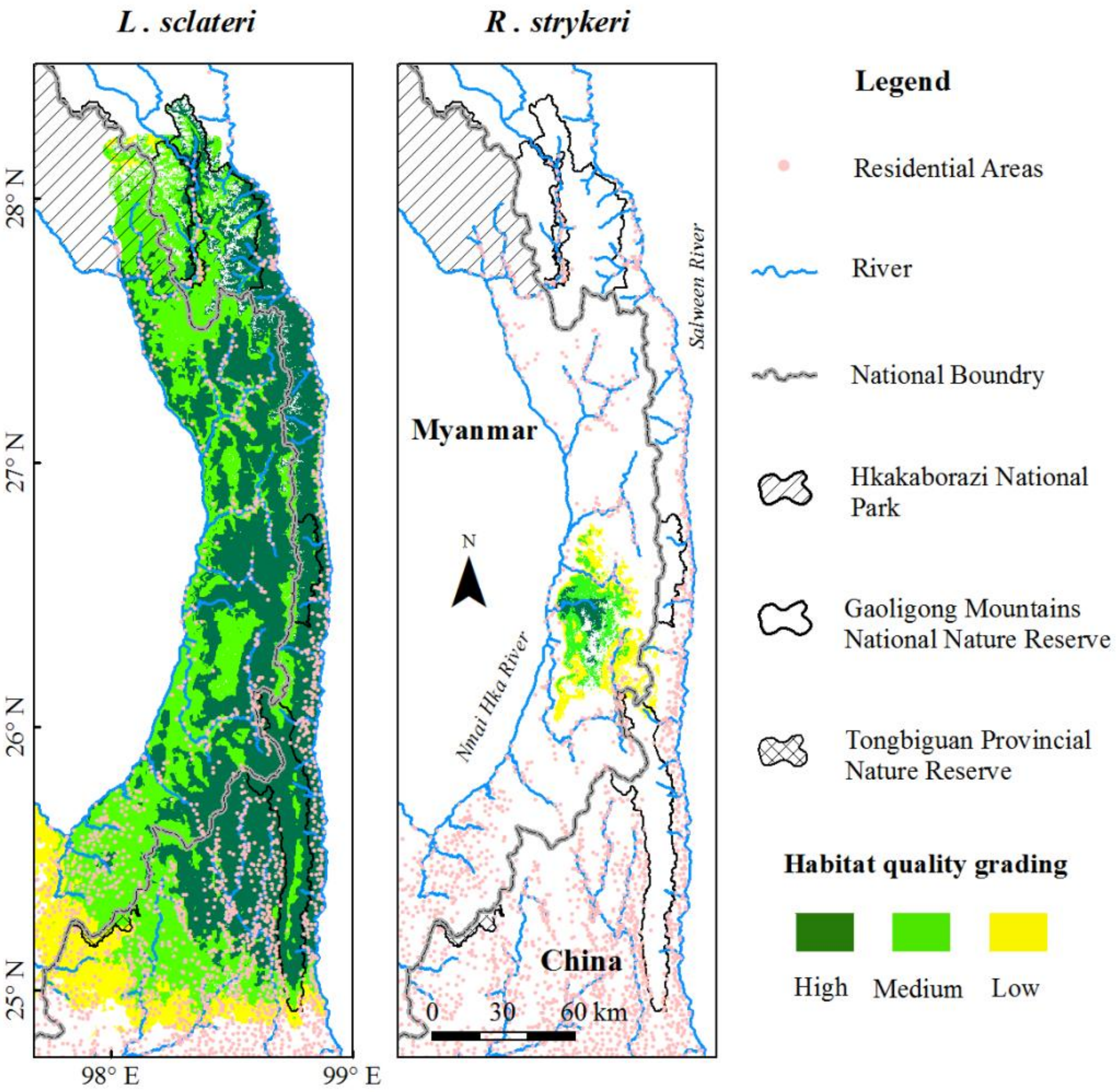


Figure Appendix VII. Predicted species ranges of *Rhinopithecus strykeri* and *Lophophorus sclateri* based on data from the IUCN species distribution range and MAXENT modelling. The left figure shows an example of commission error, the right figure shows the omission error, and both show a highly biased habitat distribution prediction and habitat suability rankings.

Text Appendix I. Procedures used for Random Forests

```
library(randomForest)
>
> driver.name = "d://"
>
> data.path = paste(driver.name,"public//yangyin//",sep="")
> nutrition.path = paste(data.path, "nutrition//",sep="")
>
> data.csv = paste(data.path, "20180813-new.csv" ,sep="")
>
> my.data = read.csv(data.csv)
> my.data = my.data[my.data[,"Eat.or.Not"]!="O",]
>
> test.data = my.data[my.data[,"Eat.or.Not"]!="O",]
> eat.not = as.character(test.data[,"Eat.or.Not"])
>
> # my.data = my.data[,1:18]
>
> var.list = colnames(my.data)
>    #########################################################################
> ###### eat or not test
> x.list = var.list[-c(1:3)]
> x.list = setdiff(x.list,c("Eat.or.Not","Type" ,"Colour" ))
>
>
> predict.var = setdiff(var.list,c("Eat.or.Not","编号","Name", "AW", "Type", "Item.Position") )
> my.randfor = randomForest(x = my.data[,predict.var], y = as.factor(eat.not) )
> my.randfor

Call:
randomForest(x = my.data[, predict.var], y = as.factor(eat.not))
Type of random forest: classification
Number of trees: 500
No. of variables tried at each split: 3

OOB estimate of error rate: 26.7%
Confusion matrix:
   N      Y      class.error
N 32     33     0.5076923
Y 14     97     0.1261261
```

Text Appendix II. Modelling setting for *Rhinopithecus strykeri*

The default setting was used for the climatic and species distribution parameters in the R package dismo:

maxent.sample = maxent (x = trainvals.sample, p = species.sample)

in which, trainvals.sample is a 4772 × 67 matrix, and species.sample is a vector of length 4772 .

Text Appendix III. Modeling Setting for selected flagship species

The default setting was used for the climatic and species distribution parameters in the R package dismo:

maxent.sample = maxent (x = trainvals.sample, p = species.sample)

in which, trainvals.sample is a 1400 × 67 matrix, and species.sample is a vector of length 1400 .

Text Appendix IV. Review of distribution, and population and conservation status of the selected flagship species: black snub-nosed monkey (*Rhinopithecus strykeri*), Skywalker Hoolock Gibbon (*Hoolock tianxing*), Shortridge's Langur (*Trachypithecus shortridgei*), Sclater's monal (*Lophophorus sclateri*), and Mishmi takin (*Budorcas taxicolor*).

Black snub-nosed monkey

The black snub-nosed monkey was discovered in 2010 in the Northern Kachin State of Myanmar (Geissmann et al., 2011). Later, additional sub-populations were found on the western slopes and eastern slopes of Gaoligongshan Mountains, Yunnan of China (Long et al., 2012; Yang et al., 2018). The major forest types in the area of *R. strykeri* are semi moist evergreen broad-leaved forest, mid-montane moist evergreen broad-leaved forest, and coniferous broadleaved mixed forest at an elevation of 1,720 to 3,300 m. Until recently, scientists have confirmed five sub-populations and about 400 individuals in Sino-Myanmar border (Meyer et al., 2017). *R. strykeri* is recognized as Critically Endangered by the IUCN (Geissmann et al., 2012). At present, very little is known about the general biology and ecology of *R. strykeri*, and an accurate account of its geographical distribution range and population size require further research.

Skywalker hoolock gibbion

In China and Myanmar, the Skywalker hoolock gibbion occurs in tropical monsoon forest to moist evergreen broad-leaved forest at an elevation of 200-2,700 m from east of the Irrawaddy River up to west of the Salween River (Geissmann et al., 2013; Fan et al., 2017). In our study area, the natural range of this ape species extends from the east bank of the N'mai Hka River in the Myitkyina and Bhamo districts to the Sino-Myanmar border in Kachin State of Myanmar (Geissmann et al., 2013). The species enters China in westernmost Yunnan, where it inhabits the GLGM region and ranges between the west bank of the Salween River and the Chinese border (Fan et al., 2017). Its population status in the middle and lower reaches west of the N'mai Hka River remain unclear. In China, the Skywalker hoolock gibbion was historically documented in nine counties but now is restricted to three counties, namely Baoshan, Tengchong, and Yingjiang, with less than 200 individuals in 21 fragmented sub-populations (Fan et al., 2011). Thus, it is listed as Critically Endangered in China and its status is unknown in Myanmar. Hunting, cardamom cultivation, and commercial logging are the main threats to the Skywalker hoolock gibbon in the border area (Fan et al., 2011; Geissmann et al., 2013). Accordingly, more information on population size, distributional and habitat range, and trans-boundary conservation are needed along with a national-level awareness in order to save this species from extinction in the Sino-Myanmar border region.

Shortridge's Langur

Shortridge’s langur occurs at low (200 m a.s.l.) to medium (2,500 m a.s.l.) elevation in tropical monsoon forests and mid-montane moist evergreen broad-leaved forests east of the Chindwin River in Kachin State, north to the Myitkyina District, Myanmar, and north to the Duong River in Gongshan District, China (Cui et al., 2016; Meyer et al., 2017). The total population size is not known, but is reported to be low and decreasing due to hunting and habitat destruction for wood extraction and land exploitation (Cui et al., 2016). This species has been listed as Endangered by the IUCN, with an estimated 50% reduction in the total population over the past 35 years (Htun et al., 2008). Based on limited field data and interviews, the population size, distribution and conservation status of this species was estimated in the Dulong River Basin

(Cui et al., 2016). Clearly more data are needed in order to develop effective conservation programs to protect this species.

Mishmi takin

The Mishmi takin is found in the Gaoligong Mountain region between the west bank of the Salween river (from Longling to Gongshan county) and the east bank of the N'mai River, northern Myanmar (Hkhakabo-Razi National Park), Bomê- Mêdog - Dzayul region, and Mishmi hills in Tibet (Rabinowitz & Khaing, 1998; Song et al., 2008; Momberg et al., 2010). This species usually ranges between 1,500 and 4,500 m a.s.l. in monsoon evergreen broad-leaved forests and alpine bamboo dominated bushes (Rabinowitz & Khaing, 1998; Song et al., 2008; Momberg et al., 2010). Based on interview information, Mishmi takin group size can be as small as 5-20 and as large as 50-65 individuals (Rabinowitz & Khaing, 1998; Zhang et al., 2014). The total population is unknown in China, but Zhang et al., (2014) estimated 400-450 individuals in Dulong Valley. In Myanmar, populations are decreasing as a result of hunting for bush meat and trade of horns and skins (Rabinowitz & Khaing, 1998; Momberg et al., 2010). The species has been listed as Vulnerable by IUCN because there appears to be at least a 30% population decline as a result of hunting and habitat loss over the past three decades (Song et al., 2008). Due to the lack of detailed studies, surveys are needed to obtain information on its population, habitat distribution, and conservation threats.

Sclater's monal

Sclater's monal is endemic to the eastern Himalayas, from South Tibet, China, through northern Myanmar to northwestern Yunnan, China (BirdLife International, 2013). It inhabits coniferous forest, subalpine thicket, alpine meadows, and rocky crags at an elevation of 2,800 to 4,000 m (Han et al., 2004). The behavior and ecology of this beautiful pheasant is not well known. It is categorized as Vulnerable due to its small and naturally fragmented population, habitat degradation, and the fact that it is hunted by local communities (BirdLife International, 2013). Field surveys are needed to understand its population and distribution status, assess the effectiveness of the current conservation network, and to develop conservation strategies and management plans for this species.

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Text Appendix V. Justification of Outstanding Universal Value of the proposed additional World Natural Heritage area to Hkakabo Razi Landscape.

Criterion (vii.) Natural phenomena or beauty

The beautiful scenery to the east of the N'Mai Hka River is characterized by virgin forest-covered mountains and non-navigable rivers that originate from stunning snow-capped ridges and peaks that form hundreds of rugged deep valleys and many waterfalls. The area includes several of the tallest snow covered mountains of Southeast Asia, including Imaw Bum (4,033 m), Nikhu Razi (4,059 m), and Pumtang Razi (3,881 m). The long mountain chains and diverse environment have formed a remarkably scenic landscape in this region.

Criterion (ix.) Significant ecological and biological processes

The area to the east of the N'Mai Hka River is the fault line where the Indian plate and the Eurasian plate collided. Rising to more than 6,000 meters from 200 meters, the multifaceted geography, climate and elevational gradients of the north-south mountains encompass a variety of ecosystems and thus enable many different plant and animal species or communities to evolve and thrive, making these mountains at the transition zone of three biodiversity hotspots, two ecoregions, and two endemic bird areas. Vegetation of the mountains varies from lowland subtropical rainforests to high altitude moist conifer forests and alpine shrubs, harboring high endemism and diversity that showcase evolutionary and ecological processes. Taking into account of the national borders, it also indicates the importance for transboundary collaboration in the area for the conservation of regional biodiversity and ecosystem integrity at a landscape scale.

Criterion (x.) Significance of natural habitats for biodiversity

The area east of the N'Mai Hka River is a junction zone of different biogeographical regions of fauna and flora, from tropical to alpine, culminating in extraordinary environmental diversity, inter- and intra-species variation and species richness. Biodiversity surveys show that forests to the east of the N’Mai Hka River have irreplaceable conservation value. More than 5,000 seed plant species, eight primate species and a variety of Asian birds and mammals have been recorded in the area (Li Hen, pers. comm.; Momberg et al., 2010; Geissmann et al., 2013). Examples of rare or threatened wildlife recorded in the region include

Muntiacus gongshanensis, *Cuon alpinus*, *Ailurus fulgens*, *Helarctos malayanus*, *Neofelis nebulosa* and *Manis pentadactyla*. Overall, the montane forests in east of the N'Mai River has the highest level of biodiversity in the eastern Himalayas and Asia (Table S5).

Table AV.1 Flora and fauna species distributed to the east of the N'Mai Hka River

Mammals	CR: *Rhinopithecus strykeri*; *Manis pentadactyla*
	EN: *Trachypithecus shortridgei*; *Trachypithecus phayrei*; *Hoolock tianxing*; *Cuon alpinus*; *Ailurus fulgens*;
	VU: *Nycticebus bengalensis*; *Macaca leonina*; *Macaca arctoides*; *Ursus thibetanus*; *Helarctos malayanus*; *Neofelis nebulosa*; *Budorcas taxicolor*;
	NT: *Macaca assamensis; Elaphodus cephalophus*; *Capricornis milneedwardsii*; *Capricornis rubidus*;
	LC: *Macaca mulatta*; *Prionailurus bengalensis*; *Martes flavigula*; *Prionailurus bengalensis*; *Muntiacus muntjak*; *Hystrix brachyura*; *Atherurus macrourus*; *Prionodon pardicolor*; *Sus scrofa*
	DD: Muntiacus gongshanensis;
Birds	CR: *Ardea insignis*
	EN:
	VU: *Tragopan blythii*; *Sitta formosa*; *Lophophorus sclateri*; *Aceros nipalensis*
	NT: *Sitta yunnanensis*
	LC: *Arborophila rufogularis*; *Pernis ptilorhynchus*; *Buteo rufinus*; *Ictinaetus malayensis*; *Streptopelia orientalis*; *Treron apicauda*; *Ducula badia*; *Cuculus canorus*; *Cuculus micropterus*; *Hierococcyx bocki*; *Glaucidium brodiei*; *Megalaima asiatica; Megalaima virens*; *Sasia ochracea*; *Dendrocopos darjellensis*; *Blythipicus pyrrhotis*; *Psarisomus dalhousiae*; *Pteruthius melanotis*; *Coracina macei*; *Coracina melaschistos*; *Pericrocotus brevirostris*; *Pericrocotus ethologus*; *Oriolus tenuirostris*; *Oriolus traillii*; *Tephrodornis pondicerianus*; *Dicrurus aeneus*; *Dicrurus leucophaeus*; *Dicrurus macrocercus*; *Dicrurus paradiseus*; *Dicrurus remifer*; *Urocissa flavirostris*; *Dendrocitta formosae*; *Corvus levaillantii*; *Lanius tephronotus*; *Aethopyga saturata*; *Arachnothera magna*; *Chloropsis hardwickii*; *Motacilla cinerea*; *Certhia hodgsoni*; *Sitta cinnamoventris*; *Sitta himalayensis*; *Muscicapa ferruginea*; *Ficedula albicilla*; *Ficedula westermanni*; *Cyornis unicolor*; *Tarsiger indicus*; *Enicurus immaculatus; Parus monticolus*; *Parus spilonotus*; *Sylviparus modestus*; *Culicicapa ceylonensis*; *Pycnonotus flavescens*; *Pycnonotus jocosus*; *Alophoixus flaveolus*; *Ixos mcclellandii*; *Hypsipetes leucocephalus*; *Cecropis daurica*; *Aegithalos concinnus*; *Abroscopus schisticeps*; *Cettia major*; *Tesia castaneocoronata*; *Lioparus*

	chrysotis; *Yuhina castaniceps*; *Yuhina bakeri*; *Yuhina diademata*; *Yuhina flavicollis*; *Yuhina gularis*; *Alcippe nipalensis*; *Pomatorhinus ochraceiceps*; *Gampsorhynchus rufulus*; *Pseudominla castaneceps*; *Garrulax leucolophus*; *Trochalopteron chrysopterum*; *Chrysominla strigula*; *Heterophasia picaoides*; *Malacias gracilis*; *Malacias pulchellus*; *Actinodura egertoni*; *Actinodura waldeni*; *Prinia superciliaris*
	DD: *Chelidorhynx hypoxantha*; *Muscicapa strophiata*; *Muscicapa strophiata*; *Suthora beaulieui*
Reptiles	EN: *Cuora mouhotii; Kachuga trivittata*
	VU: *Manouria impressa*
	LC: *Lycodon subcinctus*; *Calamaria pavimentata*; *Oligodon cinereus*; *Varanus nebulosus; Draco maculatus*; *Sphenomorphus indicus*; *Bungarus magnimaculatus*; *Bungarus multicinctus*; *Hebius modestum*; *Oligodon cinereus*; *Ovophis monticola*; *Protobothrops mucrosquamatus*; *Pseudoxenodon macrops*; *Rhabdophis leonardi*; *Xenopeltis unicolor*; *Cryptelytrops albolabris*; *Rhabdophis nuchalis*; Cylindrophis ruffus; *Dendrelaphis gorei*; *Oligodon cyclurus*; *Plagiopholis blakewayi*; *Rhabdophis subminiatus*; *Sibynophis collaris*; *Sinonatrix percarinata*; *Viridovipera yunnanensis*;
	DD: *Draco blanfordii*; *Calotes kingdonwardi*; *Eutropis long*icaudata; *Ptyctolaemus gularis*; *Draco blanfordii*; *Japalura hamptoni*; *Scincella doriae*; *Hemiphyllodactylus yunnanensis*; *Salea kakhienensis*; *Cyclophiops doriae*; *Rhabdops bicolor*; *Bungarus bungaroides*; *Protobothrops kaulbacki*; *Rhabdophis himalayanus*; *Azemiops feae*; *Blythia reticulata*; *Viridovipera medoensis;*
Amphibians	EN: *Nanorana yunnanensis*
	VU: *Scutiger gongshanensis*
	NT: *Nanorana arnoldi*; *Bufo pageoti*; *Amolops viridimaculatus*
	LC: *Chirixalus nongkhorensis*; *Limnonectes kuhlii*; *Ingerophrynus macrotis*; *Occidozyga martensii*; *Hydrophylax leptoglossa*; *Limnonectes hascheanus*; *Kurixalus verrucosus*; *Rhacophorus feae*; *Raorchestes parvulus*; *Megophrys parva*; *Rhacophorus maximus*; *Amolops bellulus*; *Leptobrachium chapaense*; *Megophrys feae*; *Megophrys major*; *Odorrana andersonii*; *Polypedates mutus*; *Amolops kaulbacki*; *Amolops afghanus*; *Rhacophorus bipunctatus*; *Leptobrachium huashen*; *Rhacophorus rhodopus*; *Megophrys glandulosa*; *Hylarana margariana*; *Microhyla heymonsi; Microhyla berdmorei*; *Sylvirana nigrovittata*;
	DD: *Rhacophorus taronensis*; *Philautus tytthus*; *Nanorana yunnanensis*
Plants:	Whole Gaoligong Mountains harbors 382 local endemic seed plant species, 235 species of Ericaceae and 364 species of Orchidaceae
Reference: Momberg et al., 2010; Geissmann et al., 2013; Fan et al., 2017; IUCN, 2018-1; Wogan et al., 2008	

Statements of Integrity

The area from the N'Mai Hka River to the China-Myanmar border is a large expanse of extensive and intact forests and provides opportunities for transboundary conservation initiatives. Threats to this region include dam building, hunting, non-timber forest product collection, shifting cultivation, commercial logging and mining concessions. Such threats endanger the long-term integrity of forest habitats in the area (Meyer et al., 2017). However, much of the area is still untouched due to inaccessibility, difficult terrain, limited human population density and limited human activities. Commercial logging has been banned in this area since 2016 (Meyer et al., 2017), which will help to enhance the integrity of forest ecosystems of this area.

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