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Front cover: The bonnet macaque *Macaca radiata*. Two mothers and their infants on the roadside in the north of the species' range in Goa, India. Photo by Asmita Sengupta. See page 109.

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Primate Surveys in the Marañón-Huallaga Landscape, Northern Peru with Notes on Conservation

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Abstract: With about 50 species and subspecies, the diversity of primates in Peru is amongst the highest in the world. Primate field surveys, however, are still scarce for many areas. Peru's northern forests lie at the heart of the Tropical Andes and are home to four of the country's endemic primates: *Aotus miconax*, *Callicebus oenanthe*, *Oreonax flavicauda*, and *Saguinus leucogenys*. We present here an overview of results from more than five years of primate surveys in northern Peru. Surveys were carried out in the valleys and intervening highlands between the ríos Marañón and Huallaga in the departments of Amazonas, La Libertad, Huánuco and San Martín. Data were gathered between March 2007 and October 2012 from 36 locations at 26 sites. Surveys were carried out as part of biological inventories in proposed conservation areas. In all, 14 species were recorded from 11 genera. The presence or absence of a species was largely determined by altitude. The four endemic species were present in the majority of survey sites. The most common species encountered were *Oreonax flavicauda*, *Ateles belzebuth*, *Cebus yuracus*, and *Aotus miconax*. The highest species diversities were found at lower altitudes further from human development, although some species previously registered in these regions were not encountered. All areas visited suffered from at least some anthropogenic disturbance, with the majority showing high levels of habitat disturbance, deforestation and hunting.

Keywords: Río Marañón, Río Huallaga, altitude, cloud forest, deforestation, conservation

Resumen: Con aproximadamente 50 especies y subespecies, Perú posee una de las mayores diversidades de primates entre todos los países del mundo, pero los estudios de campo son escasos en muchas áreas. Los bosques del Norte del Perú se encuentran en el corazón de los Andes Tropicales y son el hogar de cuatro especies de primates endémicos. Presentamos una visión general de los resultados de más de cinco años de trabajo de campo con primates en el Norte del Perú. Los trabajos fueron realizados en los valles y partes altas entre los ríos Marañón y Huallaga, en los Departamentos de Amazonas, La Libertad, Huánuco y San Martín. Los datos fueron reunidos entre marzo del 2007 y octubre del 2012 en 36 localidades de 26 sitios. Los trabajos de campo fueron realizados como parte de inventarios biológicos llevados a cabo durante la creación de propuestas para áreas de conservación. Un total de 14 especies fueron registradas para 11 géneros. Altura fue el factor más importante para la presencia de especies. Las cuatro especies endémicas del Perú fueron encontradas en la mayoría de los lugares de trabajo. Las especies comúnmente más encontradas fueron; *Oreonax flavicauda*, *Ateles belzebuth*, *Cebus yuracus* y *Aotus miconax*. La más alta diversidad de especies fue registrada en los lugares de menor altitud en bosques alejados de impacto antropogénico. Algunas de las especies que fueron previamente registradas en estas regiones no fueron encontradas. Todas las áreas visitadas durante el estudio presentan algún tipo de disturbio antropogénico, en la mayoría se puede observar destrucción de hábitat, deforestación y cacería.

Palabras Clave: Río Marañón, Río Huallaga, altitud, bosque montano, deforestación, conservación

Introduction

Peru is one of the most biodiverse countries in the world in terms of mammals, birds, amphibians and plants (Pacheco *et al.* 2009; Rodríguez and Young 2000; Schulenberg *et al.* 2010). Aquino and Encarnación (1994) carried out the only recent comprehensive revision of the primates of Peru, listing 32 species. Following taxonomic revisions since then, 44 species and 50 subspecies are now recognized by the IUCN/SSC Primate Specialist Group; as such, the sixth highest primate diversity of any country, following only Brazil, Madagascar, Indonesia, the Democratic Republic of the Congo (DRC), and Colombia (Mittermeier 2013). Despite this, field studies of primates in Peru are still lacking for many areas and species, and the distributional limits of most species are as yet poorly understood.

Peru's northern regions of Amazonas, La Libertad, Huánuco and San Martín present a great variety of ecosystems in a relatively small area (Peru, ONREN 1976; Brack Egg 1986). This ecological complexity comes from altitudinal gradients, local and general rain shadow effects in inter-Andean valleys, and the influence of the dry coastal climate, which penetrates further inland here than anywhere else in Peru (Rodríguez and Young 2000). This complexity is reflected in the diversity of species present; able to colonize new areas by the reduced size of rivers near their headwaters and, in lowland areas, to migrate along the valleys of the ríos Marañón and Huallaga and their tributaries.

Of Peru's 44 recognized primate species, six are endemic to Peru (Matauschek *et al.* 2011; Mittermeier 2013). Of these, four—the yellow-tailed woolly monkey (*Oreonax flavicauda*), the Andean night monkey (*Aotus miconax*), the Andean titi monkey (*Callicebus oenanthe*), and the saddle-back tamarin (*Saguinus leucogenys*)—are endemic to the north of the country. They are all found on the eastern slopes of the Andes in the regions of Amazonas, La Libertad, Huánuco and San Martín (Leo Luna 1987; Buckingham and Shanee 2009; Schjellerup *et al.* 2009) an area at the heart of the 'Tropical Andes Biodiversity Hotspot' (Myers *et al.* 2000; Myers 2003). *Saguinus leucogenys* has a larger distribution, found as far south as the region of Pasco (Aquino and Encarnación 1994).

Due to their inaccessibility and steep terrain, the forested areas between the ríos Marañón and Huallaga remained relatively undisturbed until recent decades (Leo Luna 1987; Buckingham and Shanee 2009; Schjellerup *et al.* 2009). A surge in immigration and development projects since the 1970's, however, has opened many new frontiers in deforestation (Dourojeanni *et al.* 2009). Sustained deforestation rates are particularly high because the low productivity of montane soils and slow adaptation of migrants to new agricultural conditions has led to a reliance on unsustainable farming methods (Bebington 1990; Loker 1996; MINAG 2010; N. Shanee 2012a).

We conducted rapid biological inventories at sites throughout Amazonas and San Martín and neighboring areas of La Libertad and Huánuco. Our surveys were part of a broader community conservation initiative in this landscape

(<http://www.neoprimate.org/index.php/en/projects-npc>). We visited areas during surveys for the creation of private, communal or government protected areas, or in areas where baseline data were needed for community conservation work to evaluate possibilities and the need for conservation.

Methods

Study area

We conducted surveys of primates between March 2007 and October 2012. All sites were in areas on the eastern side of the Río Marañón valley, the western side of the Río Huallaga valley and the intervening highlands in the regions of Amazonas, La Libertad, Huánuco and San Martín, between 05°31'S and 09°44'S and 78°39'W and 76°15'W (Fig. 1). We surveyed forested areas between 300 m and 2900 m above sea level. Habitat types we surveyed included montane and pre-montane cloud forests, lowland *terra firma* forests, shade coffee plantations, and the dry forests of the Marañón and Huallaga valleys (Table 1). Daytime temperatures for these

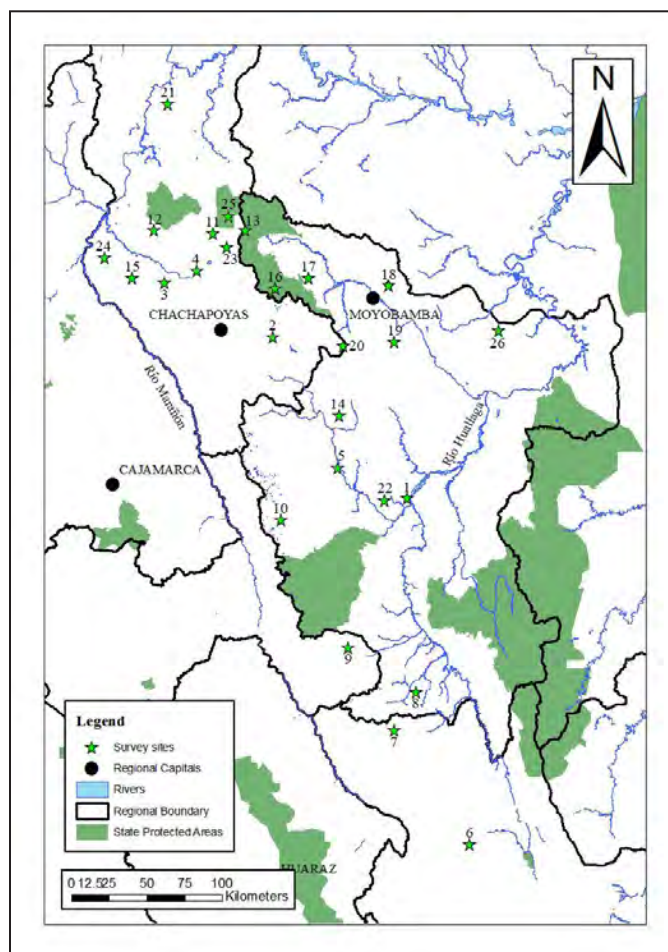


Figure 1. The study area showing survey sites, state protected areas, major rivers and political boundaries. 1 - Pucuncho; 2 - Ocol; 3 - Berlin; 4 - Shipasbamba; 5 - Simacache; 6 - Monzon; 7 - Ajenco; 8 - Shunte; 9 - Poroto; 10 - Alto Huayabamba; 11 - Yambrasbamba; 12 - Copallín; 13 - Venceremos; 14 - Paujil; 15 - Delta; 16 - Bosque Protección Alto Mayo; 17 - Colca; 18 - Nuevo Paraiso; 19 - Gira-Sisa; 20 - Paitoja; 21 - Halcabamba; 22 - Pachiza; 23 - Corosha; 24 - Campo redondo; 25 - Río Nieva; 26 - Cordillera Escalera.

areas vary from 8°C to 36°C. Average monthly rainfall ranges from approximately 15 mm to 1500 mm.

Field Surveys

We expected to encounter 18 species of primate previously recorded in the area. We used methods developed for

rapid biological inventories that have been used successfully in Peru in previous studies (for example, S. Shanee 2011b; Vriesendorp *et al.* 2004). Sites varied in size from isolated forests of about 400 ha to areas of contiguous forest, as well as areas of forest mosaics with patches of <10 ha. Data were gathered using a combination of field surveys and

Table 1. Survey site locations and habitat types.

Survey site	Department	Coordinates	Altitude (m)	Habitat type	Level of anthropogenic disturbance	Conservation status
Pucunhucho	San Martín	07°13'46"S 76°45'22"W	300–400	Huallaga semi-arid forest	Advanced secondary re-growth	Private Conservation Area (23.5 ha), in process
Ocol	Amazonas	06°16'04"S 77°33'27"W	2200–2300	Montane palm forest	Highly fragmented	Private Conservation Area (16 ha), awarded
Berlin	Amazonas	05°55'08"S 78°24'45"W	2000–2200	Montane cloud forest	Highly fragmented	Private Conservation Area (59 ha), in process
Shipasbamba	Amazonas	05°54'35"S 77°58'50"W	2000–2300	Montane cloud forest	Moderate disturbance and hunting	Not protected
Simacache	San Martín	07°2'39"S 77°11'59"W	200–500	Primary rainforest	Moderate hunting	Conservation Concession (51,269), in process
Monzon*	Huánuco	09°15'29"S 76°23'36"W	700–1500	Pre-montane cloud forest	Low disturbance and moderate hunting	Not protected
Ajenco*	Huánuco	08°39'06"S 76°47'28"W	1800–2100	Montane cloud forest	Low disturbance and moderate hunting	Not protected
Shunte*	San Martín	08°24'40"S 76°43'07"W	1200–1700	Montane cloud forest	Low disturbance and moderate hunting	Regional Conservation Area, in process
Poroto*	La Libertad	08°07'02"S 77°08'8"W	1700–1900	Montane cloud forest	Moderate disturbance and hunting	Not protected
Alto Huyllabamba*	San Martín	07°19'14"S 77°27'39"W	2200–2600	Montane cloud forest	Low disturbance and moderate hunting	Conservation Concession (143,928 ha), awarded
Yambrasbamba*	Amazonas	05°39'56"S 77°54'36"W	1700–2100	Montane cloud forest	Highly fragmented and Little hunting	Private Conservation Area (2,776 ha), in process
Copallin	Amazonas	05°38'30"S 78°15'03"W	1700–2500	Montane cloud forests	Low disturbance and little hunting	Private Conservation Area (11,549 ha), awarded
Venceremos	San Martín	05°40'18"S 77°45'52"W	1800–2000	Montane cloud forest	Low disturbance and moderate hunting	Alto Mayo Protected Forest (182,000 ha).
Paujil*	San Martín	06°42'28"S 77°13'29"W	1600–1800	Pre-montane cloud forest	Moderate disturbance and heavy hunting	Conservation Concession (7,418 ha), in process
Delta	Amazonas	05°47'57"S 78°34'16"W	600–800	Maranon dry forest	Fragmented and moderate hunting	Conservation Concession (423 ha), Awarded
Bosque Protección Alto Mayo	San Martín	05°57'43"S 77°35'38"W	2500–2800	Montane cloud forest	Low disturbance no hunting	Alto Mayo Protected Forest (182,000 ha)
Colca	San Martín	05°53'40"S 77°23'15"W	1700–1900	Pre-montane cloud forest	Moderate disturbance and hunting	Not protected
Nuevo Paraiso	San Martín	05°57'44"S 76°57'16"W	1200–1500	Lowland rainforest	Highly fragmented and heavy hunting	Not protected
Gira-Sisa*	San Martín	06°17'34"S 76°54'24"W	100–1700	Pre-montane cloud forest	Moderate disturbance and hunting	Conservation Concession (c.3,000 ha), in process.
Paitoja	San Martín	06°21'42"S 77°04'52"W	1600–1900	Pre-montane cloud forest	Moderate disturbance and hunting	Not protected
Nuevo Halcabamba	Amazonas	04°52'59"S 78°12'24"W	200–300	Lowland rainforest	Moderate disturbance and heavy hunting	Not protected
Pachiza*	San Martín	07°14'20"S 76°53'25"W	500–1200	Lowland seasonal moist forest	Low disturbance and little hunting	Two Conservation Concessions (5,768 ha), in process
Corosha	Amazonas	05°47'43"S 77°47'12"W	2100–2500	Montane cloud forest	Minimal disturbance no hunting	Private Conservation Area (2,282 ha), awarded
Campo Redondo	Amazonas	06°13'00"S 78°19'47"W	1200–1500	Shade coffee plantations	100% altered environment	Not protected
Rio Nieva	Amazonas	05°34'31"S 77°50'54"W	1500–1900	Montane cloud forest	Moderate disturbance and hunting	Reserved Zone (36,348 ha)
Cordillera Escalera*	San Martín	06°27'45"S 76°17'23"W	900–1500	Pre-montane cloud forest	Low disturbance and little hunting	Regional Conservation Area (149,870 ha)

* Sites where surveys were carried out in more than one location; for these sites coordinates represent an area between survey locations.

key-informant interviews. We surveyed areas using existing trail systems and, when necessary, purpose-cut trails. We walked trails accompanied by local residents as guides. The cutting of new trails was kept to a minimum to limit forest disturbance. Inventory trips took between three and seven days. The location of all sites was recorded with a handheld GPS, as were points of visual, audio or incidental detection (for example, food residues showing clear bite marks). We never inferred a species' presence from bite marks without additional secondary evidence. We also collected additional data on threats to habitat in each area.

We collected secondary data on species occurrence from local informants. Species identification was made during individual and group interviews. We used photographs and drawings of the various primate species that may have been present in each area to help identification. Positive identifications were cross referenced between several informants and

we asked further details of behavior, diet, and locomotion to ensure correct identification. We also collected additional information on hunting practices, forest resource use and anthropological disturbance such as deforestation, logging, burning and contamination to better assess threats and for planning of conservation initiatives.

Results

We surveyed 36 locations, totalling 213 days of field surveys. Due to their proximity, the results of 10 of them were combined, giving 26 localities in all. The habitat types visited and the presence of primate species was recorded at all sites. Fourteen species were recorded from 10 genera in eight habitat types (Table 1); 12 species were recorded by direct observation, an additional two through interviews (Table 2).

Table 2. Species records.

Survey site	Species encountered	Recorded from secondary evidence
Pucuncho	<i>Saguinus leucogenys</i> ; <i>Callicebus oenanthe</i>	-
Ocol	<i>Aotus miconax</i>	<i>Cebus yuracus</i> ; <i>Ateles</i> sp.; <i>Oreonax flavicauda</i>
Berlin	<i>Cebus yuracus</i> ; <i>Aotus miconax</i> ; <i>Oreonax flavicauda</i>	-
Shipasbamba	<i>Oreonax flavicauda</i> ; <i>Aotus miconax</i>	<i>Cebus yuracus</i> ; <i>Ateles belzebuth</i>
Simacache	<i>Cebuella pygmaea</i> ; <i>Saguinus leucogenys</i> ; <i>Cebus yuracus</i> ; <i>Aotus nigriceps</i> ; <i>Callicebus cupreus</i> *; <i>Callicebus oenanthe</i> ; <i>Alouatta seniculus</i>	<i>Saimiri macrodon</i> ; <i>Sapajus macrocephalus</i> ; <i>Aotus</i> sp.; <i>Pithecia</i> sp.
Monzon	<i>Cebus yuracus</i>	<i>Callicebus discolor</i>
Ajenco	-	<i>Cebus yuracus</i> ; <i>Oreonax flavicauda</i>
Shunte	<i>Cebus yuracus</i>	<i>Callicebus discolor</i> ; <i>Aotus</i> sp.; <i>Oreonax flavicauda</i>
Poroto	-	<i>Aotus</i> sp.; <i>Oreonax flavicauda</i> ;
Alto Huayabamba	<i>Oreonax flavicauda</i>	<i>Cebus yuracus</i> ; <i>Ateles</i> sp.; <i>Aotus</i> sp.
Yambrasbamba	<i>Cebus yuracus</i> ; <i>Aotus miconax</i> ; <i>Ateles belzebuth</i> ; <i>Oreonax flavicauda</i>	<i>Alouatta seniculus</i>
Copallín	<i>Ateles belzebuth</i> ; <i>Oreonax flavicauda</i>	<i>Cebus yuracus</i> ; <i>Aotus miconax</i>
Venceremos	<i>Cebus yuracus</i> ; <i>Oreonax flavicauda</i>	
Paujil	<i>Cebus yuracus</i> ; <i>Aotus miconax</i> ; <i>Alouatta seniculus</i> ; <i>Ateles belzebuth</i>	<i>Saguinus leucogenys</i> ; <i>Saimiri macrodon</i>
Delta	<i>Aotus</i> sp.	<i>Cebus yuracus</i>
Bosque Protección Alto Mayo	-	<i>Aotus</i> sp.; <i>Oreonax flavicauda</i>
Colca	-	<i>Oreonax flavicauda</i>
Nuevo Paraiso	<i>Saimiri macrodon</i>	<i>Saguinus leucogenys</i> ; <i>Sapajus macrocephalus</i> ; <i>Cacajao calvus</i> **;
Gira-Sisa	<i>Callicebus oenanthe</i> ; <i>Aotus nigriceps</i>	<i>Saguinus leucogenys</i> ; <i>Saimiri macrodon</i> ; <i>Cebus yuracus</i> ; <i>Alouatta seniculus</i>
Paitoja	<i>Oreonax flavicauda</i>	<i>Cebus yuracus</i> ; <i>Aotus</i> sp.
Halcabamba	-	<i>Saguinus leucogenys</i> ; <i>Saimiri macrodon</i>
Pachiza	<i>Saguinus leucogenys</i> ; <i>Callicebus oenanthe</i> ; <i>Aotus nigriceps</i> ; <i>Alouatta seniculus</i>	<i>Saimiri macrodon</i> ; <i>Cebus yuracus</i> ; <i>Ateles</i> sp.
Corosha	<i>Aotus miconax</i> ; <i>Oreonax flavicauda</i>	<i>Cebus yuracus</i>
Campo redondo	<i>Aotus miconax</i>	<i>Cebus yuracus</i> ; <i>Ateles belzebuth</i> ; <i>Oreonax flavicauda</i>
Río Nieva	<i>Aotus miconax</i>	<i>Cebus yuracus</i> ; <i>Alouatta seniculus</i> ; <i>Ateles belzebuth</i> ; <i>Oreonax flavicauda</i>
Cordillera Escalera	<i>Saguinus leucogenys</i> ; <i>Aotus</i> sp.	<i>Cebus yuracus</i> ; <i>Ateles</i> sp.; <i>Oreonax flavicauda</i>

* Registered on south bank of the Río Huallaga. **Record from local informant

The most commonly seen in montane and pre-montane forests were *Oreonax flavicauda* and *Aotus miconax*. Both species were found in the majority of sites above 1500 m. *Oreonax flavicauda* was not found in *Podocarpus*-dominated or white-sand forests in central Amazonas, whereas *Aotus miconax* was found at all montane sites including *Podocarpus*-dominated and white-sand forests, and *Alzatea verticillata*-dominated forests in the Utcubamba valley south of Bagua Grande in Amazonas. No large species were found in the dry or semi-arid forests of the Utcubamba and Marañón valleys in Amazonas or central Huallaga Valley in San Martín. The most diverse primate assemblage was found at Simacache, an area of lowland *terra firma* forest in San Martín (Tables 1 and 2), where seven species were seen, with an additional four recorded from interviews. The least diverse (a single species) was in the Área de Conservación Privada Ocol, Amazonas; a large fragment of montane palm forest (Tables 1 and 2).

Important records of range extensions were obtained for *Cebuella pygmaea*, which was found in Simacache, about 100 km west of its previously recorded range, and crossing the Río Huallaga. Similarly, *Cacajao calvus* was recorded from interviews far from its recorded distribution, only the second record of this species this far west, in San Martín (Tello-Alvarado *et al.* 2012). Neither of these species was recorded in intervening areas. We also recorded the presence of *Aotus miconax* in Campo Redondo, Amazonas, at an elevation of 1300 m, below its expected altitudinal range. *Ateles belzebuth* and *Cebus yuracus* were commonly found at high elevations in montane cloud forest. *Ateles belzebuth* was widely distributed at sites in central Amazonas from the ACP Copallín east to Yambrasbamba and south along the border between Amazonas and San Martín. This species was reported for both high and low elevation sites but seems to be locally more common at high elevations. *Cebus yuracus* was found at the same high elevation sites as *A. belzebuth* but was also found much further to the south in Monzon, Huánuco. Of the 14 species encountered, six are categorized as threatened, either by the IUCN or under Peruvian law.

Many local informants reported that primate populations are receding farther and farther from human settlements, requiring several hours walking in order to find them. They blamed this on hunting and deforestation. Many primates have survived in fragments; we found groups of *O. flavicauda*, *A. miconax*, *C. oenanthe* and *S. leucogenys* surviving and reproducing in small patches, <3 ha, where small-scale logging for firewood was still practiced.

No areas we surveyed were free from current anthropogenic disturbance. At least low-level hunting was encountered at all sites except Venceremos, in the Alto Mayo Protected Forest (Table 1). Habitat disturbance was omnipresent except at Simacache (Table 1). The hunting of primates in areas we surveyed was usually for subsistence with just occasional commercial sale as bushmeat, in the local and national pet trade, or as tourist attractions. In some areas we found increasing instances of hunting as a control measure against

crop raiding, particularly for *Cebus yuracus*, *Sapajus macrocephalus* and *O. flavicauda* in corn fields and fruit plantations.

In the highlands, deforestation is mainly for cattle ranching and small-scale agriculture. Hillsides are burnt to clear the area for farming or in the belief that smoke from fires promotes rainfall. Mining concessions were also abundant in the highlands. Mining is a potential threat to primates through deforestation and pollution, but the majority of concessions were still in exploratory stages, and the full extent of their impact is hard to assess. In lowland and dry-forest areas, conversion for monocultures such as oil palm, rice, cacao and exotic timber trees were the main causes of deforestation.

The majority of human populations in the areas surveyed were migrants. In many cases, movements were fuelled by land trafficking, with new settlements established up to 15-hours walk from the nearest access route. Migrants were generally impoverished and relied entirely on natural resources for subsistence. Pioneer settlers often solicited regional authorities to construct roads to connect them to the main highways. These small roads attract more migrants to the area, resulting in further loss and fragmentation of primate habitat.

The number of protected areas in the region increased considerably over the five years of the study period. Of the sites we surveyed, 14 (15 including the Alto Mayo Protected Forest – BPAM) are now afforded some level of protection (Table 1). New protected areas include one state protected area, the Río Neiva Reserved Zone, six Private Conservation Areas (ACP), and five Conservation Concessions (CC), officially awarded or in process, and one new Regional Conservation Area (ACR). Together these areas cover 553,331 ha. Nine areas are still unprotected, with only one, Campo Redondo, the site of a possible new conservation project. A further area, Gira-Sisa, is considered here as a conservation concession although officially the area is designated as a reserved zone by the regional government of San Martín. It has received no on-the-ground protection and has been invaded by farmers and loggers. Informal landscape-level conservation initiatives were also found to be common, resulting from direct promotion and conservation education efforts of NGOs and grassroots organizations. These were most commonly in the form of rural villagers controlling deforestation and the provision of protection for endangered species through local institutions.

Discussion

Peru has the sixth highest primate diversity of any country, and, correspondingly, regions with extraordinarily rich primate communities, such as Manu National Park with 13 species (Terborgh *et al.* 1984) and the Pacaya Samiria National Reserve with 12 (Neville *et al.* 1976). Overall, we recorded 14 species during our surveys in north-eastern Peru, and in one site, Simacache, we found evidence of 11. The rich primate communities we encountered are probably allied to the diversity of vegetation types in the areas between the ríos Marañón and Huallaga: *terra firma* rain forest in the lowland areas of San Martín and northern Amazonas, pre-montane

and montane cloud forests of the intervening highlands, and the unique Huallaga, Marañón and Utcubamba dry forests. Altitudes range from 200 m to over 4000 m (above the snow line). Primates were also recorded in secondary forest and timber and shade-coffee plantations.

We failed to see three species expected for the region. *Lagothrix poeppigii* is undoubtedly heavily hunted in most areas we visited (see Peres 1990; Bodmer *et al.* 1997). *Aotus nancymae* and *Saimiri boliviensis peruviensis* have been reported for San Martín, but it appears that both are limited to the east of the Río Huallaga and south of the Río Marañón as indicated by Hershkovitz (1983, 1984) and Aquino and Encarnación (1994). In the Comunidad Campesina de Leymebamba on the border of Amazonas and San Martín, *A. belzebuth* and *A. chamek* have been reported to be sympatric, even occurring in mixed species groups (Barrio and Dignum 2003), but this may have been due to the misidentification of the darker pelage of some *A. belzebuth* populations that are confused with the black *A. chamek* (H. Dignum pers. comm.).

Human population growth in the area is among the highest in Peru (Peru, INEI 2007). The majority of migrants arrive from Cajamarca, where mining concessions occupy almost 50% of the region (calculated from Peru, MEM 2011). This has resulted in increasing land prices, social instability, and a lack of drinking water and fertile land (Bury 2007; N. Shanee 2012a). Likewise, the proliferation of large-scale monocultures in the lowlands forces migration into new areas, generally into steeper areas (S. Shanee 2011). Migrants are usually poor, and use unsuitable farming methods (Bebbington 1990; N. Shanee 2011a). Illicit coca (*Erythroxylum coca*) cultivation is also a cause of deforestation in the central and southern portions of the survey areas. Deforestation from illicit crops is augmented by the effects of the control measures (herbicides and burning) used against the production of cocaine. In some areas of San Martín and Huánuco this has become the major cause of deforestation (Young 1996; Fjeldså *et al.* 2005).

Hunting in northern Peru varies greatly between sites. Indigenous groups, which generally occupy lowland areas, hunt heavily for subsistence, while migrants generally hunt opportunistically (N. Shanee 2012b). Live animal and bushmeat trades exist mainly in the lowland areas of Amazonas and San Martín. The smaller species are often sent to the coast, but the larger, more endangered monkeys are generally kept in Amazonas and San Martín to be sold as bushmeat or as pets and tourist attractions (Pautrat 2002; Altherr 2007; N. Shanee 2012b). In many of the areas surveyed, especially the lowlands, the larger primates were relatively scarce; a common phenomenon in the Amazon (Bodmer *et al.* 1997; Chapman and Onderdonk 1998; Robinson and Bennett 2000; Peres 2001; Jerozolimski and Peres 2003). *Ateles belzebuth*, a predominantly lowland species (S. Shanee 2009), was more commonly found at high elevation sites during our surveys probably because of greater levels of habitat disturbance and hunting in the lowlands.

Peru is one of the countries facing serious challenges from climate change, with rural areas being the most

vulnerable (Brooks and Adger 2003; UNFCCC 2007). Tropical montane cloud forests are considered amongst the most vulnerable to climate change of all tropical forests systems (Markham 1998; Still *et al.* 1999; Bubb *et al.* 2004). Changes in climate described by the local residents in the areas that we visited could be among the biggest threats to the primates of the Marañón-Huallaga basins. Predictions of climate change effects, some of which have already been observed by local residents that could further endanger Andean primates include: a significant up-slope shift in species and ecosystems; reduced cloud immersion by vegetation; recurring droughts and extreme rains; intensification of wildfires; and the alteration of plant phenologies, affecting reproduction of animal and plant species and consequently food availability (Bawa and Dayanandan 1998; Corlett and Lafrankie 1998; Chapman and Peres 2001; McCarty 2001; Bubb *et al.* 2004; Lenoir *et al.* 2008; Fisher 2011). The increasing effects of climate change and land degradation also increase pressure on farmers, leading to more land clearance, thus creating a positive feedback loop (Laurance and Williamson 2001; Young and Lipton 2006; N. Shanee 2012a).

Protected areas when small, few and isolated, may often contribute little to species conservation and must be complemented by strategies for management at the landscape level (Harris 1984; Newmark 1987; Margules and Pressey 2000; Acrenaz *et al.* 2007). Private Conservation Areas (ACP) and Conservation Concessions (CC) are generally relatively small (Table 1), and therefore offer limited protection for primates, especially the larger species. Informal landscape-level conservation can, however, serve well in complementing the systems of formal protected areas by offering partial protection or having a role as buffer zones, and should be promoted throughout the north of Peru through local authorities, NGO projects and grassroots organizations.

Understanding species' distributions is a basic requirement for conservation planning. Good distribution maps are lacking for many of Peru's primate species, particularly in the eastern Andes. Existing maps of primate distributions often assume that major rivers and other geographical barriers delimit species distributions. Rivers in northern Peru mark the limits to the ranges of a number of species, for example *Callicebus oenanthe* and *C. cupreus*, but rivers are narrower in their upper reaches, reducing their effectiveness as barriers to dispersal (Ayres and Clutton-Brock 1992). In more mountainous areas, distributional limits are less defined by clear geographic features, and in many cases more complex environmental changes reflecting differences in elevation, soils, climate and floristic communities evidently form less tangible barriers, more difficult to discern. Field studies are needed to 1) define species' ranges more precisely in the eastern Peruvian Andes and 2) understand the physical, biological or ecological barriers involved. This will not only provide for more informed conservation planning for these species but will also contribute to the broader understanding of primate biogeography in general and in relation to anthropogenic environments.

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Strategies for the Conservation of Two Critically Endangered, Endemic Primates in Panama

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Abstract: Neotropical Primates are threatened because of deforestation and hunting. There are numerous species and subspecies of primates today restricted entirely to small forest patches. Many are poorly studied due to the remoteness of their ranges, political situations, or the expensive logistics required. Although population surveys are important to obtain baseline information for the conservation of threatened primates, there are opportunities where population censuses could be used in conjunction with rapid conservation strategies. In this report we encourage scientists to use collateral techniques that could positively impact local communities while undertaking scientific studies. They include the involvement of biology students from the target country, the use of informative material to divulge information on the natural history and conservation of the target species, and the creation of interest among community leaders so that a long-term conservation program can be established in the future.

Key Words: Primate conservation, Azuero Peninsula, Azuero howler monkey, Azuero spider monkey, environmental education, Panama

Resumen: Los Primates Neotropicales se encuentran en peligro debido a la deforestación y a la cacería. Existen diferentes sub-especies de primates hoy en día restringidas solo a pequeños parches de bosques. Muchas han sido poco estudiadas dado a su rango de distribución remoto, situaciones políticas, o al costoso equipo requerido. Aunque censos poblacionales son importantes para obtener información de línea base para la conservación de primates amenazados, existen oportunidades donde los censos poblacionales podrían ser llevados a cabo en conjunto con estrategias rápidas de conservación. En este reporte incentivamos a los investigadores a utilizar técnicas colaterales para un impacto positivo en las comunidades locales mientras realizamos estudios de investigación. Ello incluye la incorporación de estudiantes de biología de los países visitados, el uso de material informativo para divulgar información de la historia natural y la conservación de las especies estudiadas, y el incentivo de un interés a líderes de las comunidades, de manera que se puedan desarrollar programas de conservación a largo plazo en el futuro.

Palabras Claves: Conservación de primates, Península de Azuero, mono aullador de Azuero, mono araña de Azuero, educación ambiental, Panamá.

Introduction

The Azuero Peninsula of southwest Panama, Central America, is an extensively deforested zone that has three provinces: Herrera (with 27,202 human dwellings and 183 secondary schools); Los Santos (with 25,052 human dwellings and 163 secondary schools); and the east of Veraguas (with 49,102 human dwellings and 518 secondary schools) FAO (2002). This area is home to two subspecies of primates

endemic to Panama, the Azuero howler monkey (*Alouatta coibensis trabeata*) and the Azuero spider monkey (*Ateles geoffroyi azuerensis*). These subspecies are considered to be the two of the most endangered primates in Panama (Méndez-Carvajal 2011). Other primates such as the Panamanian white-throated capuchin (*Cebus imitator*) have been identified as at high risk regionally by the Fundación Pro-Conservación de los Primates Panameños (FCPP). This is due to factors such as very small remaining wild populations, their damaged

and fragmented habitats, regional endemism, and the lack of conservation plans (Jacobson *et al.* 2006; Méndez-Carvajal 2011). The primates are affected by exposure to crop spraying, the pet trade, agriculture, cattle ranching, hunting, mining, tourism, expansion of teak plantations, and, most recently, real estate investments (Méndez-Carvajal and Ruiz-Bernard 2010). The FCPP is a Panamanian non-profit organization that has initiated the first primate conservation plan for the Azuero Peninsula, monitoring the primate populations remaining in the area. The FCPP has developed an educational program to conserve the Azuero wildlife in accordance with previous conservation assessments (Horwich 1996; Rodríguez-Luna *et al.* 1996; Cowlishaw and Dunbar 2000; Rabinowitz 2003; Jacobson *et al.* 2006; Méndez-Carvajal *et al.* 2006). In this project, we attempt to maximize our approach any time we visit the study sites by involving local people, as they are the key to helping in the conservation of the native fauna of Azuero. We developed a questionnaire to evaluate each town surveyed ($n = 150$) and contacted community leaders to introduce our project and our main goal. The creation of good attitudes and practices in the local communities helps to mitigate the decline in primate populations living in forest fragments and the corridors of trees often extending from and between them (living fences) in the protected areas. To meet this goal we developed the following objectives:

- Provide information on the biology, ecological role and land use activities that affect the primates in their communities;
- promote the interest of the locals as volunteers for tracking and protecting the primates in their areas;
- involve Panamanian biologists interested in primates studies; and
- develop educational activities that can be used to evaluate the effectiveness of our presence in their areas.

The Azuero Peninsula

Land use

Cattle ranching first began in Panama in 1521, when the colonial administrator, Pedro Arias de Ávila, brought 50 cows from Jamaica (Gligo 1999; Castro 2004). In the middle of the 17th century, the people who lived in Panama City started to migrate west to start up ranches, and cattle ranching quickly became the main economic activity of the country (Heckadon-Moreno 1998, 2001). The Spaniards developed Azuero as one of the first production centers in Panama, growing rice, sugar cane, corn, pineapple and, most importantly, raising cattle. This was made easier by the flat terrain that was partially covered by savannas (Suárez 1981; Gligo 1999). Panamanians in this region also hunted wildlife for subsistence (Heckadon-Moreno 2001). Today, the Azuero Peninsula is a mixed landscape with extensive pastures and secondary

and remnant forests, as well as mature primary and secondary forests in the mountains. Some of the forests are protected as national parks, forest reserves, and other categories by the Panama National Environmental Authority (ANAM 1999). This interaction with the environment was a fundamental element of the lifestyles of the Azuerense (people native to the Azuero Peninsula), who are known today as outstanding hunters, farmers and cowboys. Traditional farming practices involve conserving native forest patches or “Chapas,” which serve as a resource for medicines, fruits and firewood, together with living fences (Brandaris 1983). “Living Fences” consist of specific trees that the campesinos use to support their needs, timber for construction, fruits, and shelter for cattle (Suárez 1981). The living fences have created effective corridors for the dispersal of wildlife between the forest patches (Méndez-Carvajal 2008). As new generations of Azuerenses are getting better education in careers not related to agriculture, most of the farming areas are now being abandoned, overused or sold to immigrants. Consequently other farm owners hire indigenous people to work on their lands, creating another pressure on the primates living in the area as they eat monkeys as part of their culture. Azuerenses are also looking for more land to expand and “develop” cattle ranches, and since the 1980s have been migrating to forested regions such as Colon Province, Chagres and Darien (eastern side of Panama). The impact of their regional culture is now threatening the remaining forested areas in Panama and the lack of environmental education campaigns in Azuero in the past years is resulting in them making the same mistake of extreme deforestation. FCPP thus considers Azuero to be in need of permanent exposure to conservation activities, as is true of other regions in Panama.

Environmental aspects

Environmental variables such as cold spells with extended periods of rain have a great influence on the Azuero primate populations. In addition to being susceptible to the cold spells, Azuero howlers are suffer from parasitism by the howler botfly (*Cuterebra baeri*) (Méndez-Carvajal and Ruiz-Bernard 2009). The most prolonged episodes of rain and cold are normally in October and November. They can be a cause of mortality in a number of species, including howler and spider monkeys, as has also been reported in Costa Rica (A. Bustamante pers. comm) (Fig. 1).

Methods

To evaluate the situation of the Azuero primates, the authors carried out surveys and interviews around the towns, areas of cattle ranching, rivers, and mountains that contain remnant forests (Méndez-Carvajal 2011). The survey project was begun in April 2001 in order to establish a long-term population project to assess the primate communities and their ecology. The environmental education activities began shortly after support was gained from international



Figure 1. The Azuero landscape, showing cattle pastures bordered with living fences, one of the remaining vegetation types where it is still possible to find native primates.

institutions (Ruiz-Bernard *et al.* 2010). Each year, different strategies were adopted for these surveys, including the following.

Training students from the Biology School of the University of Panama

This training is provided free, and includes teaching different techniques for surveying primates, specifically for *Cebus*, *Alouatta*, and *Ateles*. The curriculum includes methods for the collection of field data on primate group structure, mapping and compass use, and the calculation of distances and heights for habitat descriptions. After five hours of theory, the project offers the students the opportunity to practice these methods in the field by acting as volunteers in five to ten day surveys (see “youtube” site “atelesaz”). Field training includes the use of GPS, recording equipment to study vocalizations, fecal sampling and observational techniques. The goal of this initiative is to develop the interest of senior biology students in primate research for their bachelors theses, while also supporting our project with their fieldwork.

Educational activities in the primary and secondary schools in Azuero

Educational talks have been developed to offer basic information to the children in the regional schools in the northeastern areas of Azuero. Here the remaining Azuero howler and capuchins monkey populations live close to the villages and are at greater risk. In these presentations, we talk about primate biodiversity, the Azuero primates, and the characteristics that humans share with them as part of the same mammalian order. We also talk about how similar we are in our anatomy, the social and family structure, and daily necessities (for example, food, refuge, and use of plants). The talks also touch on basic information about primate food, their importance in the ecosystem as seed dispersers and pollinators, and how they contribute to the survival of wild animals (for example, deer, paca, agoutis, and peccaries). We normally

conclude with ten minutes of questions or drawing activities, where we measure the knowledge of the students after each talk and answer any remaining questions that they might have. We give each school a poster with information on the three subspecies of Azuero primates, indicating how they can help to minimize the risk of their extinction. The initiative also gives away t-shirts that have images of the most endangered non-human primates in Azuero and a message on the back of how they can help the primates, which are a part of their heritage and responsibility.

Educational activities around natural reserves and villages in Azuero

The Azuero Peninsula has three main natural zones under government protection: El Montuoso Forest Reserve in the north of the Herrera province, Cerro Hoya National Park at the southwestern part of the peninsula (eastern part of Veraguas province), and La Tronosa Forest Reserve at the southern end of Los Santos province. We have contacted members of the ANAM and the forest rangers of these protected areas every year before and during our field work. Our visits with the forest rangers include an open discussion where we exchange information about our results, what we do in Azuero, their experiences seeing monkeys and conservation issues in the area. We give them new information, and also t-shirts with conservation messages. Our goal is focused on obtaining information on the ranger’s needs and how they work to improve the protection of the natural areas under their charge. That information is used to evaluate the effectiveness of the parks and reserves (see Rodríguez-Luna *et al.* 1996), and make recommendations to the ANAM’s directors for the future management of the parks and reserves of the zones. As we visit the remote areas of the peninsula, we inform the communities about what we are doing and also obtain information from them that can facilitate our encounters with primates. We are interested in knowing their point of view about having primates close to their houses, as well as the past history of the occurrence of the primates and local extinctions of isolated populations. With this information and contacts, our goal is to involve local people and have them participate in our activities. We want to obtain information about the history of the fauna, native plant knowledge, and the presence and problems related to the primates.

Road signs, ‘youtube’ videos, and newspaper articles

The information obtained is passed to the general public beyond the scientific community by putting signs along the roads, posting videos with conservation messages on the internet, and publishing articles in national newspapers. After realizing that groups of Azuero primates can be found in gallery forests along some secondary roads and close to bridges, we have posted some signs beside the roads and bridges to warn people not to feed the monkeys. Feeding monkeys is practiced in some areas by people thinking that feeding them is a good and humanitarian thing for them to do. However, other local people feed them poisoned foods, arguing that

they damage their crops and plantations, and believing it is a way to protect their cattle from the larvae of the botfly (*Cuterebra baeri*) that they see on the howler monkeys.

Some locals used to put out bananas and leftovers from restaurants to feed the monkeys on the bridges, causing digestive problems for the primates. There have been at least three food-related instances that led to the death of nine or ten monkeys in the past three years. Azuero howlers in those areas have also been killed by electrical installments, and cars when they walk on the bridges looking for food placed there by people. Our yearly visits and the volunteers visiting the community are now helping to protect the monkeys in these vulnerable areas of Azuero.

Most of the younger generation in northeastern Azuero now use the internet. Secondary school students who are very familiar with the internet are now less familiar with the primates living in their own town. FCPP has posted videos with topics related to environmental education on <youtube.com>. This is done every year to keep the students and the general public informed about the fauna and flora in the region. By viewing these videos the students can also learn about our activities around the peninsula. Newspaper articles providing general information related to this project and the Azuero primates are also important. This information is accessible to the local community where we work and encourages the conservation of the fauna and flora in the region (Fig. 2).



Figure 2. Environmental education activities developed for the Fundación Pro-Conservación de los Primates Panameños-FCPP, Azuero Peninsula, Panama. These include the following: road signs, informative talks to primary and secondary schools, creating educational tools such as masks and paintings related to the primates and their habitat, informative posters, and educational talks with the forest rangers of the reserves.

Results

We have evidence of positive effects of the program on the local people who await our annual visits and collect anecdotal information for us. Based on our recent questionnaires, the people are showing an increase of 30% in basic knowledge related to the monkeys, and owners of agricultural lands are avoiding unnecessary tree-felling. The knowledge of the land owners has further improvement of (71% for 2011) compared with the first pre-survey (42% for 2001), but more people have to be reached. As this program has been supported by local biologists, their interaction with the local community has created an excellent way to teach conservation practices. In addition, the children respect them as leaders of their communities (Fig. 3).

Training of students from the Biology School of the University of Panama

Ten undergraduate students have participated in the program; three represented Panama in the XI Mesoamerican Congress of Biology and Conservation at Oaxtepec, Morelos, Mexico, and the First Scientific Congress of the Azuero University Regional Center (CRUA) in Azuero, Panama in 2007. Other students have assisted primatologists visiting Panama and some others have been part of our environmental education team, giving talks in secondary schools around the Azuero Peninsula. FCPP organized an international primate symposium in 2012, in the XVI Mesoamerican Society for Biology and Conservation Congress held in Panama City.

Educational activities around natural reserves and villages in Azuero

The educational talks given during our surveys took place mostly at the following schools and colleges of Herrera province: Rafael Quintero Villarreal from Ocu El Calabazal School, School Cristina R. de Pinzón of La Polonia, School Aleida Agustina from Llano Grande; Los Santos province: La Miel of Las Tablas, and the Agronomy School of Tonosi. Educational guides, t-shirts, posters, stickers and bookmarks have been distributed to the Llano de Piedras School, El Montuoso School, La Corocita School, El Cacao of Tonosi School, Canajaguas School, Valle Rico School, besides other public centers around Azuero. FCPP have addressed around 300 students per year. We have visited the El Montuoso Forest Reserve (seven rangers) and Arenas ANAM station (four rangers) in Veraguas, Cerro Hoya. Our evaluation of the natural reserves is presented in Tables 1 and 2.

Road signs, youtube videos, and newspaper publications

We have put up signs that provide basic information related to the primates living in the area, along with our contact information and messages that inform the public not to feed the monkeys, and basic biological information in English and Spanish. Signs were placed along the ríos Oria and La Palma (Los Santos). Another two were placed near the Río Cañas in Tonosi (Los Santos), and two along the Río Pavo in Veraguas province. Two road signs alerting people to reduce speed were placed on the Tonosi road in Los Santos Province. Since 2001 we have published ten articles on the critical

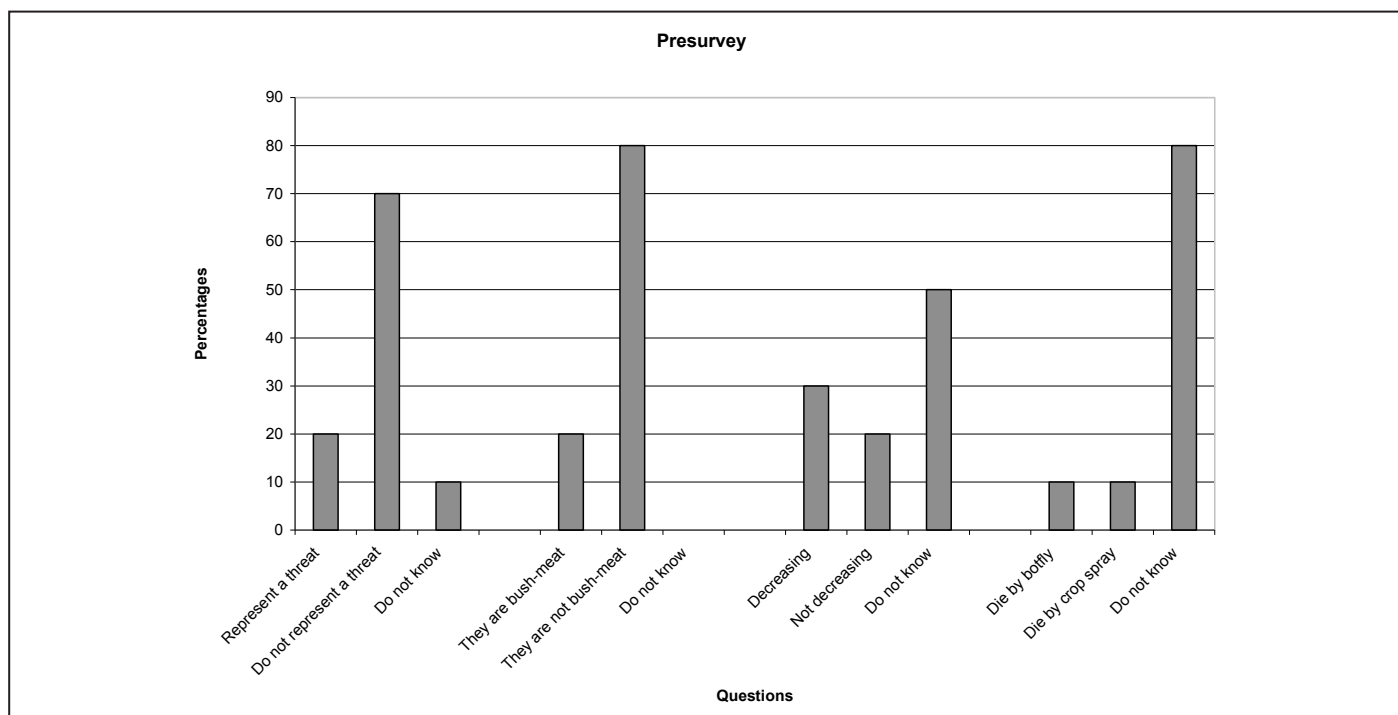


Figure 3. Pre-survey to evaluate basic knowledge about the primates in Azuero, n = 150, April–May 2001 (20 questions). The preliminary evaluation detected that there was rather little interest in hunting primates for food, although interviewees said that monkeys are shot sometimes as an easy target in hunting practice. They had little notion of the ecological role of primates in the forest. The local people considered the white-faced capuchin (*Cebus imitator*) to be a serious threat to their crops, but otherwise showed considerable interest in learning more about the conservation of the peninsula's primates.

situation of the monkeys in the area and one regarding the vocalization behavior of the Azuero howlers.

Future plans include engaging the Panama Ministry of Education (MEDUCA) as a collaborator in the creation and distribution of an educational guide for school teachers. Other governmental institutions such as the Gorgas Commemorative Institute (ICGES) and the Health Ministry of Panama (MINSAs) have contacted FCPP to obtain assessments and information about our primate monitoring program around Panama. The document “Guía Didáctica por la Conservación de los Primates de Azuero” prepared by FCPP includes topics related to the primates and to the yearly study plan for those communities. Measures of the effectiveness of these conservation activities include the continual monitoring and evaluation of primate populations and vegetation in old and new forest

patches. To date, all the primate groups that have been identified and censused since 2001 have survived, and their habitats have been almost untouched. This could be considered an important achievement of our project. Future projects include the measurement and comparison of forest expansion and regeneration. This will be done using advanced techniques of GIS. We will also study the correlation between fragment regeneration and the presence of primates and other mammals.

Discussion

These activities were all begun to study and conserve two endemic and Critically Endangered primates from southwestern Panama (Azuero Peninsula); the Azuero howler and spider monkeys. However, direct access to the community

Table 1. Achievements of FCPP’s environmental educational activities from 2001 to 2013.

Past situation	Activities developed (2001–2013)	Improvement	Actual situation/needs
Lack of basic information in the local communities	<ol style="list-style-type: none"> 1. Environmental education talks to schools, farmers and local people in general, University of Panama 2. Distribution of: stickers, posters and bookmarks with information 3. Educational Guide 4. Local publication in newspapers 5. Basic webpage 6. Road signs 7. Cultural activities 8. Local TV participation 	<ol style="list-style-type: none"> 1. Stopped feeding monkeys with bananas. 2. Wrong perception of abundance due to overcrowded population 3. Monkeys are not good for pet campaign 4. Local observers 5. Environmental Guide printed first set of 100 6. At least one article related with primates is published yearly in a local newspaper 7. Basic webpage 	<ul style="list-style-type: none"> • Lack of better webpage • Lack of enough environmental material • Budget to print our guides • More video or visual material • Lack of documentary informing about our projects • Improve quality of road signs
Lack of scientific information	<ol style="list-style-type: none"> 1. Population surveys 2. Distribution map 3. Vocalization studies 4. Behavior studies 5. We expanded our projects to Coiba Island, Darien, Boquete, Burica Peninsula, Panama Canal Zone and San Blas Mountain Chain 	<ol style="list-style-type: none"> 1. Informed about total population, group’s structure, birth and mortality rate per area 2. Accuracy in right distribution 3. Actual biological information 4. First evaluation of vocalization structure for <i>A. coibensis trabeata</i> 	<ul style="list-style-type: none"> • Ecological projects related to study the seed dispersers and pollinators in Azuero Peninsula • Budget to monitor actual groups found • Expand our surveys
Lack of support and interest from national and international organizations in Panama	<ol style="list-style-type: none"> 1. Communicate with new people 2. Train ourselves and look for international training related with primates 	<ol style="list-style-type: none"> 1. University of Panama has been interested in cooperating with our organization 2. We have produced: 1 master thesis, 5 newspaper articles, 7 scientific articles, 1 educational guide, 1 book chapter, 7 contributions at an international level 	<ul style="list-style-type: none"> • Standardize methods. • PhD and Masters on Primatology or Anthropology from Panama are in need
Lack of funds	<ol style="list-style-type: none"> 1. Generate scientific proposals 2. Develop activities to obtain support 3. Government opportunities 	<ol style="list-style-type: none"> 1. We have generated more than 100 proposals, received support from more than 10 organizations 2. We did bookmarks representing each species of Panamanian primates for sell and distribute for free in local villages 	<ul style="list-style-type: none"> • Seeds funds available
Lack of biology students interested in primatology	<ol style="list-style-type: none"> 1. Talks to the University of Panama 2. Visits to branches of the Universities. 3. Contacting Biology professors 	<ol style="list-style-type: none"> 1. 10 students trained in surveying primates. 2. Participation in two regional congresses, and three international congresses 3. Organized the first Primatological symposium in Panama, the III for Mesoamerica 	<ul style="list-style-type: none"> • Lack of funds to support our trained students and do not lose their valuable skills
Lack of a conservation plan	<ol style="list-style-type: none"> 1. Elaborate a conservation plan through an environmental education and a long-term survey 	<ol style="list-style-type: none"> 1. We created our own plan for conservation following an action plan suggested by Rodríguez-Luna <i>et al.</i>, (1996) 	<ul style="list-style-type: none"> • Achieved
Lack of a national entity dedicated to study and conserve the Panamanian primates	<ol style="list-style-type: none"> 1. Legal process to create a NGO. 2. Expand studies of non-human primates to the rest of Panama 	<ol style="list-style-type: none"> 1. We created the For-Conservation Foundation of Panamanian Primates (FCPP). 	<ul style="list-style-type: none"> • Achieved

and interaction with the local people have called our attention to other species that can be positively impacted by our work, helping in the end, the entire native flora and fauna (Ruiz-Bernard *et al.* 2010). The capuchin *Cebus imitator* that is vulnerable according to Cuarón *et al.* (2013) is one of the species that could be considered as Endangered at the regional level, if we take into account the significance of this monkey for the local people. It is a crop-raider, and when there was no action from the government to mitigate the problem, farmers decided to hunt them to extermination. Our research and direct observations in the area have shown us that the only way to save this species is through our efforts and activities. The participation of the local biologists and support from national and international institutions is crucial to keep up this long-term effort.

Based on the “Action Plan for the Mesoamerican Primates”, prepared by Rodríguez-Luna *et al.* (1996), the Primate Specialist Group has classified the two most important steps toward the conservation of these primates. They are the Study Priorities (E) and the Conservation Actions (AC). Taking this document as a guide, the FCPP has achieved the following: E1 – Identify and monitor populations in protected areas; E2 – Determine densities and population growth; E3 – Determine actual distribution; E8 – Habitat preferred; E9 – Long-term evaluations and viability of populations. However we still need to complete the following steps, which are: E4 – Taxonomic studies; E5 – Measurement of hunting impact; E6 – Habitat transformation effects; E7 – Habitat fragmentation effects (already underway) (Tables 1 and 2).

For Conservation Actions, we have accomplished the following: AC1 – Encourage and promote more primatologists to study the primates and continue long-term projects; AC6 – Advise governmental authorities in the management of the primates and their habitat; AC7 – develop an environmental education program that promotes the collaboration between local people and the primate conservation plan. Regarding AC4 (develop more protected areas), what we have seen while working in the communities is that people reject the idea of strictly protected areas, not because they do not care about the species but because they depend on them for their subsistence needs. This was discussed by Cowlishaw and Dunbar (2000), who mentioned the implications of restricting the use of land by local people without incentives. We do not recommend the idea of creating more natural reserves in zones that are already being heavily exploited such as the Azuero Peninsula; it is not a priority for Azuero. The people of Azuero have the capacity to conserve their natural resources and fauna. This can be carried out through a program of intense environmental education. Our yearly monitoring of primates and our educational efforts in the area will encourage the cooperation of the locals and help us to stay vigilant to any changes. Our idea moreover, is to improve the use of living fences, which is already a cultural practice, and promote the connectivity of patches of forest with gallery forest and reserves already established (recommended also by Oates 1996).

The three most important natural reserves in Azuero Peninsula were evaluated based on the criteria of Mackinnon *et al.* (1986), proposed by Rodríguez-Luna *et al.* (1996). The information obtained reflected the commitment and effectiveness of the administration in El Montuoso Forest Reserve; however, spider monkeys are no longer found there. The Azuero spider monkey was hunted out about 20 years ago (Méndez-Carvajal 2011).

These point to weaknesses in the laws and the extent of population pressure on the protected area (see Pimbert and Pretty 1995). The effectiveness of La Tronosa Forest Reserve was lowest in the evaluation due to its greater accessibility to locals and also by people from the city, but this reserve still has Azuero howler and spider monkeys. Cerro Hoya Natural Park is a remote location and is difficult to reach, and is the best and safest place for Azuero howlers and spiders monkeys to survive (Table 2).

Table 2. FCPP’s evaluation of the management of three natural protected areas in Azuero Peninsula, Panama, according to the guidelines suggested by Rodríguez-Luna *et al.* (1996). The three reserves fall in the Medium Point Level. EMFR: El Montuoso Forest Reserve; CHNP: Cerro Hoya National Park, LTFR: La Tronosa Forest Reserve.

Category	EMFR	CHNP	LTFR
Legislation	4	4	4
Mammal inventory	1	1	1
Bird inventory	1	1	1
Reptile/amphibian inventory	1	1	1
Fish inventory	1	1	1
Plant inventory	1	1	1
Vegetation map	1	1	1
Invertebrate inventory	1	1	1
Geographic maps	1	1	1
Soil map	1	1	1
Climatic data	1	1	1
Hydrology data	1	1	1
Topographic maps	1	1	1
Aerial photos	0	0	0
Bibliography	1	1	1
Studies of fauna/flora	0	0	0
Population studies	0	0	0
Rel. wildlife/habitats	0	0	0
Predator/prey studies	0	0	0
Carrying capacity studies	0	0	0
Parasite information	0	0	0
Ecological succession	0	0	0
Life history information	0	0	0
Management plans	3	2	2
Limits	2	2	2
Natural resources protection	1	1	1
Research level	3	3	3
Formal education	4	3	3
Educational offers	3.5	1.5	0
Political support	4	4	4
Local participation	2	4	2
Benefits for locals	3	2	1
Budget	2	2	2
Personnel/training	2	2	2
External support	2	2	2
Score	48.5	42.5	41

Conclusion

The conservation activities for these species have been developed with the strong support of volunteers from the local communities, the University of Panama, and the support of national and international organizations. The objective of this article is to encourage native biologists or primatologists from developing countries to promote similar plans in order to protect their threatened species and obtain accurate data for their conservation. Efforts for conservation do not have to start with large sums of money. Instead, they should start with real commitment allied with voluntary efforts to work for the conservation of these species and their ecosystems. Money, however, is crucial for long term presence of these activities in target areas. One of the important steps in this conservation project has been the education of the communities and the understanding of their needs. We have also tried to take into account recommendations of conservation practices, which state that: “involvement of the local people into the conservation activities will promote good opportunities and actions for and from them, making the locals feel like a real part of the solution” (Horwich 1996; Rabinowitz 2003; Curtis and Valdez 2009). Our activities have caught the attention of the Panamanian authorities, international scientific institutions and NGO’s, stimulating possibilities for new projects related to the conservation of the Azuero primates; the Azuero Earth Project is an example; see Metzler (2011).

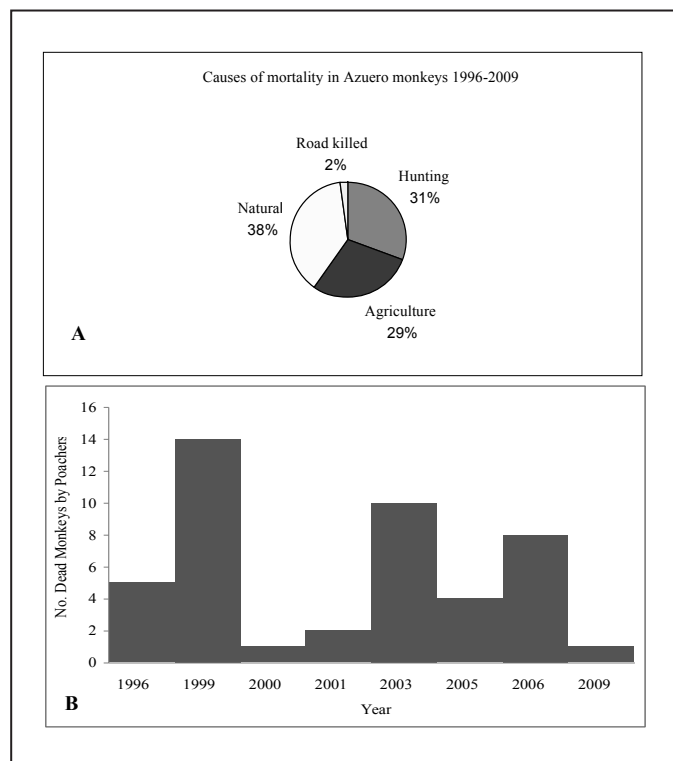


Figure 4. A) Percentage of the more important threats of Azuero primates expressed in a pie graph, for the period of 1996 to 2009. Natural factors have been influencing primate populations and other fauna probably due to the climate changes by deforestation. B) Incidence of poaching on the howler population per year.

However, our experience working with the Azuero communities has also suggested that our substantial efforts to conserve these species could come to naught without collaboration and alignment from new research initiatives and other organizations, such that are equally committed to our cause. Diverse initiatives with differing goals and methodologies could well confuse the local communities and undo the goodwill and understanding that we have fostered. Even creation of environmental laws, 25 critically endangered species lists, or national parks or reserves will not help in conservation if we do not deal with the people and inform them directly, so local people need to be trained and supported if we really want to save a species.

Panamanian newspapers have been very supportive, with local papers reporting hunting events that take place in Azuero. A series of articles written by the newspapers *La Prensa de Panamá*, *Día a Día*, *La Crítica Libre de Panamá*, and *El Panamá América* (Día “D” Supplement), has reported on hunting activity, the deaths of monkeys and their causes, helping us focus our research and conservation plans (Fig. 4). We hope that this long-term environmental education project will help with ideas for the conservation of other Neotropical primates.

Recommendations

Azuero howler and spider monkeys are endemic subspecies not only at the country level but also at the regional level, which placed them as “Maximum Priority” for conservation measures in the Mesoamerican Primate Action Plan (Rodríguez-Luna *et al.* 1996). They are ranked as Critically Endangered on the IUCN Red List (Cuarón *et al.* 2013). The only official report on their conservation status has estimated only 145 Azuero spider monkeys still surviving in the wild (Méndez-Carvajal 2011). About 3,000 Azuero howler monkeys remain in the wild (Méndez-Carvajal 2011). Considering that these calculations include infants and juveniles, the reproductive population (adults) for these primates is considerably less. With these low numbers, we recommend that any invasive method employed to study these primates, including the use of radio collars, be avoided. For our organization, it is great to share our achievements and our future goals not only for Azuero primates, but for all the non-human primates living in Panama (Tables 1 and 2).

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Rapid Survey and Assessment of the Northern Sportive Lemur, *Lepilemur septentrionalis*, in Northern Madagascar

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Abstract: The northern sportive lemur, *Lepilemur septentrionalis*, faces imminent danger of extinction, more so than any other lemur in Madagascar. The population estimates for this sportive lemur remain unknown because of habitat loss and ongoing human encroachment, but they are unlikely to number more than a few hundred individuals. We present the results of extensive surveys conducted in 2010, 2011 and 2012 of known habitat, the Sahafary and Analalava classified forests, and confirm the species designation of the sportive lemur observed in Montagne des Français in 2007. Six *L. septentrionalis* individuals were examined in Sahafary in 2011, along with eight individuals during the 2010 and 2011 expeditions to Montagne des Français. A July 2012 survey in extended forest fragments of Montagne des Français identified another 10 individuals. Surveys of the Analalava forest in 2011 detected no northern sportive lemurs, despite documenting their presence in 2005. One individual was recorded in the July 2012 survey. Although the morphological data of the sportive lemurs at Montagne des Français was comparable to that of Sahafary, the sportive lemur at Montagne des Français was subsequently verified as *L. septentrionalis* with mitochondrial DNA D-loop sequence data analyses. The confirmation of the northern sportive lemur at Montagne des Français is significant since it establishes additional habitat for this species. However, sustained human encroachment from Antsirana continues to finance the production of charcoal and collection of sand; activities that are threatening this population. Habitat loss and hunting continue to be the principal threats to the long-term survival of the northern sportive lemur. With only 19 known individuals, we urge immediate conservation action for this Critically Endangered species.

Keywords: northern sportive lemur, *Lepilemur septentrionalis*, Sahafary, Montagne des Français

Introduction

The northern sportive lemur, *Lepilemur septentrionalis*, a small sportive lemur, inhabits dry forests of northern Madagascar (Rumpler and Albignac 1975; Junger and Rumpler 1976; Rumpler *et al.* 2001). The distribution of this nocturnal lemur was initially believed to extend from the Montagne d'Ambre region in the north to the Mahavavy River in the west (Hawkins *et al.* 1990; Ratsirason and Rumpler 1998; Rumpler *et al.* 2001). Rumpler and Albignac (1975) originally described *L. septentrionalis* with four subspecies (*L. s. andrafiamenensis*, *L. s. ankaranensis*, *L. s. sahafarensis*, and *L. s. septentrionalis*). Based on the parameter of geographic separation, Groves (2001) reduced the four subspecies to two, *L. s. septentrionalis* (*L. s. sahafarensis* as a junior

synonym) and *L. s. ankaranensis* (*L. s. andrafiamenensis* as a junior synonym). Rumpler (2004) and Ravaoarimanana *et al.* (2004) subsequently elevated these two subspecies to species based on cytogenetic and molecular data. The taxonomic status of *L. septentrionalis* has continued to be supported by more detailed cytogenetic, morphological and molecular data (Rumpler *et al.* 2001; Ravaoarimanana *et al.* 2004; Andriaholinirina *et al.* 2006; Louis *et al.* 2006; Lei *et al.* 2008; Ramaromilanto *et al.* 2009). Due to the taxonomic revision confirming *L. septentrionalis* and *L. ankaranensis* as distinct species, the perceived range of the northern sportive lemur was drastically reduced; limited to a few degraded patches of dry forest in the Sahafary region just south of Antsirana (Fig. 1). Considered the southern geographic extent for this species, the Irodo and Bobakindro rivers, combined with

the limestone plateau and tsingy formation of Ankarana and Analamerana, could be acting as a significant barrier to dispersal (Louis *et al.* 2008).

Recognized as Critically Endangered on the IUCN Red List of Threatened Species and ranked as one of the world's 25 most endangered primates (Mittermeier *et al.* 2008; Mittermeier *et al.* 2009; IUCN 2012), the northern sportive lemur is in imminent danger of extinction, more so than any other lemur (Mittermeier *et al.* 2010). Expeditions carried out by the Madagascar Biodiversity Partnership and Omaha's Henry Doorly Zoo and Aquarium in 2005, recorded the northern sportive lemur in the classified forests of Analalava and Sahafary; its identity subsequently verified through molecular genetics analysis (Louis *et al.* 2006). Analalava classified forest persists as a very small scrubby and degraded habitat of about 80 ha (Figs. 1–3). Sahafary classified forest is embedded in a *Eucalyptus* plantation that is routinely used for charcoal production, and there is less suitable habitat even than in Analalava. An independent survey led by R. A. Mittermeier in 2005 visited the forest fragments of Andrahona and Tsaratanana near Ankarongana. They also went to Madirobe, mentioned by Y. Rumpler (*pers. comm.*) as a possible site for *L. septentrionalis*, but no forest was found there and

the local people seemed not to know anything about the animal. Andrahona is a sacred mountain, located 36 km south of Diego-Suarez. The forest there was in reasonably good condition, especially around the peak, but there were many trails and significant numbers of saplings were being taken out for construction. Local people said that sportive lemurs occurred there, but none were found. Further surveys there in September 2012, led by E. E. Louis, recorded only *Microcebus tavaratra* and *Eulemur coronatus*. The forest at Tsaratanana was a tiny tract of degraded, heavily used forest in the middle of a large *Eucalyptus* plantation, 5.4 km north-west of the town of Ankarongana. A guide from Ankarongana was familiar with the animal and a single *L. septentrionalis* was found (Fig. 4). It had a small "V"-shaped cut in its ear, so was perhaps an animal that had been studied by Y. Rumpler's team. The chances of protecting the forest there were considered very remote.

Due to the political instability created by the 2009 coup, little if any security is being provided for any protected or non-protected area in Madagascar. According to a survey in 2007, approximately one hundred northern sportive lemurs were estimated in the Andrahona, Ankarakataova and Sahafary regions, none of which are protected (Mittermeier *et*

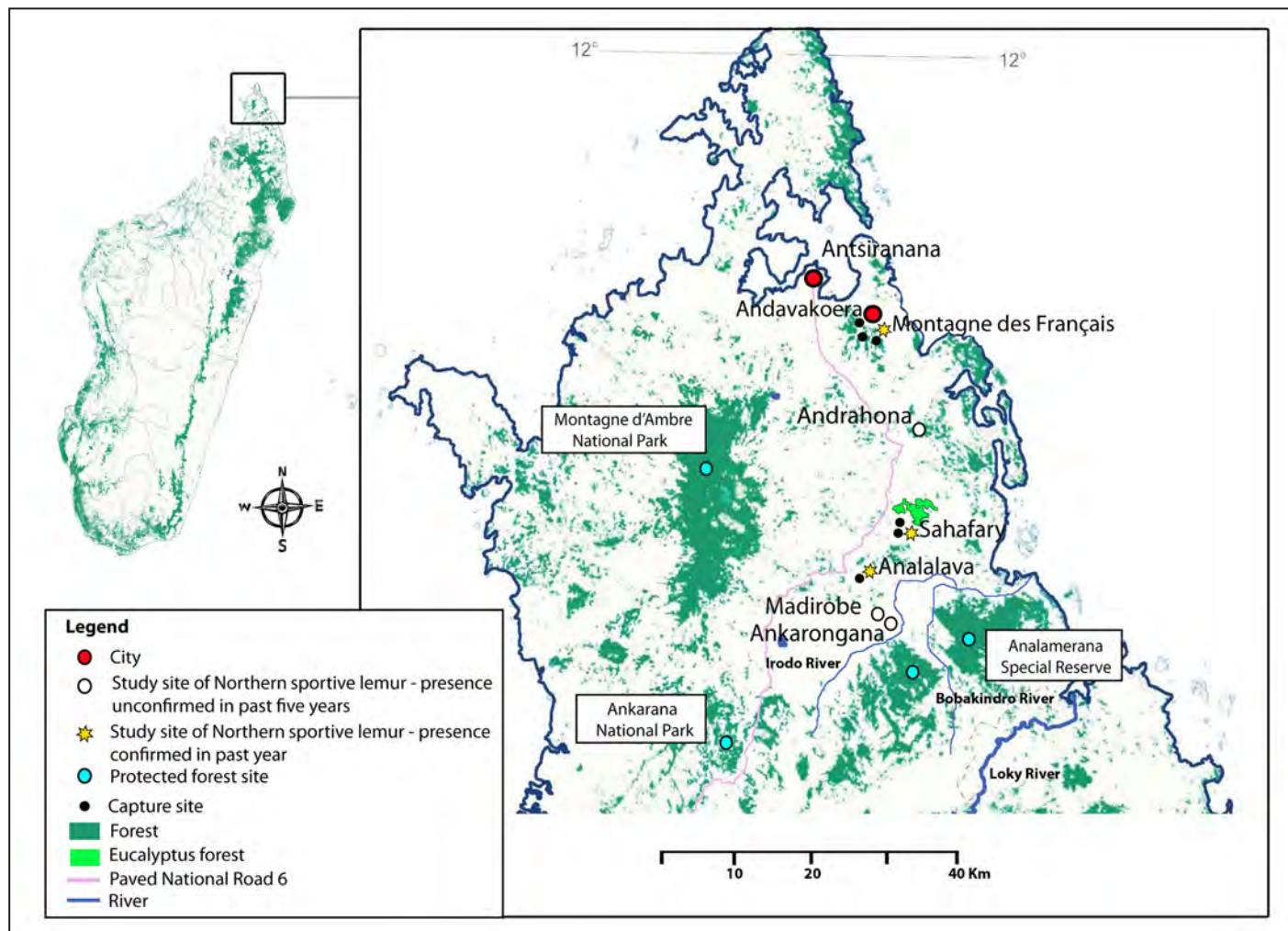


Figure 1. Distribution range and capture localities of the northern sportive lemur, *Lepilemur septentrionalis*.

al. 2008; Mittermeier *et al.* 2010). During a comprehensive survey of amphibians and reptiles at Montagne des Français, D’Cruze *et al.* (2007) photographed what they presumed to be *L. septentrionalis*. However, without a reference to indicate the relative dimensions in the photograph, direct confirmation could not be made without a clear-cut comparison to the nominal species in the region, *L. septentrionalis* and *L. ankaranensis*. Here, we present the results of three extensive surveys conducted in 2010, 2011 and 2012 that revisited the Sahafary and Analalava forests, and determined the species status and range of the unspecified sportive lemur noted at Montagne des Français (Fig. 3). We also suggest conservation measures for the remaining populations of the Critically Endangered northern sportive lemur.



Figure 2. Landscape of the Analalava region showing the remaining forest in the valley (center of photograph). Loss of habitat is evident throughout the horizon. Introduced *Eucalyptus* forests can be seen on the ridge above the Analalava forest (left side of photograph), along with the effects of erosion from long-term slash-and-burn practices creating a massive ridge. Photograph taken by Jean Freddy Ranainarisoa.



Figure 3. Landscape of the Sahafary region showing the tremendous loss of habitat and crater-like landscape created by effects of erosion from long-term slash-and-burn practices. Photograph taken by Jean Freddy Ranainarisoa.

Methods

Study area

The northern sportive lemur is found in dry deciduous forest fragments, restricted to the extreme northern region of Madagascar (Fig. 1). Research was conducted at Montagne des Français (12°20' 02.7"S, 49°21' 21.9"E) in 2010, 2011 and 2012, and in Sahafary (12°36'20.7"S, 49°26'32.8"E) and Analalava (12°39'25.6"S, 49°24'36.5"E) in 2011 and 2012 (Figs.1–3; Table 1). This forest belt is found south-east of the seaport city of Antsiranana, in the district of Diego I, near the commune of Ramena, extending south into the district of Diego II, near the communes of Sadjoavato and Ankarongana.

Sample collection

All lemurs investigated in this study were wild-caught, free-ranging individuals immobilized with a CO₂ projection rifle or blowgun with 10 mg/kg of Telazol® (Fort Dodge Animal Health; Overland Park, Kansas; Table 1). Four 2.0 mm biopsies and 1.0 cc per kilogram of whole blood were collected from each sedated animal and immediately stored in room temperature storage buffer (Seutin *et al.* 1991). Fecal samples were collected and stored immediately in 10 ml of RNALater® (Life Technologies, Grand Island, New York). A HomeAgain® microchip (Schering-Plough Veterinary Corp.; Kenilworth, New Jersey) was placed subcutaneously between the scapulae of each lemur. This procedure was used to field-catalog each animal with a unique recognition code in order to re-identify all captured individuals during any future immobilizations. Morphometric measurements were also taken following Louis *et al.* (2006). For presentation purposes, we summarize all measurements by population in this publication following the guidelines of Smith and Jungers (1997). Genomic DNA was extracted from samples using a phenol-chloroform/isoamyl extraction (Sambrook *et al.* 1989).

Data generation

From these samples, the displacement loop or control region of mitochondrial DNA (530-553 base pairs) was amplified (D-loop; Baker *et al.* 1993; Wyner *et al.* 1999) under the conditions described in other studies (Louis *et al.* 2006; Lei *et al.* 2008; Ramaromilanto *et al.* 2009). The samples were electrophoresed on a 1.2% agarose gel to verify the PCR product and purified with Exonuclease I and shrimp alkaline phosphatase (EXOSAP; Silva *et al.* 2001).

The purified products were cycle sequenced using a BigDye® terminator sequencing kit (Life Technologies, Carlsbad, CA). The sequences were analyzed by capillary electrophoresis with an Applied Biosystems 3130xl genetic analyzer. The PCR and sequencing primer suite from Louis *et al.* (2006) were used to generate the D-loop fragment sequences. The sequences were aligned to generate a consensus contig using Sequencher® 4.10 (Gene Codes Corporation; Ann Arbor, MI). All sequences have been deposited in GenBank with accession numbers JQ771832-JQ771834. We used accessioned GenBank D-loop sequences for sportive lemur

species to establish reference baselines (Louis *et al.* 2006; Lei *et al.* 2008; Ramaromilanto *et al.* 2009).

Phylogenetic analysis

Maximum likelihood (ML) analyses for the D-loop sequence data were performed with PhyML 3.0 software under HKY+I+G model selected by Modeltest 3.7, along with bootstrap percentages computed after 1000 replicates (Guindon and Gascuel 2003; Guindon *et al.* 2005; Posada and Crandall 1998). The best-scoring ML-trees were searched and saved in PAUP* 4.0b10 (Swofford 2001). Bayesian inference analysis was conducted using MrBayes 3.0b4 (Huelsenbeck and Ronquist 2001; Ronquist and Huelsenbeck 2003). A Markov Chain Monte Carlo (MCMC) run with four simultaneous chains and 1,000,000 generations was performed with HKY+I+G model selected for the ML inferences by using MrModeltest 2.2 (Nylander 2004). Every hundredth generation, the tree with the best likelihood score was saved, resulting in 10,000 trees. These were condensed in a majority rule consensus tree using PAUP* 4.0b10 (Swofford 2001) and clade posterior probabilities (PP) were computed. MEGA 4.0 (Tamura *et al.* 2007) was used to calculate uncorrected pairwise distances ('p') and absolute differences.

Results

Six *Lepilemur septentrionalis* were captured in Sahafary in 2011 (Table 1). Ten individuals were sampled in Montagne des Français: five in 2010 and five in 2011, along with two recaptured individuals identified by previously implanted microchips. Unfortunately, no northern sportive lemur was detected during the 2011 expedition into the Analalava forest despite its noted presence in 2005, but one individual was documented during the July 2012 expedition (Table 1). Furthermore, only one solitary female crowned lemur, *Eulemur coronatus*, was

seen during the week-long field survey at this site in 2011, but none in 2012. Morphological data are presented in Table 2. The northern sportive lemur individuals from Montagne des Français are slightly larger in average size than the individuals from Sahafary (650 g and 630 g, respectively).

The uncorrected pairwise differences among all recognized *Lepilemur* species for D-loop sequences ranged from



Figure 4. Sahafary sportive lemur (*Lepilemur septentrionalis*), Tsaratanana Forest, north-west of Ankarongana village. Photo by R. A. Mittermeier.

Table 1. List of northern sportive lemur, *Lepilemur septentrionalis*, examined during this study.

ID Number	Location	Sex	Microchip ID	Weight (kg)	Global Positioning System Coordinates		Survey Year
MDF10.1	Montagne des Français	Female	47311E676C	0.58	S12°20'07.4"	E:049°21'17.4"	2010
MDF10.2	Montagne des Français	Female	460B1D4B08	0.73	S12°20'04.3"	E:049°21'16.2"	2010, 2011
MDF10.3	Montagne des Français	Male	473110605D	0.63	S12°20'04.6"	E:049°21'19.0"	2010
MDF10.4	Montagne des Français	Female	46703A4210	0.58	S12°20'05.9"	E:049°21'18.2"	2010, 2011
MDF10.5	Montagne des Français	Female	460A1C2344	0.68	S12°20'05.3"	E:049°21'19.3"	2010
MDF11.1	Montagne des Français	Female	4732703611	0.73	S12°20'02.7"	E:049°21'21.9"	2011
MDF11.2	Montagne des Français	Female	46702C5F16	0.72	S12°20'04.2"	E:049°21'26.3"	2011
MDF11.3	Montagne des Français	Male	4730466E37	0.70	S12°20'05.0"	E:049°21'25.4"	2011
MDF11.4	Montagne des Français	Female	4731203336	0.65	S12°20'59.2"	E:049°21'34.5"	2011
MDF11.5	Montagne des Français	Female	460COC2B45	0.56	S12°20'03.5"	E:049°21'16.3"	2011
FARY11.1	Sahafary (Madirobe)	Male	467025447A	0.47	S12°36'20.7"	E:049°26'32.8"	2011
FARY11.2	Sahafary (Madirobe)	Female	476D115C41	0.70	S12°36'25.2"	E:049°26'34.2"	2011
FARY11.3	Sahafary (Madirobe)	Female	4773303F28	0.69	S12°36'35.7"	E:049°26'35.5"	2011
FARY11.4	Sahafary (Madirobe)	Male	47682A370D	0.67	S12°36'35.7"	E:049°26'35.5"	2011
FARY11.5	Sahafary (Madirobe)	Male	47732C6D2C	0.61	S12°36'34.2"	E:049°26'35.1"	2011
FARY11.6	Sahafary (Madirobe)	Female	4777511349	0.68	S12°36'32.4"	E:049°26'32.7"	2011

2.3% to 15.5% (Tables 3a and 3b). The uncorrected pairwise differences for D-loop sequences between *L. septentrionalis* and the other 25 nominal sportive lemur species ranged from 5.1% to 13.4%. The uncorrected pairwise differences representing interspecific variation between *L. septentrionalis* from Montagne des Français and Sahafary/Analalava were 1.4%. In the maximum likelihood and Bayesian phylogenetic tree reconstructions, similar tree topologies were obtained (Fig. 6). Three individual northern sportive lemurs from Montagne des Français, MDF10.2, MDF10.4 and MDF10.5, clustered closely with the one haplotype of *L. septentrionalis* from Sahafary/Analalava (Fig. 6).

Discussion

In 2005, the only known populations of the northern sportive lemur were reconfirmed as originally recorded by Rumpler and Albignac (1975; Louis *et al.* 2006). Both morphometric and molecular analyses confirm that the sportive lemur at Montagne des Français is indeed *L. septentrionalis*, extending the distribution from the Sahafary and Analalava regions (Fig. 6; Tables 1–3). The confirmation of the northern sportive lemur is significant since it establishes a new locality for this species. The expeditions in 2010 and 2011 verified the continuing existence of the northern sportive lemur in Sahafary classified forest, but not a single animal was detected during these expeditions in the Analalava forest where it formerly existed in 2005. During the July 2012 expedition, however, one individual was reported when Analalava was revisited. The northern sportive lemurs from Montagne des Français were slightly larger in average size than those from Sahafary (0.65 kg and 0.63 kg, respectively), but this difference could be attributed to the level of an individual's maturity (Tables 1–2). Another possible explanation for the slight size difference could be severe habitat degradation in Sahafary versus higher quality forest forage in Montagne des Français (Fig. 3).

In 2008, Service d'Appui à la Gestion de l'Environnement (SAGE; <www.MadagascarSAGE.org>) promoted the designation of Montagne des Français as a newly protected area, and supported the development of a Vondron'Olona Ifototra (VOI) in Andavakoera, the primary village of this mountain forest. A VOI is a local government organization at the village level that manages the region's resources. Montagne des Français is the most pristine forest available for the northern sportive lemur, providing natural quality habitat for this Critically Endangered species.

SAGE estimates that approximately 20,000 people are exploiting this forest even though Andavakoera has only 200 residents. Business interests from the nearby seaport of Antsiranana finance the exploitation of the remaining Montagne des Français habitat through the production of charcoal and collection of sand for city and port construction projects (D'Cruze *et al.* 2007). Habitat loss from uncontrolled long-term slash-and-burn practices, regrowth of non-endemic invasive plants, and the desertification effects of deforestation and erosion has resulted in a mosaic landscape of forest fragments, grasslands, and craters (Figs. 2–3). Human encroachment is compounded by debilitating poverty and exacerbated by the political instability of the past four years. Moreover, opportunistic hunting and illegal logging over the past seven years has impacted all lemur populations throughout the region. During the previous 2005 expedition in the Analalava forest, numerous northern sportive lemurs were noted, along with multiple family groups of crowned lemurs, *Eulemur coronatus*, and Sanford's lemurs, *E. sanfordi*. During the 2011 week-long survey, however, only one solitary female crowned lemur was observed and the majority of the forest was gone (Fig. 2). No other lemurs were documented in the 2012 survey except for one *L. septentrionalis*. Even in the Sahafary region, habitat loss has been severe due to the effects of erosion during this same period. The folivorous diet of the northern sportive lemur and this species' predilection for leaves complicates any attempts or plans to maintain it in captivity. There is no record of a sportive lemur held in any zoological park, as all known attempts to maintain them in captivity have failed on average within one week of capture. *In situ* conservation



Figure 5. Northern sportive lemur, *Lepilemur septentrionalis*, from Montagne des Français. Photograph taken by Edward E. Louis, Jr.

Table 2. Average morphometric data (cm) summarized from northern sportive lemur, *Lepilemur septentrionalis*, populations.

	HC	Body L.	Tail L.	F-H	F-UR	F-Hd	F-LD	F-Tb	H-F	H-T	H-Ft	H-LD	H-Tb
MDF	7.6±0.5	20.7±1.1	26.7±1.0	6.2±1.1	6.6±0.5	5.4±0.3	2.6±0.2	1.4±0.2	10.1±0.8	9.0±0.8	7.8±0.5	2.6±0.2	2.7±0.3
FARY	7.6±0.2	20.7±2.9	23.9±1.6	5.4±0.2	6.3±0.6	4.9±0.4	2.6±0.2	1.3±0.1	10.5±0.6	9.5±0.8	6.4±0.8	2.8±0.2	2.3±0.3

Note: F-H: Humerus, F-Hd: Hand, F-LD: longest digit (Forelimb), F-Tb: Thumb (forelimb), F-UR: Ulna/radius, HC: head crown, H-F: Femur, H-Ft: foot, H-LD: longest digit (Hindlimb), H-T: Tibia, H-Tb: Thumb (Hindlimb), L.: length, MDF: From Montagne des Français, RATY: From Sahafary (Madirobe)

Table 3a. Genetic distance matrix for D-Loop fragment sequence data for the 26 nominal sportive lemur species. The numbers represent the following *Lepilemur* species: 1. *L. ankaranaensis*, 2. *L. milanoii*, 3. *L. tymerlachsonorum*, 4. *L. dorsalis*, 5. *L. ahmansorum*, 6. *L. sahalazensis*, 7. *L. septentrionalis*, 8. Population from Montagne des Français, 9. *L. edwardsi*, 10. *L. grewcockorum*, 11. *L. otto*, 12. *L. microdon*, 13. *L. leucopus*, 14. *L. petteri*, 15. *L. randrianasoloi*, 16. *L. aeeclis*, 17. *L. ruficaudatus*, 18. *L. hubbardorum*, 19. *L. fleuretiae*, 20. *L. mustelinus*, 21. *L. betsileo*, 22. *L. jamesorum*, 23. *L. wrightae*, 24. *L. seali*, 25. *L. hollandorum*, 26. *L. scottorum*. Genetic distances based on absolute differences are displayed above the diagonal, and genetic distances based as a percentage are displayed below the diagonal.

	1	2	3	4	5	6	7	8	9	10	11	12	13
1		12.7	19.7	30.8	31.2	29.7	34.0	33.8	33.0	45.2	44.7	48.8	43.7
2	2.6±0.6		11.6	24.4	30.2	28.0	29.2	27.0	35.8	42.5	48.1	47.5	43.7
3	4.0±0.8	2.3±0.6		23.5	29.0	21.0	28.0	25.0	37.0	41.5	45.3	50.3	44.0
4	6.2±1.0	4.9±0.9	4.8±0.9		25.8	16.8	29.5	29.5	32.8	35.2	40.2	49.8	49.0
5	6.3±1.0	6.1±1.0	5.9±1.0	5.2±0.9		26.0	37.0	34.0	27.0	37.0	38.0	44.8	45.4
6	6.0±1.0	5.7±1.0	4.3±0.8	3.4±0.7	5.3±0.9		33.0	33.0	28.0	34.5	37.7	44.3	45.2
7	6.9±1.0	5.9±1.0	5.7±1.0	6.0±1.0	7.5±1.1	6.7±1.0		7.0	39.5	41.5	47.3	51.3	40.4
8	6.8±1.0	5.5±0.9	5.1±0.9	6.0±1.0	6.9±1.1	6.7±1.0	1.4±0.5		40.5	40.5	47.7	52.3	42.6
9	6.7±1.0	7.2±1.1	7.5±1.1	6.6±1.0	5.5±1.0	5.7±1.0	8.0±1.1	8.2±1.2		27.5	22.0	33.5	31.4
10	9.1±1.1	8.6±1.2	8.4±1.2	7.1±1.1	7.5±1.1	7.0±1.0	8.4±1.2	8.2±1.2	5.6±0.9		29.5	37.8	42.7
11	9.0±1.2	9.7±1.3	9.2±1.2	8.1±1.1	7.7±1.1	7.6±1.1	9.6±1.2	9.6±1.3	4.5±0.8	6.0±0.9		39.4	44.1
12	9.9±1.2	9.6±1.2	0.2±1.3	0.1±1.2	9.1±1.2	9.0±1.2	10.4±1.3	10.6±1.3	6.8±1.0	7.6±1.1	8.0±1.1		42.3
13	8.8±1.2	8.9±1.2	8.9±1.2	9.9±1.2	9.2±1.2	9.1±1.2	8.2±1.1	8.6±1.2	6.4±1.0	8.6±1.2	8.9±1.1	8.6±1.2	
14	9.1±1.2	9.2±1.2	9.1±1.2	8.8±1.2	10.0±1.2	8.0±1.1	7.6±1.1	8.2±1.2	6.6±1.0	8.6±1.2	9.0±1.1	9.0±1.2	2.9±0.6
15	9.0±1.2	9.4±1.3	9.7±1.3	10.2±1.3	9.2±1.2	10.2±1.3	8.6±1.2	8.5±1.2	8.5±1.1	9.0±1.2	9.9±1.2	10.1±1.2	5.9±1.0
16	10.5±1.2	10.7±1.3	10.4±1.3	10.4±1.3	9.5±1.2	10.1±1.2	9.7±1.2	10.3±1.3	8.2±1.0	8.9±1.2	9.8±1.2	9.8±1.2	8.4±1.1
17	10.9±1.3	11.2±1.3	11.0±1.3	10.8±1.3	11.3±1.3	9.9±1.3	9.9±1.2	10.0±1.2	10.1±1.3	10.4±1.3	10.0±1.2	11.0±1.3	9.3±1.2
18	10.1±1.2	10.3±1.3	10.3±1.3	9.9±1.2	10.4±1.3	10.8±1.3	9.3±1.2	9.1±1.2	9.5±1.2	9.5±1.2	9.8±1.2	11.3±1.3	8.3±1.2
19	12.6±1.4	12.4±1.4	13.2±1.4	13.2±1.4	11.4±1.3	12.1±1.4	13.2±1.4	12.3±1.4	12.4±1.4	12.9±1.4	12.9±1.4	13.1±1.4	13.9±1.5
20	14.0±1.4	13.3±1.4	13.3±1.4	13.4±1.4	13.3±1.4	12.7±1.4	13.3±1.4	13.6±1.4	13.1±1.3	14.1±1.4	13.9±1.3	12.7±1.3	13.3±1.4
21	12.1±1.3	11.6±1.3	11.7±1.3	12.4±1.4	12.2±1.4	11.2±1.4	13.2±1.4	12.6±1.4	12.2±1.4	13.1±1.4	12.9±1.4	11.8±1.3	13.6±1.5
22	13.2±1.4	12.8±1.3	13.2±1.4	13.3±1.4	12.5±1.3	12.3±1.4	13.0±1.4	12.4±1.3	13.1±1.4	14.0±1.4	13.3±1.3	13.1±1.4	14.4±1.4
23	13.2±1.4	12.7±1.4	11.6±1.3	11.9±1.3	11.2±1.3	11.8±1.3	12.3±1.3	11.8±1.3	11.7±1.3	11.1±1.3	11.5±1.3	12.2±1.4	13.6±1.4
24	12.8±1.4	12.4±1.4	13.2±1.5	13.7±1.4	13.0±1.4	13.4±1.4	13.4±1.4	13.2±1.4	13.1±1.4	12.4±1.4	13.9±1.4	12.3±1.3	15.3±1.5
25	13.1±1.3	13.5±1.4	12.3±1.4	13.9±1.4	14.2±1.4	12.6±1.3	12.8±1.4	12.4±1.4	13.5±1.4	14.3±1.4	13.7±1.4	13.7±1.3	14.7±1.4
26	12.4±1.4	12.4±1.4	13.0±1.4	13.5±1.4	12.6±1.4	12.6±1.4	13.4±1.4	12.8±1.4	12.8±1.4	14.1±1.5	14.7±1.5	14.7±1.4	16.0±1.5

Table 3b. Genetic distance matrix for D-Loop fragment sequence data for the 26 nominal sportive lemur species. The numbers represent the following *Lepilemur* species: 1. *L. ankaranaensis*, 2. *L. milanoii*, 3. *L. tymerlachsonorum*, 4. *L. dorsalis*, 5. *L. ahmansorum*, 6. *L. sahalazensis*, 7. *L. septentrionalis*, 8. Population from Montagne des Français, 9. *L. edwardsi*, 10. *L. grewcockorum*, 11. *L. otto*, 12. *L. microdon*, 13. *L. leucopus*, 14. *L. petteri*, 15. *L. randrianasoloi*, 16. *L. aeeclis*, 17. *L. ruficaudatus*, 18. *L. hubbardorum*, 19. *L. fleuretiae*, 20. *L. mustelinus*, 21. *L. betsileo*, 22. *L. jamesorum*, 23. *L. wrightae*, 24. *L. seali*, 25. *L. hollandorum*, 26. *L. scottorum*. Genetic distances based on absolute differences are displayed above the diagonal, and genetic distances based as a percentage are displayed below the diagonal.

	14	15	16	17	18	19	20	21	22	23	24	25	26
1	44.8	44.5	51.9	53.6	49.7	62.1	69.0	60.0	65.2	65.4	63.4	64.6	61.5
2	45.3	46.6	53.0	55.2	51.0	61.4	65.5	57.2	63.1	62.5	61.4	66.5	61.2
3	44.8	47.8	51.5	54.2	50.8	65.0	65.9	58.0	65.0	57.5	65.0	61.0	64.0
4	43.4	50.6	51.4	53.4	49.0	65.2	66.1	61.2	65.7	58.8	67.5	68.7	66.5
5	49.6	45.4	46.8	56.0	51.5	56.5	65.7	60.2	61.8	55.5	64.0	70.0	62.0
6	39.4	50.2	50.0	48.7	53.2	60.0	62.6	55.3	60.7	58.5	66.0	62.0	62.0
7	37.8	42.6	48.0	48.8	45.8	65.0	65.6	65.0	64.3	60.8	66.0	63.3	66.0
8	40.6	42.0	51.0	49.5	45.2	61.0	67.2	62.0	61.3	58.5	65.0	61.3	63.0
9	32.5	41.9	40.5	49.7	46.8	61.5	64.7	60.5	64.5	58.0	64.5	66.5	63.0
10	42.7	44.7	43.8	51.5	47.2	63.5	69.5	64.5	69.2	54.8	61.5	70.5	69.5
11	44.7	48.8	48.2	49.4	48.4	63.7	68.9	63.7	65.7	56.8	68.7	67.6	72.7
12	44.4	49.8	48.5	54.6	55.6	64.8	62.5	58.3	64.6	60.4	60.8	67.8	72.5
13	14.4	28.9	41.4	45.8	40.9	68.6	65.9	67.2	71.1	67.4	75.8	72.6	78.8
14		31.7	41.1	43.5	43.6	66.6	62.5	64.1	68.1	67.0	70.2	68.3	75.0
15	6.4±1.0		30.6	37.3	28.9	66.4	66.5	66.6	67.0	64.0	70.6	67.7	74.8
16	8.3±1.1	6.2±1.0		41.7	39.0	73.0	66.1	65.0	73.0	64.4	65.0	70.0	73.3
17	8.8±1.1	7.6±1.0	8.4±1.1		31.9	62.2	56.9	62.2	60.8	67.5	66.7	66.9	76.5
18	8.8±1.2	5.8±1.0	7.9±1.1	6.5±1.0		67.3	63.0	61.1	64.6	66.1	68.3	73.5	74.2
19	13.5±1.5	13.4±1.4	14.8±1.5	12.6±1.4	13.6±1.4		37.7	42.0	37.0	50.2	45.0	52.7	52.0
20	12.7±1.4	13.5±1.4	13.4±1.4	11.5±1.3	12.7±1.4	7.6±1.0		26.2	27.4	46.3	44.5	52.9	50.8
21	13.0±1.4	13.5±1.4	13.2±1.4	12.6±1.4	12.4±1.4	8.5±1.2	5.3±0.8		26.6	44.8	46.0	50.0	47.7
22	13.8±1.4	13.6±1.4	14.8±1.4	12.3±1.3	13.1±1.3	7.5±1.0	5.5±0.8	5.4±0.8		43.5	47.7	45.7	45.3
23	13.6±1.4	13.0±1.4	13.0±1.3	13.7±1.4	13.4±1.4	10.2±1.2	9.4±1.1	9.1±1.2	8.8±1.1		49.3	49.9	56.7
24	14.2±1.5	14.3±1.5	13.2±1.5	13.5±1.4	13.8±1.4	9.1±1.2	9.0±1.2	9.3±1.3	9.6±1.2	10.0±1.3		35.7	29.0
25	13.8±1.4	13.7±1.4	14.2±1.4	13.5±1.4	14.9±1.4	10.7±1.3	10.7±1.3	10.1±1.2	9.2±1.1	10.1±1.2	7.2±1.1		30.7
26	15.2±1.5	15.1±1.5	14.8±1.5	15.5±1.6	15.0±1.5	10.5±1.3	10.3±1.3	9.6±1.3	9.2±1.2	11.5±1.4	5.9±1.0	6.2±1.0	

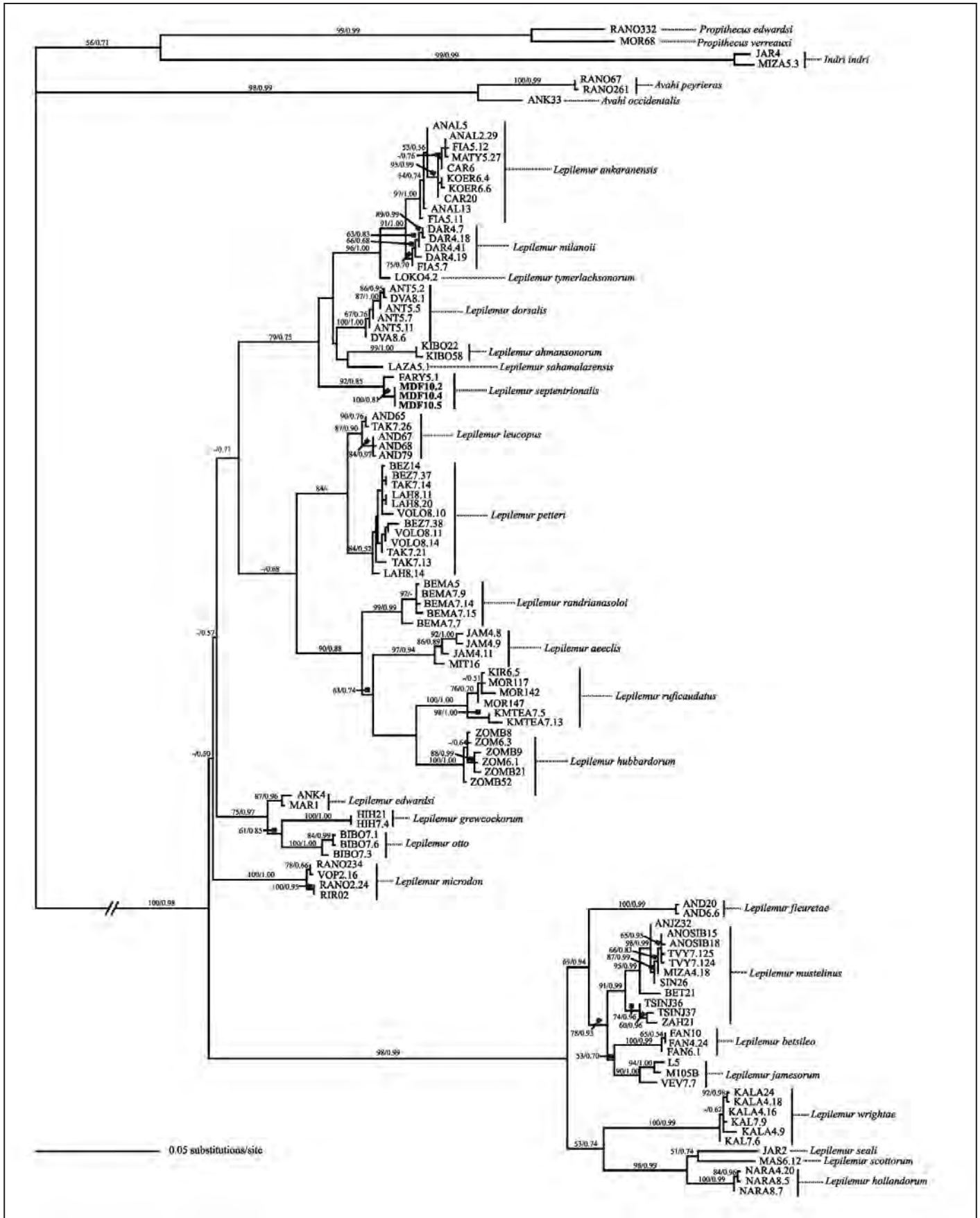


Figure 6. Phylogenetic relationships between *Lepilemur* species inferred from the maximum likelihood and Bayesian approaches of D-loop sequences from 108 sportive lemur individuals with seven outgroup taxa. BP/PP represent bootstrap support (BP) and posterior probability support (PP) values, respectively. We obtained the maximum likelihood phylogram (-ln likelihood = 6522.06) from the D-loop alignment from a transition/transversion ratio of 4.37 and a shape parameter of 0.73.

programs and community-based interactions are therefore the only viable solutions.

The development of an interconnected program between the VOI, the local gendarme and community can provide the platform for reducing the illegal charcoal activities and habitat loss in the last remaining vestige of habitat for the northern sportive lemur. Conservation strategies that emphasize education, monitoring, reforestation, and research should be implemented to leverage the biodiversity and rebuild the fragile ecosystem in northern Madagascar. This multi-layered program should be linked to sustainable alternative agricultural practices and the introduction of fuel-efficient rocket stoves and briquette technology. The best measures of success will be greater accessibility to food and increased income to the region, both of which are interconnected with habitat restoration and the long-term survival of the northern sportive lemur.

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Large-culmed Bamboos in Madagascar: Distribution and Field Identification of the Primary Food Sources of the Critically Endangered Greater Bamboo Lemur *Prolemur simus*

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Abstract: The greater bamboo lemur *Prolemur simus* is a Critically Endangered lemur endemic to eastern Madagascar. Wild *P. simus* populations have diets dominated by bamboo (Poaceae: Bambusoideae), particularly large-culmed species of two endemic genera, *Cathariostachys* and *Valiha*, but also of an endemic *Arundinaria* and the pantropical *Bambusa vulgaris*. A good understanding of the distribution and biology of large-culmed bamboos would, therefore, be a major help in understanding the factors influencing the distribution and conservation requirements of *P. simus*. During four years of research within the context of a collaborative programme aimed at conserving *P. simus*, we undertook surveys at 47 low-, mid- and high-elevation sites in eastern Madagascar between November 2008 and September 2012, covering a distance of over 1000 km from Makira in the north to Midongy in the south. We provide and illustrate characteristics helpful in the field identification, at least to generic level, of the large-culmed woody bamboos we encountered. The most frequently encountered non-endemic large-culmed bamboo was *Bambusa vulgaris*, with both the green and the yellow forms commonly planted in or near rivers, towns and villages. We made 1,343 georeferenced records of endemic, large-culmed bamboos, recording *Cathariostachys* from Makira to Midongy, *Valiha diffusa* from Makira to the southern parts of the COFAV, and *Arundinaria* from Zahamena to the COFAV. *Valiha diffusa* was recorded only at low elevations (9–576 m), and *Arundinaria* only at high elevations (1018–1667 m), while *Cathariostachys* was distributed across a wide range of elevations, from 53 to 1471 m, although most records were between 600 and 1,260 m. *Valiha diffusa* had the most variable culm diameters, ranging from 1.7 to 9.5 cm. *Cathariostachys* had the largest mean culm diameter of the endemic large-culmed bamboos (6.1 cm), and *Arundinaria* the smallest (4.0 cm). Culm diameters of *Bambusa vulgaris* were larger than those of the endemic bamboos, with a mean of 9.58. *Cathariostachys* had thinner mean culms at lowland compared to higher elevation sites. The revised distribution map we provide for *Cathariostachys* appears to correspond well to the potential current distribution of *P. simus* based on recent direct sightings, indirect feeding signs and unconfirmed local knowledge. *Cathariostachys* does not, however, currently occur throughout the historic range of *P. simus*. *Valiha* does occur at one *P. simus* subfossil site outside the current range, and therefore may have been the primary food source for the species there. Further research into the historical distribution of large-culmed bamboos in Madagascar would be very helpful in establishing how eventual changes in bamboo distributions over time may have affected changes in *P. simus* distribution.

Key words: Ankeniheny-Zahamena Corridor, Nosivolo, *Arundinaria*, *Cathariostachys capitata*, *Cathariostachys madagascariensis*, *Valiha diffusa*, *Yushania*

Introduction

Listed as Critically Endangered by IUCN (2012), the greater bamboo lemur (*Prolemur simus*) is considered to be one of the most endangered primates in the world (Wright *et al.* 2009). It is also listed in the top 50 most evolutionarily distinct and globally endangered mammal species (Collen *et al.* 2011). Endemic to Madagascar, sub-fossil records show it was historically one of the most widespread and abundant of the lemurs (Mahé 1976; Vuillaume-Randriamanantena *et al.* 1985; Godfrey and Vuillaume-Randriamanantena 1986; Godfrey *et al.* 2004). In recent years, however, the species was thought to have a much reduced range, in and near the south-eastern rainforests of the island (Mutschler and Tan 2003). Recent range extensions based on confirmed sightings have shown that the present-day range is not as diminished as previously thought (Dolch *et al.* 2008; King and Chamberlan 2010; Ravaloharimanitra *et al.* 2011; Rakotonirina *et al.* 2011), and indirect evidence suggests the species may still be widely distributed through much of eastern Madagascar (Dolch *et al.* 2010; Rakotonirina *et al.* 2011, 2013). Confirmed sightings in recent years have been made in the remaining mid- to high-elevation rainforest corridors from Didy to Andasibe (Dolch *et al.* 2008; Ravaloharimanitra *et al.* 2011; Randrianarimanana *et al.* 2012; Olson *et al.* 2012), and from the Ranomafana National Park to the Andringitra National Park (Petter *et al.* 1977; Wright *et al.* 2008; Delmore *et al.* 2009). They have also been made in lowland, often degraded, landscapes in the Brickaville District (Ravaloharimanitra *et al.* 2011; Bonaventure *et al.* 2012; Lan-tovololona *et al.* 2012; Mihaminekena *et al.* 2012), the Vatomandry District (Rakotonirina *et al.* 2011), at the confluence of the Mangoro and Nosivolo rivers in the Mahanoro District (Rakotonirina *et al.* 2011; Andrianandrasana *et al.* in press), around Kianjavato in the Mananjary District (Meier and Rumppler 1987; Andriaholinirina *et al.* 2003; Wright *et al.* 2008; McGuire *et al.* 2009), and near Karianga in the Vondrozo District (Wright *et al.* 2008, 2009). The elevation range for confirmed sightings is 20 m (Bonaventure *et al.* 2012) to 1,600 m (Goodman *et al.* 2001).

Recent authors consider *Prolemur simus* to represent a monospecific genus (Mittermeier *et al.* 2008 and references therein), although it has previously been considered congeneric with the other bamboo lemurs *Hapalemur* spp. (Tattersall 1982). Most wild *Prolemur* and *Hapalemur* populations have diets dominated by bamboos (Poaceae: Bambusoideae; Mutschler and Tan 2003; Tan 2007), exceptions being some *Hapalemur* populations which occur in sites with little or no bamboo, and which feed primarily on other monocotyledon plants, particularly members of the grass (Poaceae) and sedge (Cyperaceae) families (for example, *H. alaotrensis* in wetlands at Lac Alaotra, Mutschler 1999, and *H. meridionalis* in littoral forest at Mandena, Eppley *et al.* 2011).

Two lemurs, *Prolemur simus* and the golden bamboo lemur *Hapalemur aureus*, appear to be particularly dependent on bamboo (Tan 1999). Both species must have specializations allowing them to feed on bamboo parts rich in cyanide,

although the exact mechanisms by which they avoid cyanide poisoning have not been identified (Tan 2007; Ballhorn *et al.* 2009; Yamashita *et al.* 2010). *Prolemur simus* has specializations allowing it to exploit the toughest and most mechanically-challenging parts of woody bamboos avoided by *Hapalemur*, such as the mature culm pith and mature leaf-blades of the large-culmed *Cathariostachys madagascariensis* (see Vinyard *et al.* 2008; Yamashita *et al.* 2009), and all known wild *P. simus* populations occur in sites containing large-culmed woody bamboos (Tan 2007; Rakotonirina *et al.* 2011; Ravaloharimanitra *et al.* 2011).

Madagascar is home to a large diversity of woody bamboos. Dransfield (2000, 2003) lists 32 named species endemic to Madagascar (Table 1). The taxonomy of a number of them is undergoing revision. Two are now considered conspecific (*Schizostachyum bosseri* and *S. parvifolium* synonyms of *Sirochloa parvifolium*, Dransfield 2002), several are currently placed in incorrect genera (in, for example *Arundinaria* and *Cephalostachyum*, Dransfield 2000, 2003), and at least three species remain to be described (Dransfield 2003). A further five named species are either pantropical or introduced (Dransfield 2003; Table 1).

Prolemur simus appears to be particularly associated with large-culmed species of two endemic genera, *Cathariostachys* and *Valiha*, especially *C. madagascariensis* in mid- to high-elevation rainforest, and *V. diffusa* in lowland secondary habitats (Tan 2007; Rakotonirina *et al.* 2011; Ravaloharimanitra *et al.* 2011). In addition to these two genera, *P. simus* feeds on other large-culmed bamboos, such as an endemic *Arundinaria* species (Randrianarimanana *et al.* 2012, as “volotsanganana”) and the pantropical *Bambusa vulgaris* (Ravaloharimanitra *et al.* 2011; Mihaminekena *et al.* 2012), the latter species possibly introduced, being found mainly along rivers or near villages (Dransfield 2003). Various studies indicate that these large-culmed bamboos make up over 90% of the diet of *P. simus* in the wild (Tan 1999; Mihaminekena *et al.* 2012; Randrianarimanana *et al.* 2012).

This close association between *Prolemur simus* and large-culmed bamboos implies that a good understanding of the distribution and biology of large-culmed bamboos would be a major help in understanding the factors influencing the distribution, abundance, biology and conservation requirements of this Critically Endangered lemur. However, very little is known about any of the woody bamboos of Madagascar (Dransfield 2003). We therefore present in this paper a summary of what we have learnt about large-culmed bamboos in Madagascar during four years of surveys and research in the context of a collaborative programme aimed at conserving *P. simus* (King and Chamberlan 2010; Rakotonirina *et al.* 2011). We update our knowledge on the distribution, elevation ranges, and culm diameter of large-culmed bamboos occurring in the range of *P. simus*, and provide and illustrate characteristics helpful in their field identification, at least to generic level. We hope that this analysis will facilitate further work on various aspects of the endemic bamboos of Madagascar, as suggested by Dransfield (2003).

Table 1. Woody bamboo species recorded from Madagascar, based on 1: Clayton *et al.* (2010); 2: Dransfield (1994); 3: Dransfield (1997); 4: Dransfield (1998); 5: Dransfield (2000); 6: Dransfield (2002); 7: Dransfield (2003); 8: *The Plant List* (2010).

Species (1, 6, 7)	Culm diameter (cm)	Elevation (m)	Notes
Introduced/pantropical			
<i>Phyllostachys aurea</i>	0.2–0.3 (1)		Naturalized everywhere around Antananarivo (7)
<i>Bambusa vulgaris</i>	4–10 (1,7)		Occurs nears villages or along rivers (7)
<i>Bambusa multiplex</i>	1–3 (1)		Usually planted as hedges (7)
<i>Dendrocalamus giganteus</i>	20–30 (1)		Planted at various places along the east coast (7)
<i>Gigantochloa</i> aff. <i>pseudoarundinacea</i>	5–10 (1)		Planted near Maroantsetra (7); synonym of <i>Gigantochloa</i> aff. <i>verticillata</i> (8)
Native, erect culms			
<i>Cathariostachys capitata</i>	4–5 (4)	5–700 (3)	Lowland to hill primary forest, and also disturbed forest, mainly east coast (5,7)
<i>Cathariostachys madagascariensis</i>	5–8(–12) (4)	900–1000 (7)	Lower montane forest, Andasibe to Ranomafana (7)
<i>Valiha diffusa</i>	(1.5–)7–10 (4)	50–700 (7)	Primary forest and open hills, from Nosy Be throughout eastern slopes to Ifanadiana (7)
<i>Valiha perrieri</i>			Forest, Andrafiarana, south of Anstiranana (4,5,7)
<i>Valiha</i> sp.			Dry forest, Morondava (7)
<i>Arundinaria ibityensis</i>	2 (1)	1800–2250 (5,7)	Rocky ridges, Mt Ibity, Antsirabe (5,7); synonym of <i>Thamnocalamus tessellates</i> (8)
<i>Arundinaria ambositrensis</i>		1300–1400 (5,7)	Forest, Ranomena (Ambositra) and Ranomafana (5,7); synonym of <i>Yushania humbertii</i> (8)
<i>Arundinaria humbertii</i>		2000 (5,7)	Forest, Andringitra (5,7); synonym of <i>Yushania humbertii</i> (8)
<i>Arundinaria perrieri</i>	5–8 (1)	1000 (5,7)	Forest, Manongarivo (5,7); synonym of <i>Yushania perrieri</i> (8)
<i>Arundinaria madagascariensis</i>		2000 and 2800 (5,7)	Mossy forest, Tsaratanana (5,7); synonym of <i>Yushania madagascariensis</i> (8)
<i>Arundinaria marojejyensis</i>		2000 (5,7)	Mossy forest, Marojejy (5,7); synonym of <i>Yushania madagascariensis</i> (8)
<i>Schizostachyum perrieri</i>		2000 (5,7)	Forest, Tsaratanana (5,7); maybe conspecific with <i>Arundinaria madagascariensis</i> (7)
<i>Decaryochloa diadelpha</i>	2–2.5 (3,5)	800–1000 (3); 900 (7)	Montane forest, common at Andasibe, but died in 1994 following flowering (3,7)
<i>Nastus elongatus</i>	3 (1)	1000 (5,7)	Montane or mossy forest, Andringitra and Ranomafana (5,7)
<i>Hickelia madagascariensis</i>	1–1.8 (1,2)	1000–1600 (7)	Relatively common in montane forests of Central Highlands (7)
<i>Hickelia alaotrensis</i>		1500 (5,7)	Forest at Lac Alaotra (7)
<i>Hickelia perrieri</i>		2400 (5,7)	Tsaratanana (7)
<i>Sirochloa parvifolium</i> (6)	≤ 0.7 (6)	0–70 (6,7)	Very common on white sands, Nosy Be and coastal regions of east (7); occasionally found a long way from the shore at up to 70 m a.s.l. (6); synonym of <i>Sirochloa parvifolia</i> (8)
<i>Perrierbambus madagascariensis</i>	2–3 (1)	Lowlands (5,7)	Dry lowland forest at Locky, near Mahajanga (5,7)
<i>Perrierbambus tsarasaotrensis</i>	2–3 (1)		Tsarasaotra (5,7)
Native, climbing culms			
<i>Cephalostachyum chapelieri</i>			Lower montane forest, Analamazaotra, Andasibe (7)
<i>Cephalostachyum viguieri</i>	1.5–3 (1)	50–1200 (5,7)	Widespread, from Masoala to Andasibe and maybe Ranomafana (5,7)
<i>Cephalostachyum perrieri</i>		c.900 m (5,7)	Mananara Nord and maybe Ranomafana (7)
<i>Cephalostachyum</i> spp.		c.50 m (7)	Two undescribed species in forest, Masoala (7)
<i>Hitchcockella baronii</i>	0.1–0.15 (1)		Montane forest on Manongarivo Massif (7)
<i>Nastus aristatus</i>		900–1200 (5,7)	Montane forest, Manongarivo to Andasibe (5,7)
<i>Nastus emirnensis</i>	0.1–0.2 (1)	1000 (5,7)	Montane forest, Analamazaotra, Andasibe (5,7)
<i>Nastus humbertianus</i>			Andohahela (5,7)
<i>Nastus lokohoensis</i>			Forest, Lokoho (5,7)
<i>Nastus madagascariensis</i>	1 (1)		Forest, Central plains (5,7)
<i>Nastus manongarivensis</i>		500–1600 (5,7)	Forest, Manongarivo (5,7)
<i>Nastus perrieri</i>		1700 (5,7)	Mossy forest, Tsaratanana (5,7)
<i>Nastus tsaratananensis</i>		2000 (5,7)	Mossy forest, Tsaratanana (5,7)

Methods

We carried out surveys at 47 low-, mid- and high-elevation sites in seven major survey regions of eastern Madagascar between November 2008 and September 2012, covering a distance of over 1000 km, from Makira in the north to Midongy in the south (Table 2). Each site falls in the “humid forest” or adjacent “degraded humid forest” vegetation classifications described by Moat and Smith (2007). The primary purpose of the surveys was to discover, and subsequently conserve, new populations of *Prolemur simus* (see King and Chamberlan 2010), and we have published elsewhere more detailed habitat descriptions for most of the sites, along with many of our primary results (Rajaonson *et al.* 2010; Ravaloharimanitra *et al.* 2011; Rakotonirina *et al.* 2011, 2013; Bonaventure *et al.* 2012; Lantovololona *et al.* 2012; Mihaminekena *et al.* 2012; Randrianarimanana *et al.* 2012; Andrianandrasana *et al.* in press; Rajaonson and King in press).

We recorded thousands of GPS points during these surveys. In this paper, we analyse a subset of these GPS points referring to the location of large-culmed bamboos. We mapped them using MapInfo GIS software, to illustrate their distribution across Madagascar, and calculated summary statistics and created scatter graphs to illustrate elevation ranges for each genus.

We measured culm diameters at breast height (dbh) of large-culmed bamboos at ten sites in six of the survey regions, and we present summary statistics for dbh of each genus at each of these sites. We used the *z* test for unmatched samples (Chalmers and Parker 1989) to test for differences in mean culm diameter at different sites, or under different

canopy openness at some sites, for samples with at least 25 measurements.

We photographed bamboos at each survey region, and sent photos to Soejatmi Dransfield to help with initial identifications. One of us (TK) verified that photos of each taxon identified at a survey area were available for that area. We also referred to the detailed botanical descriptions of *Valiha diffusa* and *Cathariostachys* spp. given by Dransfield (1998), and of *Bambusa vulgaris* and *Dendrocalamus giganteus* given by Clayton *et al.* (2010). We follow Dransfield (2003) for bamboo nomenclature, although we also refer to synonyms for some species following *The Plant List* (2010). Notably, *The Plant List* (2010) does not use the genus *Arundinaria* for any Malagasy bamboos, placing several in *Yushania* and one in *Thamnocalamus* (Table 1). We follow Mittermeier *et al.* (2010) for lemur nomenclature.

Results

Field identification of large-culmed bamboos

With care, identification of large-culmed woody bamboos within our survey regions was relatively simple, at least to genus. Of the large-culmed clumping bamboos (Fig. 1), the most frequently encountered was *Bambusa vulgaris*, with both the green and the yellow forms commonly planted in or near rivers, towns and villages, and we recorded it in the Makira, CAZ, Nosivolo and COFAV survey regions. The larger *Dendrocalamus giganteus* was less frequently observed. Of the non-clumping species, *Valiha diffusa* was easily recognizable within its range by its characteristic drooping culm tips, long lateral branches, stiff culm sheaths readily shed, and a thin



Figure 1. The pantropical bamboo *Bambusa vulgaris* (left: green form; centre: yellow form) and the introduced *Dendrocalamus giganteus* (right). Photographs by Tony King and Maherisoa Ratolojanahary.

whitish ring above the internodes following the shedding of the culm sheath (Fig. 2). *Cathariostachys* spp. often lacked major lateral branches, had persistent culm sheaths that were not readily shed, and often showed a pale ring under the internodes (Figs. 3–6). *Arundinaria* spp. had slimmer culms, with delicate culm sheaths often appearing torn or shredded, and a distinctive pattern of leaf growth (Figs. 7–10). The young emerging culm shoots of each genus were also highly distinctive (Figs. 1–10).

Local names for bamboos were very variable. *Valiha diffusa* was usually referred to as Vologasy or Volojatsy, although both names were used for other species in some places. *Cathariostachys* bamboos were usually called either Volohosy, Volo lagnana or Volobe, but in lowland areas Volobe was more often used for the green form of *Bambusa vulgaris*. The yellow form of *B. vulgaris* was usually called Volovanga or Volovazaha. *Arundinaria* bamboos were almost always called Volotsanganana, and we only rarely heard this name used for other bamboos. The use of local names varied from region to region, but also between local people in the same region—sometimes simply due to misidentification of bamboos to

the appropriate local name by less-experienced local guides. Names used for various smaller-culmed bamboos included Volohando, Volohoto, Voloandotra, Volomadinika, Volotami-hana, Volosodina, Volokitrana, Volovahy and Tsimbolovolo.

Distribution and elevation ranges of Valiha diffusa, Cathariostachys and Arundinaria

We made 1,343 georeferenced records of endemic large-culmed bamboos (*Valiha diffusa*, *Cathariostachys* and *Arundinaria*) across the seven survey regions. The majority of our observations (74%, n = 994) were made in and around the Ankeniheny-Zahamena forest corridor (CAZ), where we have long-term monitoring of *Prolemur simus* populations. One region had only one observation (Anjozorobe), the remaining five had between 43 and 129 observations each. *Cathariostachys* bamboos were recorded from Makira in the north to Midongy in the south, *Valiha diffusa* from Makira to the southern parts of the COFAV, and *Arundinaria* bamboos from Zahamena to the COFAV (Table 2; Figs. 11–14).

Valiha diffusa was only recorded at low elevations (9–576 m), and *Arundinaria* only at high elevations



Figure 2. The Malagasy endemic bamboo *Valiha diffusa* in lowland deforested landscapes of eastern Madagascar. Photographs by Lucien Randrianarimanana and Hery Randriaingo.

Table 2. Geographic coordinates of the 47 survey sites, within seven survey regions, with elevation ranges at each site for endemic large-culmed bamboos *Valiha diffusa*, *Cathariostachys* spp. and *Arundinaria* spp.

Region	Survey site	Latitude (°S)	Longitude (°E)	GPS points	Elevation range (m)		
					<i>Valiha diffusa</i>	<i>Cathariostachys</i> spp.	<i>Arundinaria</i> spp.
Makira	Antohaka Lava	14.73 to 14.79	49.71 to 49.75	18		454–1030	
	Mahevaratra	14.83 to 14.90	49.73 to 49.80	35	403–552	340–725	
Zahamena	Andringitra	17.64 to 17.66	48.66 to 48.67	15			1171–1164
	Volotsanganana	17.69 to 17.73	48.74 to 48.79	28		845–955	
Anjozorobe	Saha Forest Camp	18.41	47.94	1			1322
CAZ	CAZ: Didy	18.12 to 18.17	48.64 to 48.73	11		868–1007	
	CAZ: Ranomainty	18.30 to 18.39	48.44 to 48.50	352		1042–1471	1084–1467
	CAZ: Fierenana	18.42 to 18.45	48.44 to 48.59	17		992–1334	1048–1268
	CAZ: Morarano	18.66 to 18.76	48.35 to 48.43	285		893–1256	
	CAZ: Ankerana	18.47	48.82 to 48.83	2		496–516	
	Ambohimanana	18.46 to 18.53	48.72 to 48.79	8	172–370	464–678	
	Maroseranana	18.58	48.89	4	127–303		
	Vohimientana	18.59	48.76	1	145		
	Andriantantely NE	18.63 to 18.68	48.79 to 48.82	59	68–332		
	Andriantantely SE	18.71 to 18.74	48.79 to 48.83	86	103–229		
	Sahavola	18.69	48.97 to 48.98	100	26–148		
	Vohiposa	18.75 to 18.79	48.93 to 48.95	26	46–191		
	Ambalafary	18.8	48.81	35	67–124		
	Andekaleka	18.81	48.58	2	391–407		
	Ranomafana Est	18.95	48.8	1	63		
	Nosivolo	Mahatsara	18.98	48.92 to 48.93	2	16–22	
Mahalina		19.07	48.85 to 48.88	3	64–82		
Antanambao Manampotsy		19.42 to 19.45	48.56 to 48.57	2	196–217		
Tsinjoarivo		19.67 to 19.72	47.77 to 47.83	57			1294–1667
Ambohimiadana		19.89 to 19.90	47.87 to 47.92	23		1022–1072	1348–1422
Ambohimalaza		19.97 to 19.98	47.89	6		887–999	
Vohibe & Vohitrambo		19.92 to 19.96	48.46 to 48.50	37	98–576	236–536	
COFAV	Andranambolava	20.21	48.15	2		668–680	
	Beranomintina	20.25	48.13	2		784–791	
	Mananjary	21.09 to 21.18	48.18 to 48.22	17	21–68		
	Ranomafana NP	21.25	47.36 to 47.42	9		951–972	1122–1158
	Ambindrabe & Ambendrana	21.37 to 21.40	47.34 to 47.40	8			1032–1227
	Antarehimamy & Ambodiara	21.89 to 21.91	47.33 to 47.35	4	824		1018–1075
	Antaranjaha	21.97 to 22.01	47.43 to 47.44	5	324–367	302–316	
	Sahalanona	22.05 to 22.10	47.59 to 47.63	22	129–213		
	Andringitra: Manambolo	22.07	46.99	1			1238
	Sahamadio	22.52	47.58	1	27		
	Mahazoarivo: Ifasy	22.65	47.25	1	203		
	Mahazoarivo: Mitimbato	22.66	47.32	1	147		
Mahafasa		22.67	47.68	1	41		
	Iandraina	22.77	47.69	1		53	

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Table 2. continued

Region	Survey site	Latitude (°S)	Longitude (°E)	GPS points	Elevation range (m)		
					<i>Valiha diffusa</i>	<i>Cathariostachys</i> spp.	<i>Arundinaria</i> spp.
Midongy	Ambalavero	23.14 to 23.16	47.18 to 47.21	16		480–610	
	Ambadikala	23.29 to 23.30	47.30 to 47.36	2		172–187	
	Marovovo	23.38	47.12	3		649–751	
	Marovato	23.47	47.06 to 47.08	3		687–770	
	Halampo	23.63 to 23.67	46.91 to 46.95	9		718–890	
	Ampasy	23.73 to 23.78	47.02 to 47.03	19		708–961	
Total	47 sites	14.73 to 23.78	46.91 to 49.80	1343	9–576	53–1471	1018–1667
					n=396	n=710	n=237

Figure 3. *Cathariostachys* sp. near Makira in north-eastern Madagascar. Photographs by Lucien Randrianarimanana.

(1018–1667 m), while *Cathariostachys* bamboos were distributed across a wide range of elevations, from 53 to 1471 m (Table 2, Fig.11). Low-elevation records of *Cathariostachys* were rare (Fig. 11), with only five of 710 records (0.7%) below 300 m, 12 (1.7%) below 400 m, and 65 (9.2%) below 600 m. Very high elevation records of *Cathariostachys* were also rare, with only three records (0.4%) over 1,300 m. Most records of *Cathariostachys* could therefore be considered mid-elevation, with 90.4% ($n = 642$) between 600 and 1,260 m.

At the most northerly region surveyed, Makira (c. 14.8°S), there appeared to be a continuum of elevation records for *Cathariostachys*, from 340 to 1040 m (Fig. 11). At our more central survey regions, from 17 to 22°S (Zahamena, CAZ, Nosivolo, COFAV), there appeared to be two groups of elevation records, below 600 and above 800 m, with very few records between 600 and 800 m (Fig. 11). At the most southerly region, Midongy (23 to 24°S), most records were between 400 and 1000 m, with two below 200 m and none between 200 and 400 m (Fig. 11).

Culm diameters

Valiha diffusa had the most variable culm diameters at our sites, ranging from 1.7 to 9.5 cm (Table 3). *Cathariostachys* had the largest mean culm diameter of the endemic large-culmed bamboos (6.1 cm), and *Arundinaria* the smallest (4.0 cm). Culm diameters of the green form of the pantropical *Bambusa vulgaris* (measured at the Ambalafary site to the east of the CAZ) were larger than those of the endemic bamboos, ranging from 3.2 to 13.4 cm, with a mean of 9.58 cm (SD 2.04, $n = 248$).

There were some inter-site differences in culm diameters of the endemic bamboos (Table 3). For example, the *Cathariostachys* species at the lowland sites of Maherivaratra (Makira region) and Vohibe (Nosivolo region) had mean culm diameters one to 2.5 cm smaller than at the other, higher elevation, sites (Table 3). This is a statistically significant difference, when comparing each with the site with the next lowest mean *Cathariostachys* culm dbh measured by the same researcher (Maherivaratra with Ranomainty: $z = 14.08$, $p < 0.001$; Vohibe with Zahamena: $z = 8.80$, $p < 0.001$).



Figure 4. *Cathariostachys madagascariensis* in the western Ankeniheny-Zahamena Corridor in eastern Madagascar. Photographs by Lucien Randrianarimanana and Tony King.

Table 3. Summary statistics for culm dbh measurements for endemic large-culmed bamboos at some of the survey sites.

Survey region	Elevation	Valiha diffusa			Cathariostachys spp.			Arundinaria spp.		
		DBH Range	Mean (SD)	n	DBH Range	Mean (SD)	n	DBH Range	Mean (SD)	n
Makira: Maherivaratra	340–725 m	3.7–6.5	5.48 (0.71)	26	3.6–5.5	4.52 (0.60)	25			
Zahamena	863–1298 m				5.1–8.3	6.24 (0.85)	85	3.5–5.7	4.69 (0.49)	135
Anjozorobe	1322 m							3.0–4.8	3.83 (0.61)	20
CAZ: Ranomainty	1025–1471 m				5.0–8.4	6.84 (0.72)	42	2.3–4.8	3.58 (0.54)	59
CAZ: Sakalava	893–1256 m				3.2–8.9	7.08 (1.07)	478			
CAZ: Ambalafary	48–124 m	2.9–7.3	5.39 (0.93)	231						
CAZ: Sahavola	26–148 m	2.9–9.2	6.15 (1.49)	103						
CAZ: Vohiposa	46–191 m	3.5–7.0	5.33 (1.05)	12						
Nosivolo: Vohibe	167–576 m	1.7–9.5	4.42 (1.43)	76	2.4–7.2	5.18 (0.88)	127			
Midongy	687–961 m				5.5–8.5	6.82 (0.90)	17			
Total		1.7–9.5	5.4	448	2.4–8.9	6.1	774	2.3–5.7	4.0	214

**Figure 5.** *Cathariostachys* sp. in and around the Midongy du Sud National Park in south-eastern Madagascar. Photographs by Laingo Rakotonirina.

The mean culm diameter of *Valiha diffusa* was 0.6 cm lower in open areas than in areas of closed or semi-closed canopy cover at both lowland sites where this was measured, a statistically significant difference at both sites (Table 4).

Table 4. Comparison of mean culm diameters (dbh) of *Valiha diffusa* under differing levels of canopy cover at two lowland sites in the CAZ survey region, using the *z* test for unmatched samples.

		Mean DBH	SD	N	<i>z</i>	<i>p</i>
Amabalafary						
	Open canopy	5.27	0.91	181	4.31	<0.001
	Closed canopy	5.85	0.83	50		
Sahavola						
	Open canopy	5.83	1.57	48	2.05	<0.05
	Semi-closed canopy	6.43	1.37	55		

Discussion

Distribution of large-culmed bamboos in Madagascar

By comparing the distribution of our field observations with georeferenced published herbarium specimen locations (Table 5), our study has resulted in a southern extension of the published range of *Valiha diffusa* (Fig. 12), and southern and northern range extensions for species of *Cathariostachys* (Fig. 13). We have also shown that *Arundinaria* occurs in remaining high elevation forests in the Zahamena National Park, the Ankeniheny-Zahamena Corridor, the Anjozorobe Corridor, and around the high-reaches of the Nosivolo River (Fig. 14). We recommend the collection of herbarium specimens from these sites to help resolve the taxonomic uncertainties surrounding the Malagasy bamboos currently included in *Arundinaria* (Dransfield 2000, 2003; or alternatively in *Yushania* and *Thamnocalamus* by *The Plant List* 2010).

Herbarium specimens are also needed to help determine species' distributions within *Cathariostachys*, as we were unable to differentiate the two recognized *Cathariostachys* species in the field, and also found no consistent pattern that



Figure 6. *Cathariostachys* sp. (probably *C. capitata*) in the lowland forest fragment of Vohibe in eastern Madagascar. Photographs by Anjara Bonaventure, Anselmo Andrianandrasana and Tiana Ratolojanahary.

might separate them based on elevation. It may be that there is only one, highly variable, *Cathariostachys* species, as suggested by Dransfield (1998), although the thinner mean culm diameter of *Cathariostachys* at our lowland sites of Maherivaratra and Vohibe compared to our higher elevation sites is consistent with the descriptions of the two species given by Dransfield (1998). The species identity of *Valiha* in the Ankarana National Park in the far north of Madagascar also requires further study (Dransfield 2003). Although close to the type locality of *Valiha perrieri*, the large-culmed bamboos of Ankarana appear very similar to *Valiha diffusa* (L. Rakotonirina and T. King, unpubl. data), and probably are indeed the latter (S. Dransfield, in litt. 2012).

Bamboo distribution is often affected by disturbance (Griscom and Ashton 2006; Gagnon *et al.* 2007; Olson *et al.* 2013). However, disturbance may affect different bamboo species differently. *Cathariostachys madagascariensis* appears to be found at higher densities in disturbed forests than in non-disturbed forests, perhaps indicating an evolutionary

adaptation to regular cyclonic disturbance in the eastern rainforests (Olson *et al.* 2013), but does not appear to tolerate excessive disturbance, and does not persist in deforested areas far from the forest edge (Fig. 13; see also Olson *et al.* 2013). *Valiha diffusa*, conversely, appears to thrive in many deforested landscapes of lowland eastern Madagascar (Figs. 2 and 12; Dransfield 1998), and is probably more abundant now than when lowland forest cover was more extensive. However, although apparently more tolerant to major disturbance than *Cathariostachys*, *V. diffusa* probably also has a limit to its tolerance of disturbance, with culm diameters reduced in heavily-disturbed areas where it is cut or burnt regularly (Dransfield 1998); a phenomenon common to several bamboo species (Franklin *et al.* 2010). We also found that *V. diffusa* culm diameters were lower in more open habitats, possibly due to higher rates of culm production in open areas (Gagnon *et al.* 2007), or perhaps due to environmental stress related to extreme micro-habitat variables such as moisture or light levels (Reid *et al.* 1991).

Table 5. Herbarium specimen locations for *Valiha* spp., *Cathariostachys* spp., and *Arundinaria* spp., given by Dransfield (1998, 2003), georeferenced for this study.

Species	Location	Elevation (m)	Longitude (°E)	Latitude (°S)
<i>Cathariostachys madagascariensis</i>	Analamazoatra	800, 900, 1000	48.43	18.93
	East of Moramanga		48.30	18.95
	Ranomafana National Park	900, 950	47.42	21.26
<i>Cathariostachys capitata</i>	Masoala Peninsula, Antalavia, near Ambanizana	5 and 200	49.97	15.65
	Mananara	700	49.68	16.18
	Antananambe Biosphere Reserve, Ambolokely	235	49.76	16.43
	Soanierana-Ambedra	200	49.56	16.88
	Fandrangato			
	(Farafangana), Vondrozo	500	47.32	22.82
	Varahina			
<i>Valiha diffusa</i>	Nosy-be	Low	48.25	13.33
	Nossi Camba (Komba)	Low	48.34	13.47
	Sambirano		48.44	13.77
	Massif of Manongarivo		48.42	13.93
	Marojejy	200 to 500	49.63	14.54
	Mananara	300	49.76	16.21
	Toamasina (Tamatave)	30	49.37	18.15
	Ampasimanolotra (Brickaville)	50	49.06	18.82
	Fianarantsoa, Ifanadiana, below Mt. Vatovavy	150	47.94	21.40
	Fianarantsoa, 10 km east of Ifanadiana	Low	47.74	21.33
<i>Valiha perrieri</i>	Andrafianamena	Low	49.41	12.81
<i>Arundinaria ambositrensis</i>	Ranomena (Ambositra)	1300 to 1400	47.48	20.54
	Ranomafana	1300 to 1400	47.37	21.24
<i>Arundinaria humbertii</i>	Andringitra	2000	46.90	22.24
<i>Arundinaria ibityensis</i>	Mt. Ibity, Antsirabe	1800 to 2250	47.01	20.13
<i>Arundinaria madagascariensis</i>	Tsaratana	2000 and 2800	48.87	14.03
<i>Arundinaria marojejyensis</i>	Marojejy	2000	49.72	14.42
<i>Arundinaria perrieri</i>	Manongarivo	1000	48.38	13.99

Large-culmed bamboos and *Prolemur simus*

At least 29 species have been recorded as food plants of *Prolemur simus*, of which nine are woody bamboos, ten are other monocotyledons, and ten are dicotyledons (Table 6). The Poaceae family (including bamboos) makes up 45% of recorded food plants. At sites where quantitative studies have been undertaken, over 90% of the feeding time of *P. simus* is devoted to one or two of the large-culmed bamboo species *Cathariostachys madagascariensis*, *Valiha diffusa*, *Bambusa vulgaris* and *Arundinaria* sp. (Tan 1999; Mihaminekena *et al.* 2012; Randrianarimanana *et al.* 2012). *Bambusa vulgaris* may or may not be native to Madagascar (Dransfield 2003), and does not occur in natural forest in the country (this study). The native large-culmed bamboos of *Cathariostachys*, *Valiha* and *Arundinaria* are, therefore, likely to be the primary natural food plants of *P. simus*. Of these, *Arundinaria* species are restricted to high elevation forest, and appear to be less frequently consumed by *P. simus* than the more widespread *Cathariostachys* species and *Valiha*.

If *Cathariostachys* and *Valiha* can therefore be considered the principle food plants of *P. simus*, their distribution should be one of the major factors influencing the range of

P. simus itself. The distribution map we provide for *Cathariostachys* (Fig. 13) does appear to correspond well to the potential current distribution of *P. simus* (Fig. 15) based on recent direct sightings and indirect feeding signs (as given by Rakotonirina *et al.* 2011) and unconfirmed local knowledge (as given by Dolch *et al.* 2010), with perhaps the notable exception of the Masoala peninsula in the north-east of the country from where there is currently no indication of the presence of *P. simus*.

Sub-fossil records of *Prolemur simus* are more widely distributed (Godfrey *et al.* 2004), occurring in areas where *Cathariostachys* species do not occur, at least in recent times (Fig. 15). However, *Valiha* does occur in at least one of these sub-fossil sites (Ankarana), and therefore may have been the primary food plant of the species in that area, and conceivably could still be so (L. Rakotonirina and T. King, unpubl. data). There is also an unidentified bamboo resembling *Valiha* in western dry forest near Morondava (Dransfield 2003), which may have been the food plant of *P. simus* when it existed in that part of the country. Further research into the historical distribution of large-culmed bamboos in Madagascar would be very helpful in establishing how eventual changes in



Figure 7. *Arundinaria* sp. in high elevation forest at Anjozorobe in eastern Madagascar. Photographs by Tony King.

Table 6. Recorded foodplants of *Prolemur simus* at five sites in eastern Madagascar, based on 1: Petter *et al.* 1977; 2: Meier & Rumpler 1987; 3: Tan 1999; 4: Tan 2007; 5: McGuire *et al.* 2009; 6: Ravaloharimanitra *et al.* 2011; 7: Rakotonirina *et al.* 2011; 8: Bonaventure *et al.* 2012; 9: Mihaminekena *et al.* 2012; 10: Lantovolo-lona *et al.* 2012; 11: Randrianarimanana *et al.* 2012; 12: L. Randrianarimanana, unpubl. report 2011; 13: Andrianandrasana *et al.* in press.

Family	Species	English name / description	Ranomafana NP	CAZ west	Brickaville District	Vohibe	Kianjavato
			Mid to high elevation		Low elevation, degraded habitats		
Large-culmed bamboos							
Poaceae	<i>Cathariostachys madagascariensis</i>	Erect bamboo	3	11			5 (in error?)
Poaceae	<i>Cathariostachys</i> cf. <i>capitata</i>	Erect bamboo				13	
Poaceae	<i>Valiha diffusa</i>	Erect bamboo			6,8,9,10	7,13	2 ^a ,4
Poaceae	<i>Arundinaria</i> sp.	Erect bamboo		11 ^b			
Poaceae	<i>Bambusa vulgaris</i>	Erect bamboo			6, 9		
Other bamboos							
Poaceae	<i>Nastus elongatus</i>	Erect bamboo	3				
Poaceae	<i>Cephalostachyum</i> cf. <i>perrieri</i>	Viny bamboo	3				
Poaceae	<i>Cephalostachyum</i> sp.	Viny bamboo	3,4	12		13	
Poaceae	<i>Nastus</i> sp.	Viny bamboo		11 ^c			
Poaceae	?	Viny bamboo					2
Other monocots							
Arecaceae	<i>Dypsis</i> sp.	Palm		11		13	1,2
Bromeliaceae	<i>Ananas comosus</i>	Pineapple					5
Cyperaceae	<i>Scleria</i> sp.	Sedge	3				
Musaceae	<i>Musa</i> sp.	Banana			6		
Poaceae	<i>Saccharum</i> sp.	Sugar cane			6		
Poaceae	<i>Pennisetum clandestinum</i>	Kikuyugrass					2
Poaceae	<i>Oryza</i> sp.	Rice					2
Poaceae	<i>Poecilostachys festucaceus</i>	Forest grass	3				
Strelitziaceae	<i>Ravenala madagascariensis</i>	Traveller's palm		12	6	7,13	1,2
Zingiberaceae	<i>Aframomum</i> sp.	Wild ginger			6,9,10	13	2
Dicots							
Anacardiaceae	<i>Mangifera indica</i>	Mango					2
Annonaceae	<i>Annona squamosa</i>	Custard apple			6		
Lauraceae	<i>Cryptocarya</i> sp.	Forest tree		12			
Moraceae	<i>Streblus dimepate</i>	Woody shrub	3				
Moraceae	<i>Artocarpus heterophilus</i>	Jackfruit			6,9		2
Moraceae	<i>Artocarpus altilis</i>	Breadfruit			6		
Moraceae	<i>Ficus</i> sp.	Figs					2
Rosaceae	<i>Rubus moluccanus</i>	Weedy shrub			6		
Rubiaceae	<i>Coffea</i> spp.	Coffee			6		
Sapindaceae	<i>Litchi sinensis</i>	Lychee			6,9		
14 families	29 or 30 species	Total per site :	7	7	12	6	11 or 12

Notes: 2^a: as voloajatsy; 11^b: as volotsanganana; 11^c: as volohoto



Figure 8. *Arundinaria* sp. in high elevation forest of the western Ankeniheny-Zahamena Corridor in eastern Madagascar. Photographs by Lucien Randrianarimanana.



Figure 9. *Arundinaria* sp. in high elevation forest at Tsinjoarivo in eastern Madagascar. Photographs by Laingo Rakotonirina.



Figure 10. *Arundinaria* cf. *ambohitrensis* in high elevation forest at Ranomafana National Park in south-eastern Madagascar. Photographs by Tony King.

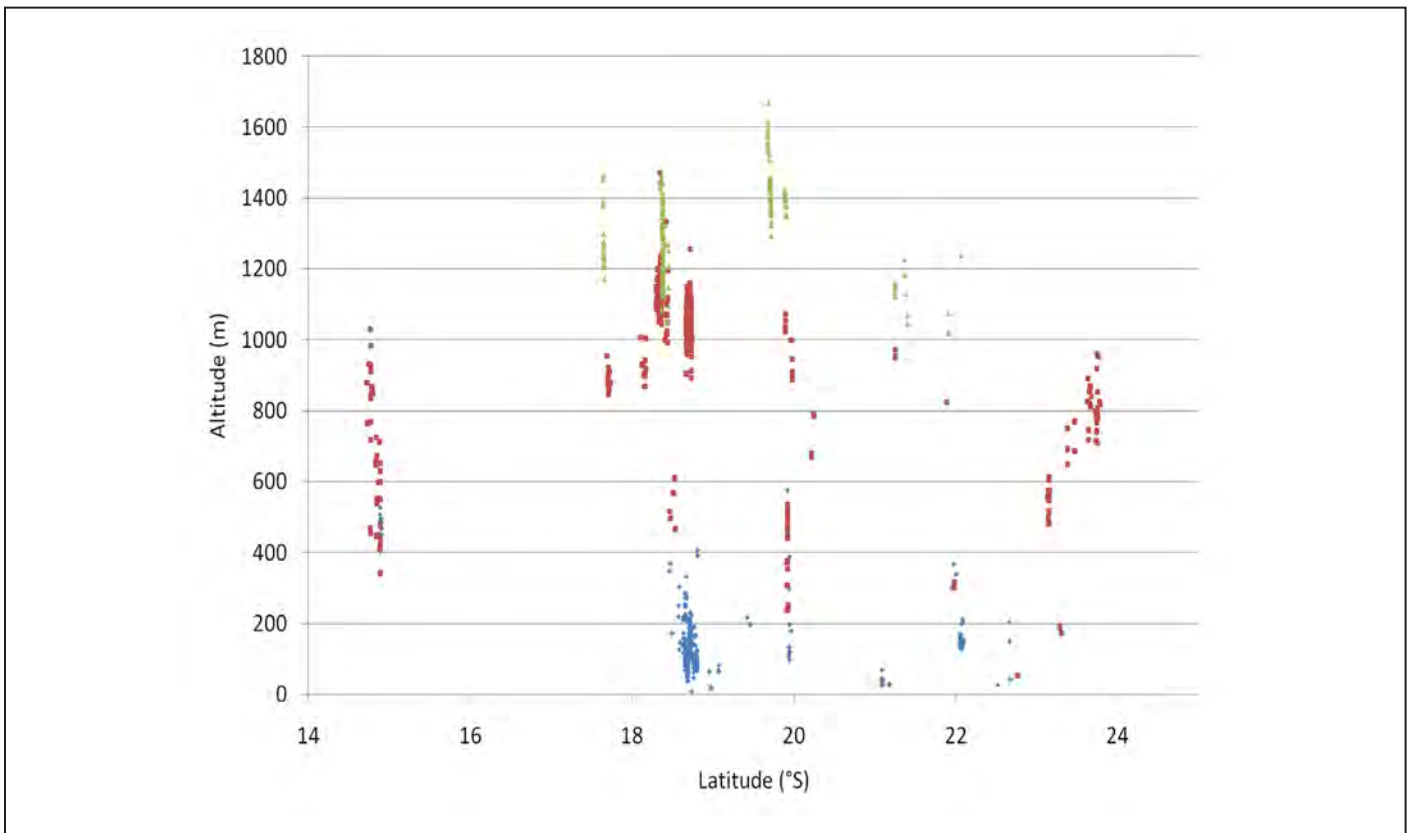


Figure 11. Scatter graph of GPS points taken for *Valiha diffusa* (blue), *Cathariostachys* spp. (red) and *Arundinaria* spp. (green), by elevation and latitude.

bamboo distributions over time may have affected changes in *P. simus* distribution.

Prolemur simus is known to have a very seasonal diet (Tan 1999), feeding primarily on young large-culmed bamboo shoots when they are available (Figs. 16–18), and switching to other bamboo parts when they are not. In Ranomafana National Park, *P. simus* feeds primarily on the young ground shoots of *Cathariostachys madagascariensis* during the early wet season months of November to February, then supplements its diet with bamboo leaves and branch shoots as the number of ground-emerging shoots declines, whilst during the drier months of June to October it feeds principally on the mature culm pith, still of *C. madagascariensis* (Tan 1999). At our study sites, the seasonality of the emergence of ground shoots of *Cathariostachys madagascariensis* was generally similar to that described by Tan (1999), as was that of *Valiha diffusa*, with the latter perhaps a month later than

C. madagascariensis. Other large-culmed bamboo species, however, did not follow the same seasonality, especially those in the genus *Arundinaria* and the yellow variety of *Bambusa vulgaris*. The presence of these bamboos may therefore influence the seasonality of *P. simus* feeding behaviour if ground shoots emerge when those of other species are not available. This is particularly apparent at the lowland Ambalafary site, where the *P. simus* group spends a lot of time feeding on the ground shoots of yellow *Bambusa vulgaris* during the dry, austral winter months from June onwards (T. H. Mihaminekena, unpubl. data; Fig. 19). Similarly, at the high-elevation Ranomainty site, ground shoots of *Arundinaria* appear to be sometimes eaten during the austral winter dry season, especially in drier years when *Cathariostachys* ground shoots are rarer (H.L.L. Randrianarimanana, unpubl. data).

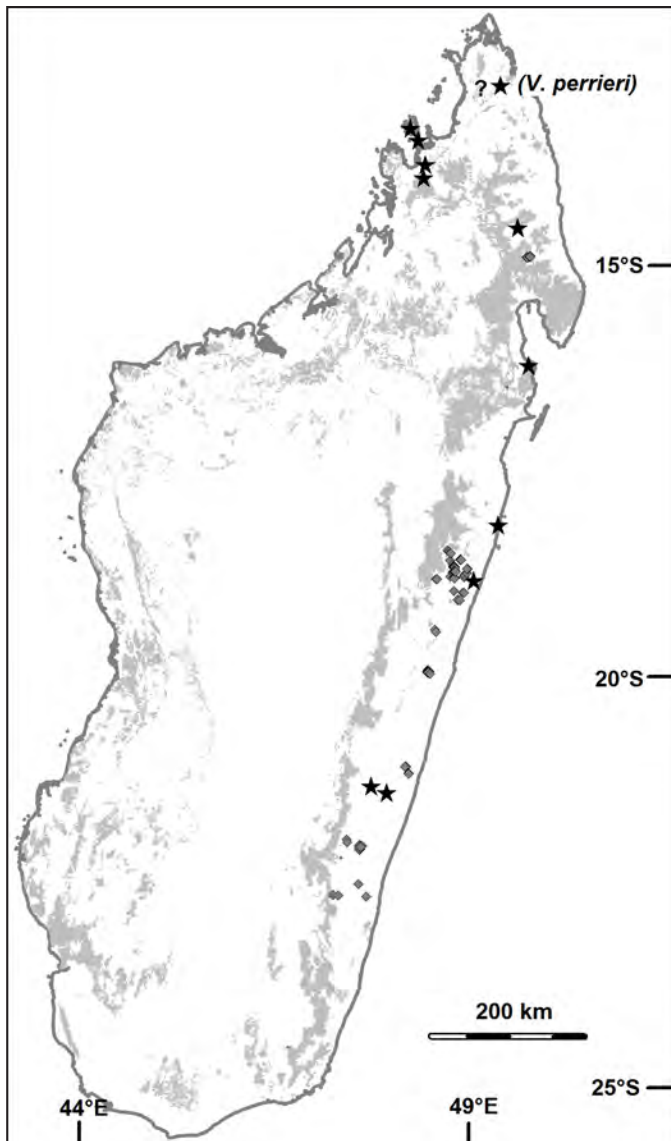


Figure 12. Distribution of herbarium specimens (black stars) and our field observations (gray diamonds) of *Valiha diffusa*, with approximate forest cover given in gray.

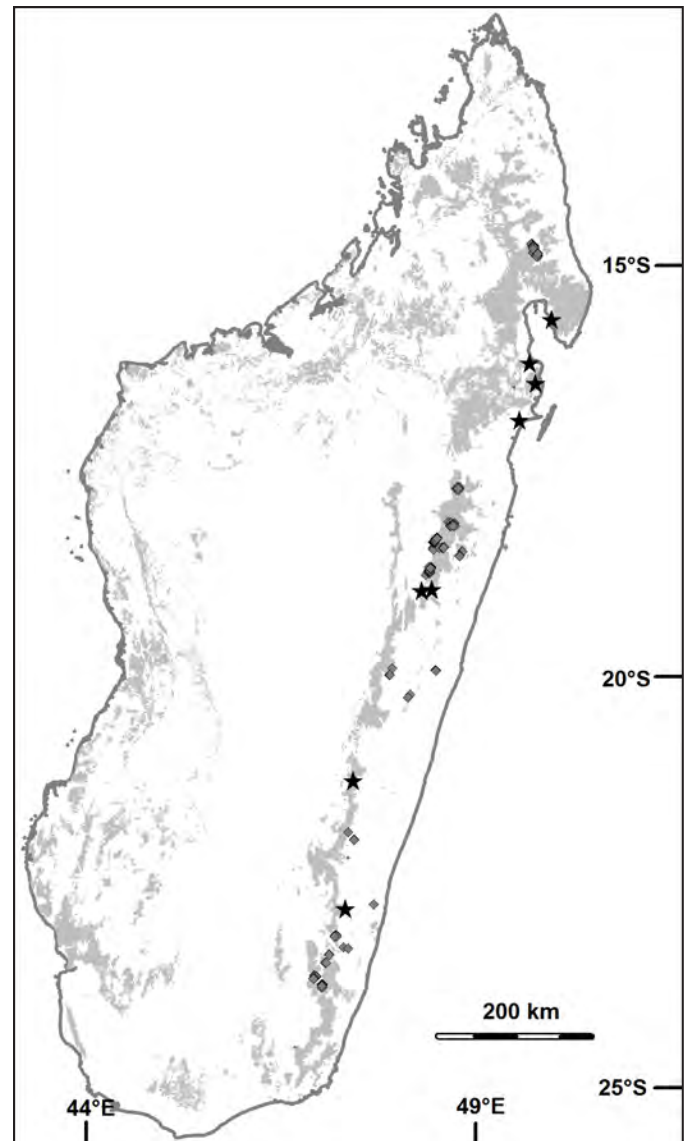


Figure 13. Distribution of herbarium specimens (black stars) and our field observations (gray diamonds) of *Cathariostachys* spp., with approximate forest cover given in gray.

Conclusions

The woody bamboos of Madagascar are in need of research and conservation activities in their own right (Dransfield 2003). A better understanding of their taxonomy, distribution and ecology will be key also to a better understanding of the factors influencing the distribution and conservation needs of bamboo lemurs in general, and of the Critically Endangered greater bamboo lemur in particular. It was this latter motivation that led us to write this paper, but in doing so we have hopefully contributed to the former. Our presentation of photos of various large-culmed bamboos in Madagascar, and preliminary distribution maps and elevation ranges, should facilitate other researchers to take on the otherwise daunting task of trying to identify bamboos, at least to genus.

This should lead to improved understanding of distribution and other aspects concerning them, and consequently to the potential distribution of associated fauna such as bamboo lemurs. The recent discoveries of previously unknown populations of the greater bamboo lemur indicate that there is still much to be learnt about the remarkable endemic biodiversity of Madagascar, not least of bamboos and bamboo lemurs.

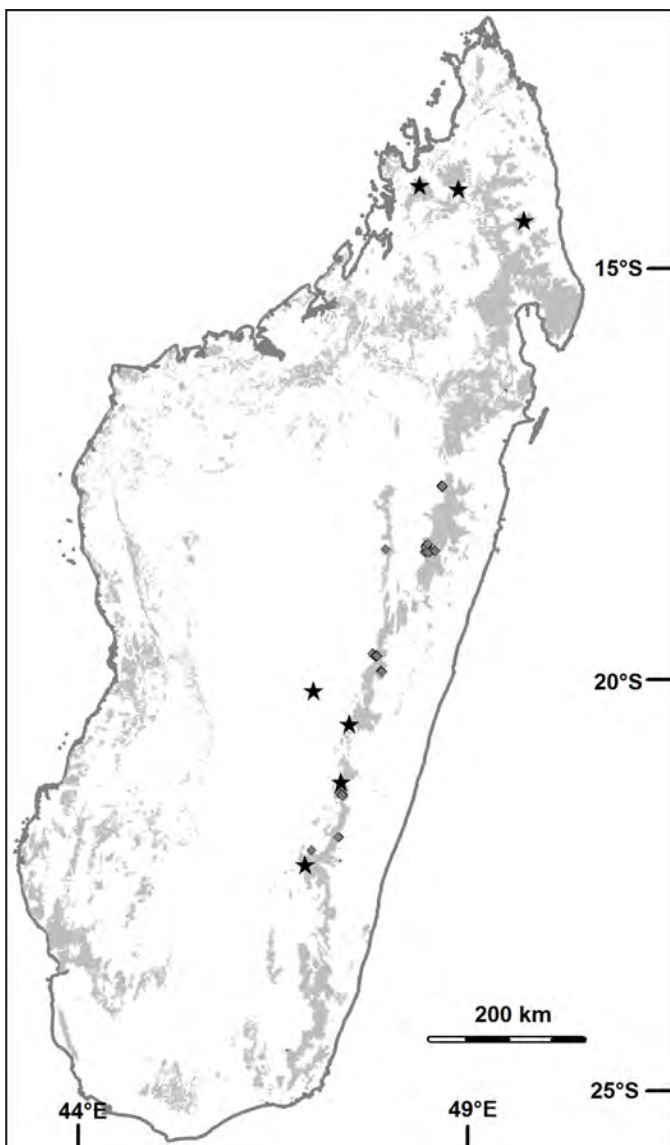


Figure 14. Distribution of herbarium specimens (black stars) and our field observations (gray diamonds) of *Arundinaria* spp., with approximate forest cover given in gray.

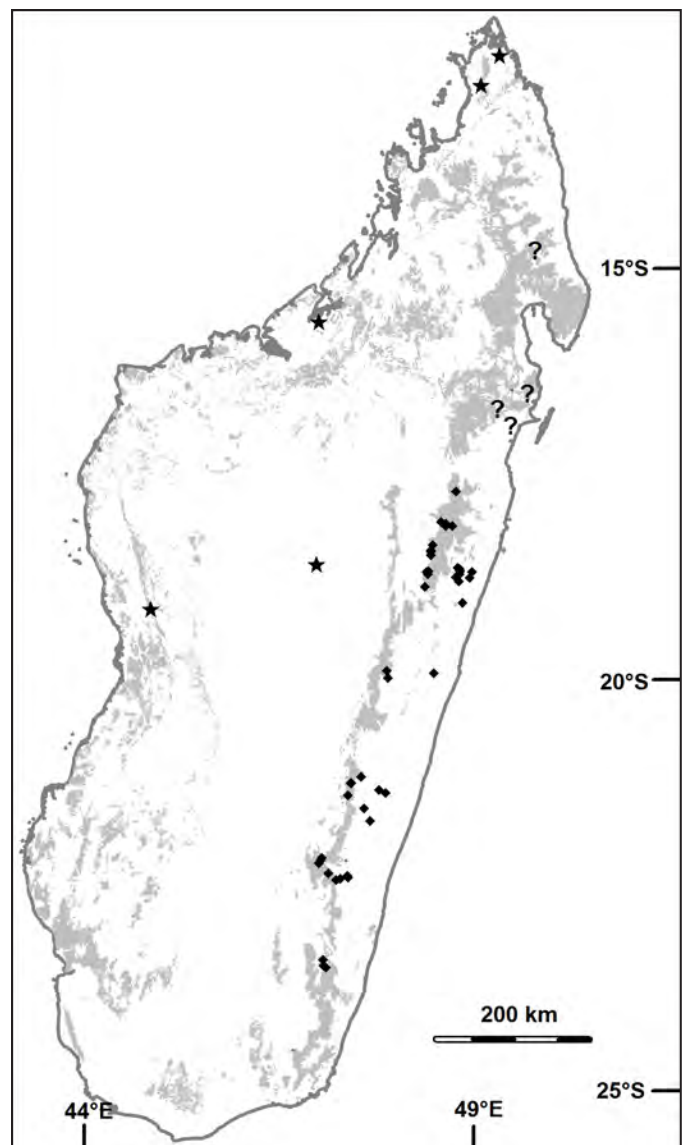


Figure 15. Distribution of sites of recent direct or indirect observations of *Plemur simus* (diamonds; from Rakotonirina *et al.* 2011), unconfirmed local knowledge suggesting possible recent presence of *P. simus* (question marks; from Dolch *et al.* 2010), sites where *P. simus* historically existed based on sub-fossil remains (stars; from Godfrey *et al.* 2004), and approximate current forest cover (gray areas).



Figure 16. *Prolemur simus* feeding on young ground shoots of *Cathariostachys madagascariensis* in Ranomafana National Park, February 2012. Photograph by Tony King.

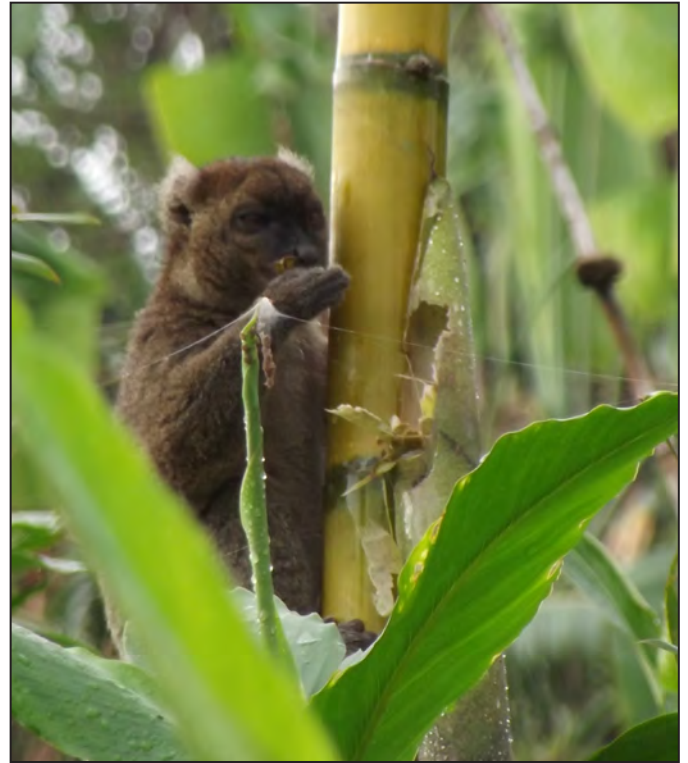


Figure 17. *Prolemur simus* feeding on a young culm shoot of *Valiha diffusa* at the Vohiposa lowland site, Brickaville District, March 2012. Photograph by Hery Randriaingy.



Figure 18. Feeding remains left by *Prolemur simus* on young ground shoots of *Valiha diffusa* surrounding Andriantantely lowland forest, Brickaville District, January 2012. Photograph by Hery Randriaingy.



Figure 19. Feeding remains left by *Prolemur simus* on young ground shoots of the yellow variety of *Bambusa vulgaris* at the Ambalafary lowland site, Brickaville District, August 2011. Photograph by Hasimija Mihaminekena.

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Guidelines for the Rehabilitation and Release of Vervet Monkeys

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Abstract: In South Africa, conflict between humans and vervet monkeys (*Chlorocebus aethiops*) has led to large numbers of the species coming into care in rehabilitation centers. An accumulation of displaced vervet monkeys in such centers has necessitated the process of troop formation and release. To date, the quality of methods used has been variable, as have the outcomes of rehabilitation and release efforts. Here we present proposed guidelines for the rehabilitation and release of vervet monkeys based on studies of vervet monkey rehabilitation, behavior and biology, in conjunction with existing guidelines for nonhuman primate reintroductions produced by the International Union for Conservation of Nature (IUCN). The aim of these guidelines is to facilitate improved rehabilitation, release, and post-release monitoring and assessment, with the ultimate goal of improving release outcomes.

Key words: *Chlorocebus aethiops*, primate, rehabilitation, reintroduction, wildlife management

Introduction

Rehabilitation involves the treatment of medical or physical disabilities, active teaching of animals to develop latent or lost skills necessary for independent survival in the wild (for example, foraging, rearing behavior and anti-predator behavior) and weaning from human contact (Masataka 1983; Cowlshaw and Dunbar 2000; Beck *et al.* 2007). There are two main types of rehabilitation projects. The first type releases animals for the purpose of conservation and in some cases is associated with zoo-based breeding projects. The second, more common form, is the rehabilitation and release of confiscated pets, orphans and animals displaced by logging, habitat destruction (Cowlshaw and Dunbar 2000) and other human influences. These welfare-based release projects address issues of illegal trade and welfare in captivity by rescuing, rehabilitating and releasing confiscated illegal pets and orphans (Cheyne 2006).

Although the International Union for Conservation of Nature (IUCN) has created rigorous, well researched guidelines for primates, they focus on conservation rather than welfare, and are generalized to either all primates (Baker 2002), or great apes (Beck *et al.* 2007). We have developed a flow chart for primate rehabilitation (Guy *et al.* in press) and a decision tree for rehabilitation of mammals in general (Guy *et al.* 2013). With the exception of gibbons (Cheyne *et al.* 2012), however, there seems to be a general lack of species-specific guidelines for primate rehabilitation.

In South Africa, vervet monkeys are frequently kept as pets or otherwise treated as vermin (Grobler *et al.* 2006). Both situations often result in vervet monkeys coming into care in rehabilitation centers due to problems such as unwanted or confiscated pets, shooting, poisoning, attacks by domestic pets, and motor vehicle accidents (Smit 2010). These problems are considerable; over 300 individuals were rescued in KwaZulu-Natal, South Africa, by the “Monkey Helpline” organization in the first six months of 2010 (Smit 2010). Many of them ended up in the care of one of 20 registered primate rehabilitation centers in South Africa (Wimberger *et al.* 2010a). Displaced monkeys accumulate in rehabilitation centers as a result of high-quality care provided by specialized orphan-care centers (for example, The Hamptons Wildlife Rescue Centre, pers. obs.) along with a reluctance to euthanize displaced primates except where severe injury or disease is involved (Guy *et al.* 2013).

As a result of these issues, a number of vervet monkey rehabilitation centers have begun building and releasing troops back to the wild in an attempt to improve their welfare. Some have been relatively successful (for example, Wimberger *et al.* 2010b; Guy *et al.* 2011; Guy 2013), while others have experienced considerable problems, including poor release-outcomes as a result of inappropriate rehabilitation and release systems, poor choice of release sites, and inadequate post-release monitoring (Wimberger *et al.* 2010b; Guy *et al.* 2012a).

Vervet monkeys (*Chlorocebus aethiops*) are classified as 'Least Concern' on the IUCN Red List (IUCN 2012) and, as such, their rehabilitation and release is motivated by welfare considerations rather than conservation. Although they have different primary objectives, both conservation and welfare re-introductions should adhere to the best practice guidelines set out by the IUCN (Beck *et al.* 2007). Beck *et al.* (2007) also recommended that taxon specific protocols be developed with customized guidelines.

Our aim here is to present guidelines for the rehabilitation and release of vervet monkeys, based on studies of vervet monkey rehabilitation, behavior and biology, and incorporating existing guidelines for nonhuman primate reintroductions (Baker 2002). Our primary goal is to improve rehabilitation methods in order to maximize opportunities for positive outcomes in future releases.

Recommendations for a Best-Practice Model

Arrival

A history should be collected for each animal upon arrival. This should include information such as the reason for the monkey coming into care, the circumstances and location where found, its approximate age, and, if a pet, how long it was kept for and how and what it was fed. This information can help to determine the kind of initial care needed and whether or not the individual is likely to be suitable for release. For example, an orphan that has been cared for by an inexperienced person may be malnourished and/or dehydrated, a monkey that has been hit by a car may be stressed and injured, while an adult vervet that has been kept as a pet for an extended period of time will be habituated to humans and is unlikely to be suitable for release (see, for example, Guy *et al.* 2011).

Following initial assessment, the monkey should be quarantined and undergo medical assessment and treatment. A minimum quarantine of 31 days is recommended by the IUCN, with 90 days being ideal (Baker 2002). This ensures that each monkey is healthy and helps to avoid transmission of any diseases to other animals at the center. Although there are costs involved, this is a vital step. Like many primates, vervet monkeys are susceptible to human diseases and so should be examined by a veterinarian after transfer from a carer or another center. Some infectious diseases known to affect South African vervets include foamy viruses, tick-bite fever (*Rickettsia conori*) and chikungunya virus (Kaschula 1978). The new arrival should also undergo a behavioral assessment. Many confiscated pets will have been socially isolated when very young, and this can have profound effects on their behavior (Suomi *et al.* 1976; Mason *et al.* 2007) and their ability to be socially integrated.

Conspecific re-socialization

Once an initial assessment has been made and the vervet monkey is deemed healthy, they may then move into the group formation stage of rehabilitation. Due to the staggered

arrival of new animals and the existence of established groupings at the center, a social group is rarely formed from scratch. An individual is likely to be introduced to an existing group or to just a few other animals to start a new group. Introductions are best made gradually, particularly in the case of young orphaned primates that have been hand-raised and have had limited or no previous contact with conspecifics.

The use of adjacent enclosures—the main one housing an existing group, the smaller one acting as an introduction enclosure for the new arrival—has been successful (Du Toit 2009). In addition, the Vervet Monkey Foundation (Du Toit 2009) and the Wild Animal Trauma Centre and Haven (WATCH) have made use of safe areas at the edge of enclosures for small, milk/formula-dependent orphans to allow feeding without interference from larger monkeys in a group. Adult females in existing groups are sometimes able to foster young orphans, facilitating their introduction to the group (Du Toit 2009). Anecdotal evidence suggests that this is particularly beneficial for adult females that have recently lost their infant (Bruce and Sandi Cronk, pers. comm.).

A number of things need to be considered when forming groups for release. The geographic origin of the monkey must be determined so as to avoid mixing distinct populations. Individuals of unknown geographic origin should not be considered for release (Baker 2002). No significant genetic differences between populations of South African vervet monkeys have been identified, however (Grobler and Matlala 2002), and this means that mixing vervets from different parts of South Africa is unlikely to pose problems for local populations. Government restrictions that prevent the movement of animals between provinces, however, may mean that they will be managed separately anyway (Ezemvelo KZN Wildlife 2008). Moreover, published genetic studies for vervet monkeys are still few, and future work might identify geographic patterning or the presence of subpopulations not currently recognized, as has occurred recently with chacma baboons (for example, Sithaldeen *et al.* 2009).

Dominance relationships need to be monitored, as a lack of a clear dominance hierarchy can lead to heightened aggression in groups (Guy and Curnoe 2011). Animal compatibility can also be a problem, as has been noted for lar gibbons (*Hylobates lar*) and capuchin monkeys (*Sapajus apella*) (de Veer and van den Bos 2000; Suárez *et al.* 2001). Although aggression during the introduction of adult or older juveniles to an existing group is relatively common (A. J. Guy, pers. obs.), prolonged physical aggression may indicate a need to rearrange groups.

Group sizes, and age and sex ratios should be as similar to wild groups as possible (Baker 2002) in order to maximize survival and improve the chances of forming a cohesive group that will remain together after release. The average wild group size for vervet monkeys is 25, with a range of 11–47 (Willems and Hill 2009). Groups of between 24 and 31 are recommended, based on group sizes in successful releases as assessed in Guy *et al.* (unpublished). Ideally, studies of wild groups close to the desired release site would be

undertaken to indicate the natural group size for that area as this is influenced by a number of factors such as the size of available territories (Jackson and Gartlan 1965), food quality and quantity, environmental temperature, and predation risk (Cowlshaw and Dunbar 2000). Detailed published wild studies of vervet monkeys largely focus on Kenya, and the average group composition in the Amboseli reserve is *c.*11% adult males, *c.*23% adult females, *c.*8% sub-adult males, *c.*4% sub-adult females, *c.*49% juveniles and *c.*6% infants (Struhsaker 1967b). As full group composition data are only available for two wild groups studied in South Africa (Cambefort 1981; Willems 2007), we recommend the following approximate proportions: *c.*10% adult males and *c.*25% adult females, providing an adult sex ratio of 1:2.5, close to 10% sub-adult males, *c.*5% sub-adult females, *c.*50% juveniles and *c.*5% infants. This may be altered if reliable studies of wild groups are carried out in the region of the planned release providing composition data for specific regions. Published studies from South Africa yield an average adult sex ratio of approximately 1:2 (males:females), ranging from 1:0.9 to 1:4 ($n = 5$, Table 1).

Housing, training, and preparation

Social groups should be housed in naturalistic enclosures to promote shelter seeking (some form of shelter should be provided, see, for example, Fig. 1) and moving on natural

Table 1. Adult sex ratios of wild troops of vervet monkeys (*Chlorocebus aethiops*) studied in South Africa.

Number of adult males	Number of adult females	Approximate ratio	Reference
5	8	1:1.6	Barrett (2005)
2	8	1:4	Willems (2007)
7	6	1:0.9	Cambefort (1981)
9	14	1:1.5	Harrison and Byrne (2000)
4	8	1:2	Baldellou and Adan (1997)



Figure 1. A naturalistic enclosure at the Wild Animal Trauma Centre and Haven in Vryheid, KwaZulu-Natal, South Africa. Note the vervet monkeys using the shelter provided on the left and enrichment items including trees, tyres, poles and ropes. Photo by A. J. Guy.

substrates through complex environments. The typical habitat of vervet monkeys is gallery forest (Tappen 1960), so provision of trees or large branches in the enclosure would assist in developing natural climbing behavior. Objects such as ropes, tyres and smaller branches can be used to add complexity and allow for regular changes to be made to the enclosure (see Fig. 1).

Vervet monkeys forage on the ground and in trees (Tappen 1960). To encourage development of foraging behavior, particularly for younger monkeys or ex-pets, natural foods should be provided in a way that encourages both behaviors. Natural foods help the animals to learn what is palatable and safe to eat in their release habitat. Vervets have a diet that includes fruit, flowers, seeds, grass, leaves, bark, resin, gum, bulbs, roots, insects, eggs and small mammals (Whittsit 1997). Food items taken from the intended release site are desirable. For example, the red currant (*Rhus chirindensis*) and strangler fig (*Ficus* sp.) are common in some parts of Kwa-Zulu Natal in South Africa (Department of Water Affairs and Forestry 1992) and are eaten by vervet monkeys (Harrison 1984; Venter and Venter 2005). If it is not feasible to do this long term, a balanced diet of fresh fruit and vegetables with added protein (for example, nuts and seeds) is recommended (as employed at WATCH: Bruce and Sandi Cronk, pers. comm.), with natural foodstuffs such as fruit, leaves and insects from the release site introduced in the months leading up to release.

Environmental (for example, altering enclosure to increase complexity as described above) and feeding (for example, food puzzles) enrichment can be used to assist the development of natural behaviors. Fruit can be attached to poles, for example, and invertebrates can be placed in boxes of organic material to encourage natural foraging behaviour (Suárez *et al.* 2001). Vervet monkeys tend to have preferred sleeping trees (Struhsaker 1967a), so suitable sleeping sites such as trees, branches, hanging baskets or boxes (Fig. 2) and nets should be provided. Loose objects such as cardboard boxes, balls and paper can be used to encourage exploration of the environment (Cheyne *et al.* 2012).

Minimizing contact with humans is also important so as to avoid excessive habituation that may lead some released animals to seek out human company post-release (see, for example, Guy *et al.* 2011). We recommend methods similar to those used in the rehabilitation of brown capuchin monkeys (*Sapajus apella*) (Suárez *et al.* 2001). To avoid association of humans and food, visual isolation was achieved by covering the mesh of the enclosure with synthetic material, with food provided through a window (Suárez *et al.* 2001). We recommend that just one or two walls be covered for this purpose rather than the entire enclosure, as complete visual isolation may reduce effective acclimatization to the area. Feeding times should be varied so as to reduce predictability (Suárez *et al.* 2001).

Training for predator awareness is an important part of the rehabilitation process (Griffin *et al.* 2000). Animals that have lived in captivity from a young age are likely to have been isolated from predators and may not express normal



Figure 2. Hanging sleeping baskets (circled, upper right of photograph) at “Vervet Haven” in Durban, South Africa. Photo by A. J. Guy.

predator-avoidance behavior. This can greatly reduce their chances of survival when released. Predator-avoidance behavior functions to reduce the risk of an animal or its relatives being killed. Vervet monkeys have a number of predator alarm calls and associated responses including: running up into the trees in response to a leopard alarm call, looking up and running into the bushes for eagle alarm calls, and standing and looking down for snake calls (Cheney and Seyfarth 1981). These behaviors need to be functional the first time that a predator is encountered if an animal is to survive, but responses will also improve with group experience (Griffin *et al.* 2000). Housing vervets in an area where they will come into contact with predators such as snakes and eagles can increase their pre-release awareness of them (see, for example, Guy *et al.* 2011, 2012b). Providing thick bushes or artificial hiding places may also act to promote natural predator avoidance behavior for birds of prey.

In addition, studies of other mammals have shown that the presence of wild-caught experienced individuals in a group can improve responses to predators (Shier and Owings 2007). If the location of the rehabilitation center prevents pre-release exposure to predators, some form of training may be required. This can include intentional direct exposure to the predators themselves (Shier and Owings 2007), or use of substitutes such as related species (McLean *et al.* 2000),

silhouettes (Brown *et al.* 1992) or stuffed or model predators (Arnold *et al.* 2008). In cases where monkeys have not been exposed to predator species or had opportunity to observe or demonstrate appropriate anti-predator behavior, an extended period of acclimatization at the release site is recommended to increase the chances of the group encountering predators from a protected environment (within an enclosure) prior to release. In this case, it would be necessary to provide an appropriately large enclosure so as to limit stress and to avoid increased aggression that sometimes occurs when vervets are moved to a smaller enclosure (Clarke and Mayeaux 1992).

Pre-release assessment

Further assessment should be carried out prior to release. An important part of this is a pre-release medical check by a qualified veterinarian. Animals kept in captivity are susceptible to infection with parasites that may be foreign to the species (Cunningham 1996). In addition, many primates, including vervet monkeys, are susceptible to human diseases. As such, human contact should be minimized and any animals for release need to be screened for common diseases. It is vital that animals for release be healthy. Failing to carry out pre-release medical checks can result in disease transmission to wild populations, both conspecifics and other species (Viggers *et al.* 1993), mortality of released animals as a result of pathogens and ectoparasites causing disease triggered by the stress of transport and release, and zoonotic diseases (for example, tuberculosis, influenza, viral hepatitis, and measles [Heuschele 1991]) being transmitted to the local human population (Viggers *et al.* 1993).

Methods used to reduce these risks include: quarantine (as described in the ‘arrival’ section), clinical examination by a veterinarian (Viggers *et al.* 1993), fecal examination for parasite eggs and larvae (services may be provided by veterinary laboratories) (Viggers *et al.* 1993), hematology and serum biochemistry profiles to aid disease detection based on comparison with normal blood values for the species, serological testing to detect infectious diseases (Viggers *et al.* 1993), microbial culture to isolate and identify the cause of the disease, and vaccination for some common diseases where there is a specific risk (Viggers *et al.* 1993). Animals that die in captivity should undergo a full post-mortem examination to determine the cause of death. Wherever possible, this should also be done for animals that die post-release so as to identify any problems with the health of rehabilitated animals.

Primates can carry tuberculosis, which can be transmitted to humans and domestic animals (Viggers *et al.* 1993). A common way to test for tuberculosis is the skin-testing procedure involving an intradermal injection, with a positive reaction being detected after three days (Viggers *et al.* 1993). This should be carried out by a qualified veterinarian.

Behavioral assessment should be carried out to confirm suitability for release. Direct comparisons of the behavior of wild and captive animals can help to determine which animals are ready for release. Individuals that show significant deviation from wild behavioral patterns are less likely to survive

post-release, while slight deviations are most likely due to captive conditions and are not a huge concern (Mathews *et al.* 2005). Monkeys ready for release should not seek human contact. Stereotypic behaviors may develop during captivity and animals should be monitored in order to identify any abnormal behavior that may affect their suitability for release.

Release site selection

One of the most important steps in a release project is the selection and assessment of the release site. This can be quite complex and a number of factors that should be assessed for any primate release have been outlined in the IUCN guidelines for non-human primate re-introductions (Baker 2002). These include availability of food and water and their seasonality, protection of the site, if conspecifics are absent and the reason for their absence, and human influence. A release site should provide appropriate habitat and ideally be free from human disturbance (for example, hunting and introduced, feral or domestic animals) (Dodd and Seigel 1991). Habitats should be chosen to minimize edge effects and should not have an unusually high concentration of predators (Dodd and Seigel 1991).

Vervet monkeys rely on riparian habitat with permanent water sources. A lack of available water increases mortality (Dunbar 1988), so the dry season should be avoided for releases, and at least one permanent water source should be located close to the release site. Release should be timed to coincide with the wet season when availability of resources would be highest (Wimberger *et al.* 2010b; Guy *et al.* 2012b). Vervets adapt well to human disturbance and are prone to raiding crops (Saj *et al.* 2001). This is a significant cause of human-monkey conflict, and often results in trapping and killing of vervets (Saj *et al.* 2001). Furthermore, hunting has been identified as a considerable cause of mortality in released vervet monkey groups (Guy *et al.* 2011, 2012b; Guy 2013). Strategies to combat this include release into areas as isolated from human settlement as possible, release habitats that contain numerous fruiting trees, and the provision of supplementary food (use of soft release procedures) for a period after release until the vervets can effectively feed themselves.

Habitat features that appear to be related to successful releases of vervet monkeys include high altitude and rainfall, a high proportion of tree or forest cover, and a minimal or zero percentage of land cover in the area being composed of dams, bare sand, grassland/bush clumps mix and bushland (Guy *et al.*, unpublished). Privately owned land with supportive landholders has also been associated with positive release outcomes (Guy *et al.*, unpublished), so agreement and co-operation of landholders in the desired release area must be obtained before release. Human disturbance of land, urban settlement, and other human activities are associated with poor release-outcomes (Guy *et al.*, unpublished) and so should be avoided.

The size of the release site is important, particularly with regard to assessing impacts on the released group from humans. The average home range for wild vervet monkeys is

0.725 km², with an average day range length of about 1 km (Willems and Hill 2009). However, released vervet monkeys have been known to have home ranges of up to 6.98 km² (Guy *et al.* 2011), regularly travelling more than 2 km from the release site (Guy *et al.* 2011). Thus our recommendation is to select a release area that can allow for the expansion of the groups' range up to 7 km², with little or no human activity within a 3 km radius of the release site.

It is important for the density of conspecifics to be lower than the carrying capacity (Cowlshaw and Dunbar 2000). Often the presence of the species in an area is used as a definitive criterion for the suitability of a release site. This simplified view fails to recognize that introducing more vervet monkey groups to an area may lead to increased competition, aggression and disease transmission. Vervet monkeys can be highly territorial and this can result in inter-group aggression, sometimes resulting in the injury of released monkeys (Wimberger *et al.* 2010b). Whether conspecifics should be present near the release site at all depends on the specific goal of the release. If the primary aim is to improve welfare of individuals through release, then it is not necessary for conspecifics to be present. If the aim is to increase numbers, however, then wild groups must be nearby as vervet populations rely on male dispersal for gene flow (Cowlshaw and Dunbar 2000). Disease risks and competitive impacts on other resident primate species must also be taken into account. Vervet monkeys share their range with a variety of primate species, ranging in IUCN Red List status (IUCN 2012) from Least concern (for example, the southern lesser galago *Galago moholi*, chacma baboon *Papio ursinus*, blue monkey *Cercopithecus mitis*, and guereza *Colobus guereza*) to Endangered (for example, Udzungwa red colobus *Procolobus gordonorum*, and chimpanzee *Pan troglodytes*).

Release

As recommended by the IUCN, a transport plan should be in place, with emphasis on minimizing stress and risk of injury (Baker 2002). All monkeys should be transported in well-ventilated, secure transport boxes (Bruce and Sandi Cronk [WATCH], pers. comm.). The hottest part of the day should be avoided, transporting either in the early morning or late afternoon. Larger monkeys, such as adult males should be placed in individual boxes to avoid injury to smaller individuals. Vulnerable individuals such as mothers with infants should, likewise, be isolated (WATCH, pers. comm.). Whenever possible, monkeys that have been known to frequently direct aggression toward each other should not be transported in the same box. If transporting a long distance, regular stops should be made to check the monkeys and to provide food and water. The group should be accompanied by experienced personnel from the rehabilitation center and, if possible, a veterinarian (Baker 2002).

A soft-release method, including an acclimatization period at the release site, is recommended to allow the animals to recover from transport, acclimatize to environmental conditions, and become familiar with their surroundings



Figure 3. Small food items such as nuts can assist monitoring. Photo by A. J. Guy.

(Baker 2002). This site is often used as the initial location for supplementary feeding post-release (Guy *et al.* 2011, 2012a, 2012b). During acclimatization, vervets should be fed natural food, preferably taken from the surrounding area. The average time spent in the release enclosure in past vervet releases has been three days (based on Wimberger *et al.* 2010b; Guy *et al.* 2011, 2012a, 2012b). A minimum period of two weeks, as recommended for gibbons (*Hylobates* sp.) (Cheyne *et al.* 2012), however, may be more appropriate. Some experimentation may be required to determine the ideal acclimatization period and enclosure size.

Supplementary feeding should be provided to ensure that the released monkeys obtain adequate nutrition whilst acclimatizing to and exploring their new habitat. The average time period of supplementary feeding for vervets is approximately two months (Wimberger *et al.* 2010b; Guy *et al.* 2011, 2012a), but this may vary due to circumstances such as wild groups eating the supplied food (Guy *et al.* 2012a) or the monkeys leaving most of the supplied food and obtaining nutrition from wild sources. The amount of food may be reduced as the released monkeys increase their foraging on natural sources. Once supplementary feeding ceases, small food items such as nuts can be useful in drawing out monkeys so that their condition can be assessed (Fig. 3).

Post-release assessment

Post-release monitoring is vital, and the data collected during this time allow the success of the project to be assessed and lessons to be learned for future release efforts. The released groups should be monitored for a minimum of one year (Beck *et al.* 2007) to encompass at least one breeding season and all climatic conditions at the release site. Most released vervet monkey groups have not been monitored for this minimum period (Wimberger *et al.* 2010b; Guy *et al.* 2012a).

Tracking devices such as radio- or GPS-collars are vital to monitoring. Studies that have not used such devices have



Figure 4. A released vervet monkey fitted with a radio-collar. Photo by A. J. Guy.

yielded very little data extending over a time frame of less than six months (Guy *et al.* 2012a). Radio-collars (Fig. 4) have been used for several releases of vervet monkeys (Wimberger *et al.* 2010b; Guy *et al.* 2011, 2012b) and at times were the only way of gathering data on the groups in inaccessible terrain (Guy *et al.* 2012b). However, data collection is also limited when collars are fitted to only a small portion (mean = 44%: Guy *et al.*, unpublished) of the group. All individuals large enough to carry a monitoring device should be fitted with one so as to limit the number of missing animals and allow monitoring of all post-release. Specialized GPS collars as used, for example, by the University of Neuchâtel for wild studies (Bruce Cronk, pers. comm.) are recommended whenever finances permit as they allow remote data collection when animals cannot be physically located. There are some published primate release studies that have made use of GPS collars (Markham and Altman 2008; Pebsworth *et al.* 2012), but the collars used exceed the maximum 5% of body mass (Animal Care and Use Committee 1998) for vervet monkeys (based on an average mass of 3 kg for females and 4.3 kg for males; Dunbar and Barrett 2000).

Monitoring data should include predator densities, competitors, human impacts and population demographics (births/deaths) (Cheyne 2006). Predators of vervet monkeys include: baboons (*Papio ursinus*, *P. cynocephalus*), Verreaux's eagle owl (*Bubo lacteus*), Martial eagles (*Polemaetus Bellicosus*), Crowned eagles (*Stephanoaetus coronatus*), and leopards (*Panthera pardus*) (Struhsaker 1967a). Poisonous snakes such as cobras (*Naja haje*) and puff adders (*Bitis arietans*) have the potential to kill vervet monkeys. These species evoke an alarm response from vervet monkeys, indicating that they recognize them as a threat (Struhsaker 1967a). Other carnivorous felines are potential predators, with high intensity predator responses observed for lions (*Panthera leo*) and servals (*Felis serval*) (Struhsaker 1967a). Human activities are possibly the most significant cause of mortality and care should be taken to avoid areas of human activity when selecting a release site.

Other key parameters for assessment could include animal numbers over time, sex ratios, and adult/juvenile ratios (which can be compared to existing wild populations), population changes, and continuing assessment of threatening processes (Fischer and Lindenmayer 2000).

Conclusion

Although rehabilitation and re-introduction programs have been in progress for many years, until very recently, vervet monkey rehabilitation was almost entirely lacking in the published literature. Even with recent studies, the number of releases that have been assessed thus far are relatively few. We hope that these guidelines will be the beginning of a continually developing manual for vervet monkey rehabilitation. Additional material should be incorporated based on lessons learned in future releases in order to maximize positive release outcomes.

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Status of Zanzibar Red Colobus and Sykes's Monkeys in Two Coastal Forests in 2005

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Abstract: We censused two discrete subpopulations of the endangered Zanzibar red colobus (*Procolobus kirkii*) and sympatric Sykes's monkeys (*Cercopithecus mitis albogularis*) between February 2004 and September 2005 in two coastal forests in Zanzibar, including the northernmost (Kiwengwa-Pongwe Forest) and some of the southernmost (Uzi and Vundwe Islands) extents of *P. kirkii*'s range. Surveys totalled 365 hours and 307.8 km along >16 km of line transects; 472 sightings were made (Red colobus $n = 252$; Sykes's monkey $n = 220$). The southern forests (coral rag with adjacent mangrove) of Uzi and Vundwe Islands were found to support *P. kirkii* at a higher estimated density (29.6 groups/km²) relative to the northern coral rag forest (lacking mangrove) in Kiwengwa-Pongwe (7.5 groups/km²). However, for red colobus in coral rag, up to *c.*40% of these "groups" could be "sub-groups" based on our observations of consistent fission-fusion and small groups seen during the census ($6.63 \pm 0.4SE$ in Kiwengwa, $6.64 \pm 0.33SE$ in Uzi) and behavioral follows in this habitat type. We therefore also include density estimates expressed as individuals/km² (49.72 individuals/km² in Kiwengwa, 196.32 individuals/km² in Uzi). On Uzi and Vundwe Islands, colobus densities were higher than those of Sykes's monkey (Sykes's monkey density = 18.9 groups/km²), while we found no difference between the density of these two taxa in Kiwengwa (Sykes's monkey density = 8.1 groups/km²) where encounters with humans were more frequent, vegetation was more disturbed, and the two species often associated. Although these populations represented a fraction of the historical total population of red colobus and Sykes's monkeys on Zanzibar, their estimated abundance was significant, and their marginal habitat and unprotected status were important in the general context of primate conservation in unprotected and fragmented landscapes.

Key Words: Coastal forest, mangrove, *Procolobus*, Sykes's monkey, Tanzania, island endemic

Introduction

Approximately 40% of *Procolobus* taxa are threatened with extinction (Struhsaker 2005). Of the five recognized species (Grubb *et al.* 2003; Grubb 2006), *P. pennantii*, *P. gordonorum*, and *P. kirkii* are Endangered (IUCN, 1990), and *P. rufomitratu*s and *P. badius waldroni* are Critically Endangered (Oates *et al.* 2000; McGraw 2005; Roberts and Kitchener 2006). The most geographically isolated of these is the Zanzibar colobus *Procolobus kirkii* (Gray, 1868), known primarily from a population of about 1,500 in the largest protected area on Zanzibar, the Jozani-Chwaka Bay National Park (JCBNP) (Mturi 1991, 1993; Siex 2003). An island-wide presence/absence survey conducted between 1977 and 1981 found colobus groups living in 13 non-adjacent forest patches and estimated the species' Zanzibar-wide population

at 1,700 individuals (Silkuluwasha 1981). Struhsaker and Siex (1994, 1996, 1998) made comparable estimates of fewer than 2,000 individuals more than a decade later. No population estimates are available for Sykes's monkey on Zanzibar.

Procolobus kirkii's resilience to habitat alteration is evident in its successful translocation to Masingini Forest Reserve, half of which is an exotic plantation (Struhsaker and Siex 1998), and its introduction to a novel but protected area (Ngezi Forest) on Pemba Island to which it is not native (Camperio-Ciani *et al.* 2001). While the species' colonizing ability may be high, competition with guenons, persecution by humans, and the high cost of translocation limit the options for such management strategies. The protection of populations where they naturally occur is preferable (Struhsaker and Siex 1998). The species' endangered and endemic status makes knowing their historical ranges and how many

are left vital to their conservation and management of forest habitats. The highest density of red colobus (550 individuals/km²) was recorded in farms adjoining JCBNP (Struhsaker and Siex 1998; Siex and Struhsaker 1999), but no other systematic surveys outside this region had been conducted at the time of our study.

This census, carried out between 2004 and 2005, and followed up with visits, most recently in 2011, to confirm presence and persistence, aimed to describe subpopulations of *P. kirkii* to the north and south of the JCBNP: the northern Kiwengwa-Pongwe forest, and Uzi and Vundwe Islands in the south. Both areas supported indigenous coral rag and mangrove forest, and had a high potential value for sustaining red colobus populations on Zanzibar. Sympatric Sykes's monkeys (*Cercopithecus mitis albogularis*) were also censused to compare the habitat-use patterns of these two genera on Zanzibar and to understand responses to habitat degradation of related species and subspecies on the mainland (e.g., Udzungwa red colobus, *Procolobus gordonorum*; Struhsaker *et al.* [2004]).

Study Sites

The Kiwengwa-Pongwe forest (5°55'–6°04'S and 39°20'–39°23'E), now a Forest Reserve, and Uzi and Vundwe Islands (6°18'–6°24'S and 39°23'–39°26'E), which remain unprotected, are located on and adjacent, to Unguja, the largest island in the Zanzibar archipelago, which lies between 4°50'–6°30'S and 39°10'–39°90'E and makes up 63% of the total area (2,332 km²) of the archipelago. The topography of Unguja is predominantly flat with a highest point of 120 m above sea level. The eastern region has shallow, arid, coral-derived, high-calcium soils supporting low-canopy, semi-deciduous dry forest known as coral rag (Masoud *et al.* 2003).

Kiwengwa-Pongwe (hereafter “Kiwengwa”) was a 33 km² forest composed of coral rag with a wide zone of transition to shrubland, scrub, and cultivation. Along the eastern edge, native forest had been cleared for plantations of *Casuarina equisetifolia*. Coral rag in Kiwengwa once supported canopy emergent trees such as *Terminalia boivinni*, *Mystrolyon aethiopicum*, *Mimusops fruticosa*, *Diospyros consolatiae*, and *Ficus* spp. and an understory dominated by cycads (*Encephalartos hildebrandtii*), all of which were red colobus food plants (Nowak 2007; Nowak and Lee 2011a).

Uzi and Vundwe islands (hereafter “Uzi”) are relatively isolated, connected to Unguja by a 3-km long, tidal isthmus of mangrove swamp in the north and surrounded by the Indian Ocean. Uzi has an area of 15.6 km², consisting of a matrix of cultivation, secondary coral rag forest, and scrub (Silkyluasha 1981). There continues to be extensive clearing of coral rag forest on Uzi, and also, recently on Vundwe Island (Nowak *et al.* 2009). Remnant patches of high coral rag could still be found along Uzi Island's western side (known as “Mchanganle”) at the time of this census (2004–2005); however, this forest was completely cleared between 2006 and 2009 (Mease 2009; Nowak *et al.* 2009). Mature mangrove stands still occur

in the north and northwest, south and southeast. The northern Uzi mangroves border JCBNP, span c.592 ha, and were gazetted as part of a 1959 Forest Reserve Decree. They are interspersed with small coral rock island “stopovers”, which once facilitated animal dispersal between Uzi and Jozani.

The southern area of Uzi was recommended for national park status by Struhsaker and Leland in 1980 as it provided the Zanzibar colobus with habitat types rare in Jozani. The southern extent of mangroves, dominated by *Rhizophora mucronata* (see Nowak 2008), adjoined, at the time of our study, coral rag forest with *Macphersonia gracilis*, *Eugenia capensis*, and *Polysphaeria parvifolia*; regenerating areas with pioneering species such as *Croton pseudopulchellus*; woody climbers such as *Monanthe taxifolia*; and beach vegetation, which included *Rhus natalensis*, *Vitex trifolia* and *Hibiscus tiliaceus* (see Nowak 2007).

Vundwe, a smaller (1.4 km²) and uninhabited island, lies 300 m from the southern tip of Uzi. Colobus have been reported by local people to cross between Uzi and Vundwe (c.300 m) at low tide. Vundwe Island had few mangroves and, at the time of this survey, still supported high coral rag forest with large baobab trees *Adansonia digitata* (see Nowak 2007). Vundwe is a popular commercial fishing site for fishermen from Unguja, Pemba, and mainland Tanzania.

No detailed studies on primates had been conducted at either of these two areas prior to Nowak (2007), although Silkyluasha (1981) described Kiwengwa as ‘low-density’ and Uzi Island as ‘high-density’ with respect to colobus. Both regions were and continue to be subject to high levels of human disturbance, including firewood and pole extraction, lime making, cultivation, trapping of guineafowl (*Guttera pucherani* and *Numida meleagris*), and hunting, sometimes with dogs, of bushpig (*Potamochoerus larvatus*), and netting of duiker (*Cephalophus adersi* and *C. monticola*).

Methods

Transects

Seven line transects were established: four parallel transects with a total length of 10.55 km, ranging from 1,250 m to 3,850 m, and spaced 2 km apart in Kiwengwa; and two transects spaced approximately 2 km apart in western Uzi Island and one transect bisecting the length of Vundwe Island with a combined length of 5.65 km and ranging from 1,850 m to 2,050 m (Fig. 1). Using standard line-transect methods (Peres 1999), censuses were conducted monthly over 12 consecutive months from February 2004 to January 2005, with each transect randomly replicated 17 times. Censuses were repeated two additional times six months later along each transect from July 2005–September 2005, for a total of 19 repeated surveys and 200.45 km traversed in Kiwengwa and 107.35 km in Uzi. We opted for replicated surveys along the same lines over one-off surveys along many lines in order to monitor the same areas over time and reduce vegetation disturbance.

Before initiating surveys, distance estimates and recognition of red colobus age-sex classes (consistent with those

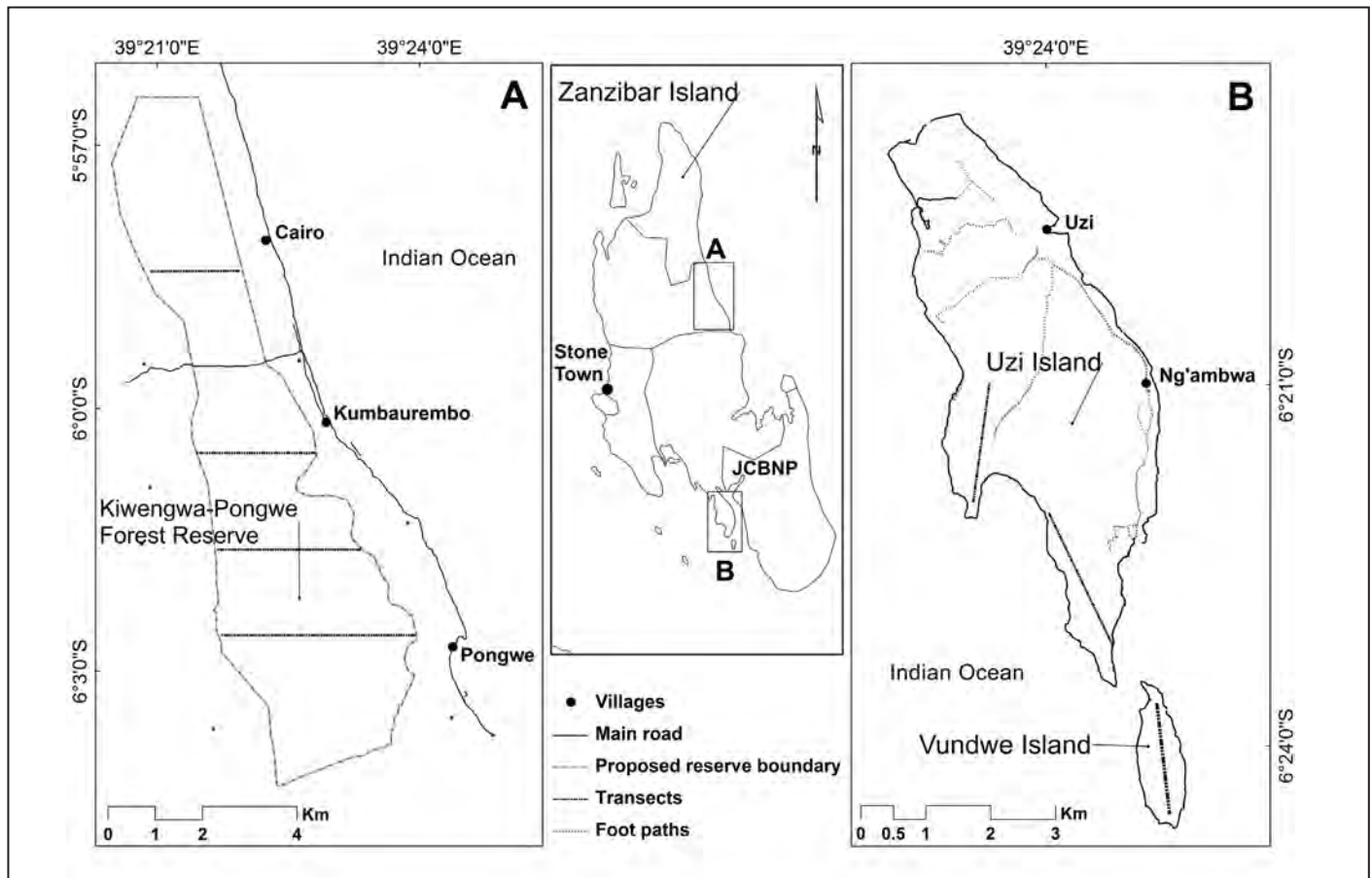


Figure 1. Study sites showing the locations of the line transects. **A.** Kiwengwa-Pongwe Forest Reserve, Zanzibar. **B.** Uzi and Vundwe islands, Zanzibar Archipelago, Tanzania.

used by Struhsaker [1975] and Siex [2003]) were practiced by observers ($n = 4$) until $>80\%$ inter-observer reliability was achieved. During encounters with colobus a cut-off of 10 minutes stationary (Marshall *et al.* 2005) was used before resuming the transect walk, and 5 minutes for the more rapidly moving Sykes's monkey. Records of humans, dogs, and signs of human disturbance along transects were also made and allotted approximately one minute for confirmation.

Statistical analyses were carried out in SPSS v. 16 (IBM Corporation).

Seasons and vegetation

Seasons were categorized into two wet and two dry periods on the basis of mean rainfall and temperature (Nowak 2007). Rainfall peaks occurred during the long rainy season, March–May (wet 2) and short rains occurred from October to December (wet 1). The extended dry season spanned June–September (dry 2) and the hottest dry period was between January and February (dry 1).

Vegetation was sampled every 50 m along transects in 5×50 m plots at the time of the census. All live woody stems ≥ 2.5 m in height were identified and measured. For multi-stemmed trees, the total number of live and cut stems was recorded, and the DBH of the five largest live and cut stems was measured (92% of cut stems were measured below breast

height). A total of 84 plots was sampled in 2.1 ha (0.75 ha along Uzi and 1.35 ha along Kiwengwa transects). Plant species' density was calculated as the number of individuals per ha. Basal area (BA) was defined as the cross-sectional area of each tree at breast height using the formula: $BA = (\pi * DBH^2 / 4)$ (Larsen 1997). For multi-stemmed trees, basal area was calculated separately for each stem and then summed for a cumulative live stem BA. Plant phenology data were collected monthly for 13 months in Kiwengwa and 10 months in Uzi to index seasonality and availability of top colobus food plants ($n = 16$; Nowak, 2007) using an index from Siex (2003) that incorporates BA.

Population estimates

Population densities were estimated in *Distance* v. 5.0 with classic distance sampling using perpendicular distances (Buckland *et al.* 2001, 2010). Observations beyond 60 m were excluded. Data were checked for observer biases detecting large over small groups and no association was found between perpendicular distance and group size for red colobus (Kiwengwa, Spearman $r_s = -0.089$, $n = 75$, n.s.; Uzi, $r_s = -0.025$, $n = 169$, n.s.) nor for Sykes's monkeys (Kiwengwa, $r_s = 0.144$, $n = 96$, n.s.; Uzi, $r_s = 0.007$, $n = 124$, n.s.), suggesting that larger groups were neither detected more frequently nor at greater distances than smaller groups.

Habitat use

To compare habitat-use, chi-square tests were used on observed and expected numbers of sightings of both primate species in six habitat types, stratified according to plant species composition using correspondence analysis (Nowak 2007). Expected values of habitat use were calculated on the basis of habitat availability along transects, estimated in GIS using Google Earth imagery and ground-truthed. Five of the six habitat types (the sixth being *shamba* or cultivated area) represented a transition zone or gradient in species diversity and canopy height that progressed from low diversity shrubland to scrub to low, medium and high coral rag forest with distance from the forest edge.

Results

Encounters

We had 857 encounters with monkeys. “Vocalization-only” encounters with no follow-up sighting made up 45% of these encounters ($n = 385$). Of the remaining 55%, we detected colobus mainly by sight, while Sykes’s monkeys were usually initially detected by vocalization and then visually detected (species by detection type, $\chi^2 = 6.076$, $df = 2$, $p = 0.048$) similar to Fashing and Cords (2000). Since 41% of detections, especially of Sykes’s monkeys, were made first by vocalization, detections based only on sound are potentially important indicators of density, especially in dense habitats, as for birds (Nelson and Fancy 1999; Lefebvre and Poulin 2003). The following analyses, however, are based only on encounters in which animals were seen, i.e. “sightings”.

In total (over the 19 repeat surveys), we saw 361 colobus groups and recorded 1662 individuals ($n_{\text{Kiwengwa}} = 130$ groups, 503 individuals and $n_{\text{Uzi}} = 231$ groups, 1159 individuals) and 496 Sykes’s monkey groups and 1,116 individuals ($n_{\text{Kiwengwa}} = 275$ groups, 471 individuals; $n_{\text{Uzi}} = 221$ groups, 645 individuals). More colobus groups were seen in Uzi than in Kiwengwa (Mann-Whitney U, $Z = 7.43$, $df = 76$, 57 , $p < 0.001$) and in Uzi, the minimum number of colobus groups seen on any census walk was between five and 14 while in Kiwengwa, the number of colobus groups seen on census ranged from one to eight. More colobus than Sykes’s monkey groups were seen in Uzi (Wilcoxon Signed Ranks, $Z = 3.26$, $n = 57$, $p < 0.001$), while no significant difference between the number of Sykes’s monkey and colobus sightings was found in Kiwengwa (Wilcoxon Signed Ranks, $Z = 0.97$, $n = 76$, n.s.). Encounter rates (number of groups/km) of both colobus and Sykes’s monkey were higher in Uzi than in Kiwengwa

(colobus, $t = -10.43$, $df = 131$, $p < 0.001$; Sykes’s monkey, $t = -4.95$, $df = 131$, $p < 0.001$) with a mean encounter rate of 0.38 (± 0.15 SD) colobus groups/km and 0.57 (± 0.70 SD) Sykes’s monkey groups/km in Kiwengwa, and 1.63 colobus groups/km (± 0.42 SD) and 1.16 (± 0.69) Sykes’s monkey groups/km in Uzi. The number of sightings did not vary with season (colobus: ANOVA, $F_{4,23} = 0.69$, n.s.; Sykes’s monkey, $F_{4,23} = 2.06$, n.s.).

Density estimates

Sightings of both species at both sites were most frequent at 10–20 m from the observer, and the number of sightings dropped off at 50–60 m from the transect line. Following truncation at 60 m, there were no significant differences in sighting distances across vegetation types (Kiwengwa, $F = 0.125$, n.s.; Uzi, $F = 0.315$, n.s.).

Population density estimates are summarized in Table 1. Mean group sizes calculated in *Distance* are based on group sizes estimated on census. These are significantly smaller than those obtained from demographic monitoring of focal red colobus groups (Nowak and Lee 2011b) and represent sub-groups, incomplete counts of groups, or small groups, difficult to distinguish while surveying. However, given our observed fission-fusion of red colobus groups in coral rag forest, we also report density expressed as individuals/km² for colobus.

Mixed-species groups

On transects, Sykes’s monkeys were more likely than colobus to occur alone. The proportion of all encounters that were with solitary individuals was 6.3% for colobus ($n = 4$ in Kiwengwa; n in Uzi = 12) and 11.8% for Sykes’s monkey (n in Kiwengwa = 11; n Uzi = 15). There were more sightings of mixed colobus-Sykes’s monkey groups in Kiwengwa than in Uzi ($\chi^2 = 3.81$ $df = 1$, $p = 0.051$). Of the 76 encounters with colobus in Kiwengwa, 18 (23.7%) were mixed colobus-Sykes’s monkey groups. Of 176 colobus encounters in Uzi, 26 (14.8%) were mixed species. Overall and for each site separately, the frequency of mixed-species groups was independent of vegetation type ($\chi^2 = 7.75$, $df = 5$, n.s.; in Kiwengwa, $\chi^2 = 9.67$, $df = 5$, n.s.; in Uzi, $\chi^2 = 6.55$, $df = 4$, n.s.).

Habitat preferences

An area of 21 000 m² was sampled in 84 plots along transects, and 16,460 trees were measured, representing 179 species from 55 families. The two sites were less similar in species composition than would be expected for sites separated

Table 1. Zanzibar red colobus and Sykes’s monkey density and population abundance estimates (“Site N”) derived from *Distance*.

Taxon	Site	N	Best-fit model	AIC	Groups/km ²	S.E.	Individuals/km ²	S.E.	Site N	S.E.
Red colobus	Kiwengwa	75	Half-normal	557.5	7.5	3.1	49.7	20.4	1149	471.8
	Uzi	174	Uniform	1294.5	29.6	2.5	196.3	17.7	1751	157.5
Sykes’s monkey	Kiwengwa	96	Uniform	740.4	8.1	2.6			922	308.6
	Uzi	124	Hazard-rate	932.1	18.9	4.7			875	224.6

by less than <30 km with a score of 0.51 on the Sorensen similarity index. Roughly half (46%) of species in Uzi were not detected in Kiwengwa, and 50% of species in Kiwengwa were not detected in Uzi. Kiwengwa and Uzi both supported representatives of 13 site-specific families. The transect in Mchangamle forest (the forest that is now gone; Nowak *et al.* [2009]) had the most unique (least similar to the other six transects) plant community assemblage including a member of the family Orchidaceae, *Vanilla roscheri*. It was also the most diverse transect in Uzi/Vundwe (88 species, 19.5 effective species, and a 2.97 Shannon Diversity Index value).

Differential habitat-use patterns of colobus and Sykes's monkeys were observed. In Kiwengwa, the pattern of habitat use by both species appeared to be preferential for high coral rag forest, whereas in Uzi, habitat use was more proportional to its availability, although a preference for medium coral rag was marked. The strongest habitat-use pattern (i.e. occurrence in high forest) was observed for colobus in Kiwengwa (Fig. 2).

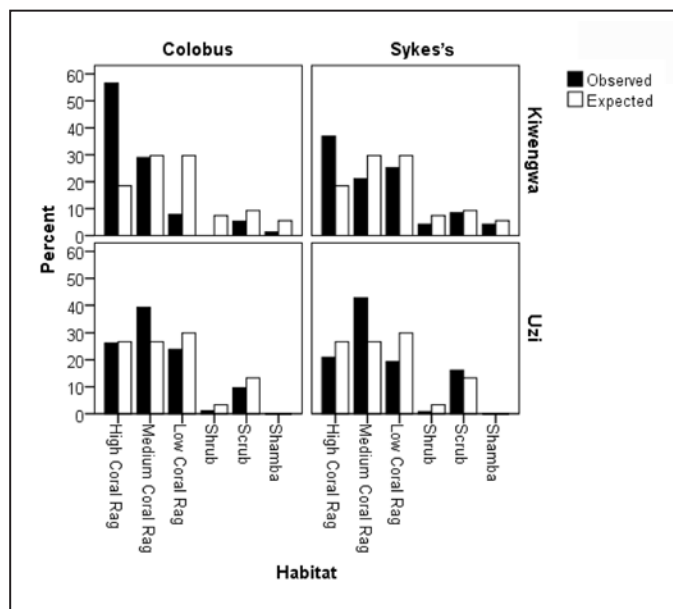


Figure 2. Percentage of observed and expected sightings of colobus and Sykes's monkeys by vegetation type and site. Shamba = cultivated land.

The higher than expected occurrence of colobus in high and medium coral rag can be attributed to the availability of colobus-preferred foods in this habitat. High coral rag in Kiwengwa and medium coral rag in Uzi had the highest sum basal area (BA) of the top ten colobus food plants. Although the mean group size of colobus was not significantly higher in medium to high canopy coral rag compared with the other vegetation types in either Kiwengwa or Uzi (t-test: Kiwengwa, -1.756, $df = 74$, n.s.; Uzi, 0.468, $df = 174$, n.s.), the maximum group sizes for colobus were recorded in medium coral rag at both sites (max in Kiwengwa = 20, mean = 6.62 ± 3.42 , $n = 76$; max in Uzi = 25, mean = 6.59 ± 4.29 , $n = 176$).

At both sites and across transects, higher plant species density was associated with higher colobus group density (Table 2), and the same general pattern was seen for Sykes's monkeys. The significant relationship between Sykes's monkeys and colobus food BA (Table 2) can most likely be explained by dietary overlap between the two species; their frequent use of *Ficus* spp. (e.g., *F. sur*, *F. natalensis*), for example.

Phenology

Production of plant parts differed between sites (ANOVA, $F_{1,115} = 23.7$, $p < 0.001$) and, controlling for site, there was significant seasonality in production (ANOVA Type I model, $F_{4,115} = 3.05$, $p = 0.02$). Kiwengwa had higher overall mature leaf availability, while Uzi had more young leaves, fruits, buds and flowers (Nowak 2007). Production as well as availability (as measured by BA and density) was less variable across seasons in Uzi than in Kiwengwa, which was a more seasonal habitat with an extended period of food scarcity from September to December. There was no evidence of a relationship between colobus density and overall mature leaf availability (ANOVA, $F_{1,27} = 15.12$, $p < 0.001$), which was higher in Kiwengwa (mean score, 8.1 ± 0.39 SE) than Uzi (6.0 ± 0.30 SE). A next step would be to evaluate leaf quality, a known correlate of colobus abundance (Fashing 2008).

Habitat disturbance

More humans, dogs, and humans with dogs were encountered in Kiwengwa ($n = 146, 13, 3$) than in Uzi ($n = 89, 0, 1$) ($\chi^2 = 7.81$, $df = 2$, $p = 0.02$). High human encounter rates

Table 2. Spearman correlations of monkey densities, human encounter rates and vegetation characteristics ($n = 7$ transects).

	Vegetation characteristics						
	Shannon Diversity Index	Density of top 10 colobus foods	BA ¹ of top ten colobus foods	Plant species density	Plant species BA	Density of cut stems	Total BA cut
Red colobus	-0.64	*0.86	0.64	**0.93	-0.25	0.36	0.00
	n.s.	$p = 0.014$	n.s.	$p = 0.003$	n.s.	n.s.	n.s.
Sykes's monkeys	-0.50	0.71	*0.82	*0.79	-0.64	0.21	0.57
	n.s.	n.s.	$p = 0.023$	$p = 0.036$	n.s.	n.s.	n.s.
Humans	0.18	-0.56	-0.25	-0.23	-0.38	0.54	0.67
	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.

¹BA = Basal area

** Correlation significant at the 0.01 level (2-tailed).

* Correlation significant at the 0.05 level (2-tailed).



Figure 3. Adult male Zanzibar red colobus *Procolobus kirkii* eating mangrove flowers on coral rag, southern Uzi Island, Zanzibar Archipelago, Tanzania. Photo by Andrew Perkin.

tended to be associated with lower numbers of monkey sightings, with an exception in Uzi, where Sykes's monkey and human encounter rates tended to correlate, suggesting their use of relatively disturbed forest at this site (Table 2).

Both overall ($r_s = 0.372$, $p < 0.001$) and in Kiwengwa, if more groups of colobus were seen, then more groups of Sykes's monkeys were seen. The other species was likely to be seen either simultaneously or at another time on the same transect in 71% of Kiwengwa and 90% of Uzi census walks. Co-occurrence between monkey species on the same transect was more likely when humans were also seen on the transect ($\chi^2 = 9.67$, $df = 1$, $p < 0.05$).

Overall, more woodcutters, dogs, and humans with dogs were encountered in medium coral rag in both sites (91 of 251 encounters). Human signs consisted mainly of cut trees, consistent with the bulk of human encounters (woodcutters), and human signs were positively associated with human encounters along the seven transects ($r_s = 0.728$, $n = 7$, $p = 0.063$). In Kiwengwa, many small stems were disturbed, while in Uzi disturbance to large trees was pronounced. Areas with higher human encounter rates tended to have smaller trees, while human sign in the form of historically cut tree BA was positively associated with human encounter rates. People appeared to be using the edge or re-growth forest having at some stage removed many larger trees.

After woodcutting, hunting in Kiwengwa was the second most common human activity, and although most hunting was for bushpig, duiker and guineafowl, shots and chases also flushed monkeys. In Uzi, boating/fishing was the second most common human activity and did not represent a direct threat to monkeys.

No significant relationships between disturbance measures and monkey densities were detected despite clear habitat-use patterns by monkeys.

Discussion

The results of these surveys, while conducted almost a decade ago, are suggestive of the relative value of degraded habitats to colobus and Sykes's monkeys. This value may



Figure 4. Adult male Zanzibar red colobus *Procolobus kirkii* sits on jagged coral rock formation, Vundwe Island, Zanzibar. Photo by Andrew Perkin.

have been elevated as a result of overall declining habitat suitability and availability on Zanzibar, and colobus and Sykes's monkeys were observed to travel through and feed in low and dense habitats (e.g., *Rhus natalensis* tangle) when moving between taller forest stands, for which they showed a preference.

The density estimates reported here act as a baseline for future, and ideally, more comprehensive, island-wide surveys of red colobus and Sykes's monkeys on Zanzibar. The sites we censused represented the second (Uzi/Vundwe) and third (Kiwengwa-Pongwe) largest populations of red colobus after JCBNP (these locations were chosen for transect surveys after visits to other parts of the island prior to establishing the study areas, as well as consultations with relevant authorities; Nowak [2007]). Colobus were encountered at a higher rate in Uzi and Vundwe islands than in Kiwengwa-Pongwe forest, where human disturbance was more frequent and dispersed, and food availability was more seasonal. In 2005, Kiwengwa and Uzi still supported populations of red colobus comparable to levels estimated by Silkiluwasha more than 30 years ago (1981) suggesting some degree of resilience to human disturbance and an ability to survive in secondary forest and at habitat edges (Onderdonk and Chapman, 2000); however, in 2011, while both species were still present at these sites, given extensive forest cutting as well as poisoning of monkeys they are presumed to have suffered significant declines (Nowak *et al.* 2009; K. Nowak, pers. obs. 2009 and 2011).

Variables other than disturbance may have contributed to observed density patterns in 2004–2005. Although colobus density estimates in Uzi were notably lower than those recorded in farm-field mosaics outside JCBNP (Siex 2003), the high density estimates in Uzi relative to Kiwengwa could have also been due to population compression. Evidence for compression (and barriers to movement) included the lack of variation in monkey densities observed across Uzi transects and between census walks relative to marked variation across Kiwengwa transects, possibly indicating wider ranging patterns and less constricted movement. Another region where population compression has occurred due to habitat loss is



Figure 5. Adult male Zanzibar red colobus *Procolobus kirkii* running along a sandy beach, Vundwe Island, Zanzibar Archipelago, Tanzania. Photo by Andrew Perkin.

Tana, Kenya, where *P. rufomitratus* lives at high densities of 165 individuals/km² (similar to the density of red colobus estimated in Uzi). The farm-field groups in Siex's study (Siex 2003), at a population density more than double of those in the Jozani groundwater forest (550 monkeys/km² compared with 235 monkey/km²), were highly unstable and living at high densities because of habitat compression rather than intrinsic growth.

The highest rate of colobus encounters, at approximately 3 groups/km (68 sightings and 90 encounters including "vocalization-only"), took place on Vundwe Island, the least human-disturbed area at the time of our study. The Vundwe population of colobus is largely isolated and we found no evidence for migration between southern Uzi and northern Vundwe; however, the island had huge potential to be a biodiversity refuge despite limits to dispersal.

Fission-fusion

One potential source of group density over-estimation was fission-fusion. This bias can be offset by reporting density as individuals/km². Siex (2003) reported that the ground-water forest groups of Jozani spent nearly 50% of their time split into two or more foraging parties, whereas the habitat-compressed shamba groups never split into subgroups. We observed a similar pattern for colobus groups in coral rag and mangrove respectively (Nowak and Lee 2011b). Fission-fusion may occur in the absence of predators and where food is scarce and clumped (Struhsaker and Leland 1979). No reports of fission-fusion in Zanzibar colobus populations other than Jozani were reported prior to this study, but fission-fusion structures have been documented in *P. gordonorum* and *P. badius badius* in heavily logged areas and areas of low food plant species diversity and density, even under risk of predation (Struhsaker 2000). Social flux in coral rag forest may be related to the observed high level of habitat degradation (consistent with the observations of Marshall *et al.* [2005] and Berenstain [1986]). As *P. badius* species rarely have group sizes of less than 10 individuals (Struhsaker *et al.* 2004; Nowak and Lee

2011b), the small mean group sizes we recorded on census suggest a habitual fission-fusion social system.

Co-occurrence of colobus and Sykes's monkeys

That Sykes's monkey sightings were positively correlated with human activity in Uzi (but not Kiwengwa) is consistent with Marshall *et al.* (2005) who found that Sykes's monkey abundance was positively related to human signs in the form of number of cut poles and timber stumps in Udzungwa Mountains, Tanzania. Cercopithecines may be more adaptable to disturbed environments and regrowth habitats than colobines (Fimbel 1994a, 1994b), but trends in Kiwengwa suggest that both species are susceptible and resilient to human disturbance. In Uzi, colobus densities were higher than those of Sykes's monkey, while no notable difference was observed in Kiwengwa, where human encounters occurred at a higher rate, vegetation was more disturbed, and the two species associated more frequently and co-occurred in the presence of humans. Unusually frequent mixed groups of *Procolobus gordonorum* and *Colobus angolensis palliatus* have been observed in areas of high human forest use in New Dabaga/Ulangambi Forest Reserve (NDUFR) in Tanzania (Marshall *et al.* 2005), suggesting that interspecific associations may increase in areas of anthropogenic disturbance or other mortality risks.

The value of unprotected areas for the future conservation of Zanzibar red colobus, and sympatric Sykes's monkeys, probably remains high, but enforcing protection of a legally protected species in unprotected areas is challenging (Mbora and Meikle 2004; Davenport *et al.* 2013). As few large forests remain on Unguja other than those described in this study—Kiwengwa-Pongwe, and Uzi and Vundwe—it is reasonable to propose that the successful conservation, timely assignment of protected status, and enforcement of regulations in these two regions (listed among 62 "Priority Primate Areas" in Tanzania by Davenport *et al.* 2013) will be important to securing the survival and genetic and behavioral diversity of Zanzibar red colobus.

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Vocal Profiles for the Galagos: A Tool for Identification



Young Kenya coast galago (*Galagoides cocos*) at Mpeketoni, north coast of Kenya. Photograph by Yvonne de Jong & Tom Butynski. For more photographs, visit wildsolutions.nl

The galagos (Family Galagidae) of Africa are nocturnal, small, and often difficult to observe, and most species are phenotypically cryptic. As such, galagos are frequently difficult to identify with confidence, particularly in the field. Being nocturnal, conspecifics mainly identify each other using auditory and olfactory cues, not visual signals. All galagos produce species-specific ‘loud calls’ (or ‘advertisement calls’). Loud calls have several functions, one of which is long-distance species identification. Knowing this, field scientists are now able to identify all currently recognized species of galagos by their loud calls.

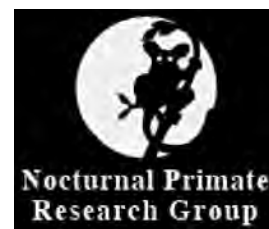
The Nocturnal Primate Research Group at Oxford Brookes University maintains a collection of the calls of African wildlife which includes more than 300 hours of recordings. These recordings have been obtained by the Group’s 24 members from many field sites over the past 40 years. From this extensive collection, 27 vocal profiles for 24 taxa of galagos have been compiled. These recordings are now freely available at: <http://www.wildsolutions.nl>

Each species presented on the website is illustrated by Stephen Nash, and there is an ‘audiomap’ that depicts the site at which each recording was made.

Additional recordings of galagos and other species will be added to this site as they become available. If you would like to hear further examples of each call type, or if you have good quality recordings of galago vocalizations that you would like to deposit with the Nocturnal Primate Research Group, please contact Simon Bearder at: skbearder@brookes.ac.uk

This product is the result of a joint initiative of the Nocturnal Primate Research Group and the Eastern Africa Primate Diversity and Conservation Program.

Simon K. Bearder, Thomas M. Butynski & Yvonne A. de Jong



Conservation and the Current Status of the Golden Langur in Assam, India, with Reference to Bhutan

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Abstract: The golden langur (*Trachypithecus geei*), which only became known to science in 1953, is endemic to western Assam, India, and southern Bhutan. The Indian population had been greatly depleted due to a fragmented range and the species was declining radically in 1997 with a pessimistic view for its future. The Golden Langur Conservation Project was begun in 1998 with the goal of protecting the golden langur within its entire Indian range. At the time of the project's initiation, the species was considered India's most endangered primate due to limited range and major deforestation (50%) as a result of a complex political situation from militants in the forest threatening the Assam Forest Department staff, and ethnic violence. The project worked with regional non-governmental organizations and government agencies using the following tools to effect conservation contagion: 1) community meetings; 2) involving villages in forest committees and "Self Help Groups" for economic development; 3) formal seminars; and 4) celebratory events for the creation of the Manas Biosphere Reserve. The project developed conservation contagion, resulting in villages creating their own conservation groups to participate in the project, eventually resulting in 18 community groups forming Forest Protection Forces collectively, under the Unified Forest Conservation Network, to protect almost the entirety of the Manas Biosphere Reserve as well as other reserve forests in Assam. This community protection resulted in an increase of the Indian population of golden langurs from c.1,500 in 1997 to c.5,600 langurs in 2007 to 2012. The project also resulted in the lifting of the "in danger" listing for the Manas Biosphere Reserve by UNESCO. The Indian population of golden langurs mainly resides in the Manas Biosphere along the Bhutan border and in a number of southern isolated reserve forests north of the Brahmaputra River. In adjacent Bhutan, the southern subspecies is contiguous with its Indian counterpart and with the northern subspecies, which has more gray on its arms and tail and inhabits higher altitudes. The Bhutan population is grossly estimated at over 6,600 langurs based on a population census of 60 km², giving a total estimate for the species in Assam and Bhutan of over 12,000 individuals. The potential for community conservation in Bhutan is also discussed.

Key Words: Golden langur, *Trachypithecus geei*, *Presbytis geei*, golden leaf monkey, Assam, Bhutan

Introduction

The Golden Langur Conservation Project (GLCP) was initiated in 1998. It brought together methods, developed elsewhere, of catalyzing communities to become involved in conservation (Horwich and Lyon 2007; Horwich *et al.* 2011, 2012), with the goal of protecting India's most seriously endangered primate (Mukherjee and Southwick 1997). In the course of its progress, the project demonstrated the power of using the golden langur as a flagship species to protect landscapes and ecosystems in western Assam, including those of the Manas Biosphere Reserve forests (Horwich *et al.* 2010).

The golden langur, *Trachypithecus geei* (Fig. 1), formerly *Presbytis geei*, was one of the more recent primate discoveries to the scientific community in India. Its striking golden-orange pelage attracted attention as early as 1907 but it was formally discovered by E. P. Gee in 1953 and described by Khajuria in 1956 (Gee 1955, 1964; Khajuria 1956, 1978). Its restricted range between the Sankosh and Manas rivers in western Assam and Bhutan (Choudhury 1996, 2008) has made it, since its discovery, one of India's rarest and most endangered primates (Mukherjee and Southwick 1997). It is listed in Appendix I of CITES, as Endangered on the *IUCN Red List of Threatened Species*, and in Schedule I of the Indian Wildlife (Protection) Act, 1972 (amended 1991).

Its range has been considerably depleted and fragmented (Choudhury 2002), with a total gross estimated population in Bhutan and India of about 4,500–5,000 individuals (Biswas 2005). This was based on absolute counts in Assam of 1,064 monkeys (Srivastava *et al.* 2001b) and an estimate for Bhutan of over 4,000 langurs based on a census in one small area (Wangchuk 1995). Their long-term survival depends on genetic exchange; golden langurs are now restricted to small, isolated populations through much of their range (Biswas 2005). In 2001, the Indian population was estimated to be less than 1,500 langurs (Srivastava 2006).

The golden langur occurs in lowland evergreen, semi-evergreen and riparian moist deciduous and sal-dominated, moist deciduous forest (Srivastava *et al.* 2001a; Biswas 2005; Bezbaruah 2004) in the Brahmaputra River valley and the foothills of the Black Mountains of Bhutan. Its range is bordered in the north by the Bhutan hills, in the south by the Brahmaputra River, in the west by the Sankosh River on Assam's western border, and by the Manas River in the east (Fig. 2). In Assam, its main population is in the Manas Biosphere Reserve, a forested area along the border of Bhutan. There are also some significant populations in isolated forests south of the Manas Biosphere Reserve (Deuti 2005) (Fig. 2).

Reasons for population decline

With the initiation of the Bodoland autonomous movement in 1993, a result of increasing numbers of non-Bodos entering Assam, extremist groups took refuge in the Assam forests. This led to major deforestation. Initially, some extremists financed their movement through timber extraction. Illegal logging increased, encouraged by the lack of forest protection by the Assam Forest Department. The local people were employed to cut the trees, receiving a pittance in return. This lack of forest protection continued as extremist groups harried the Forestry Department staff and other government workers by murders and kidnapping. Forestry



Figure 1. Male golden langur from Kakoijana with subgroup in village bamboo before reaching their new home in Bhubeshwar (see Fig. 4). Photograph by Arnab Bose.

staff were unable to enter the forest because of the extremist groups. Deforestation was further compounded by corruption on the part of some forestry staff, and exacerbated by ethnic clashes between Bodo and Adivasi tribes in 1996 and 1998. Villagers at that time were forced to leave the forest to reside in relief camps on the southern borders of the reserve forests. Although the encroaching villages could not be reestablished in the reserve forests, some villagers began to harvest timber for the illegal loggers as well as for firewood. In Kokrajhar District alone there were 200,000–300,000 refugees in relief camps, which were still in existence in 2005 if on a smaller scale (Horwich 2005). This complex political situation resulted in the deforestation of almost 50% of the reserve forests in western Assam. The increasing human population in proximity to langur populations in reduced, degraded and fragmented forests has resulted in langurs being killed by people and dogs or succumbing to electrocution when jumping onto power lines.

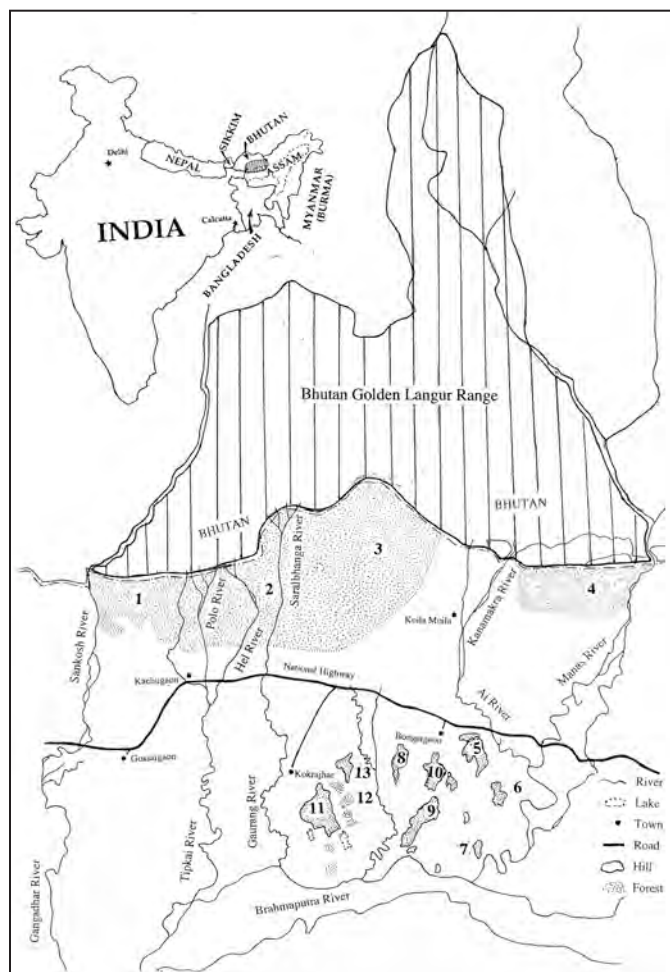


Figure 2. Map of the geographic range of the golden langur with the Indian locations censused (see Table 1). 1. Kachugaon Division, 2. Haltugaon Division, 3. Chirrang Division, 4. Manas National Park, 5. Kakoijana Reserve Forest, 6. Baumugaon Reserve Forest, 7. Khoragaon Reserve Forest, 8. Bhumeshwar Reserve Forest, 9. Bhairab Reserve Forest, 10. Nakati Reserve Forest, 11. Chakrashila Wildlife Sanctuary, 12. Abahaya rubber garden, 13. Nadangiri Reserve Forest. The range of the golden langur in Bhutan follows Wangchuk *et al.* (2008).

Conservation Action – The Golden Langur Conservation Project (GLCP)

The Golden Langur Conservation Project (GLCP) was begun in 1998. Its basis was a scheme of community conservation that has been used effectively elsewhere to bring about a regional change in the exploitation of agricultural land and forests which promotes wise use, sustainability, and the conservation of the forests and their wildlife (Horwich and Lyon 2007; Horwich *et al.* 2010). In this case, the project targeted most of the range of the species, including its principal strongholds in the Manas Biosphere Reserve.

The GLCP began as an extension of the Indo-US Primate Project to address the plight of the golden langur and its habitat in western Assam (Horwich *et al.* 2010). It was initiated by three non-governmental organizations: Community Conservation, Nature's Foster, and Green Forest Conservation. Later, five Assamese non-governmental organizations and one from the US joined forces to create the Manas Biosphere Conservation Forum (Aaranyak, Community Conservation, Green Forest Conservation, Green Heart Natures Club, Nature's Foster and New Horizon) with the specific aim of protecting the golden langurs and their habitat in western Assam. Each regional NGO focused on a specific area in the range of the golden langur. The Forum worked collectively with over 140 communities, assisting them in creating "Self Help Groups" (for the improvement of their economic situations) and forest protection committees. The interaction of these groups has empowered these communities that now actively replant, patrol, and protect their forests in coordination with the Assam Forest Department and the Bodoland Territorial Council (Horwich 2005).

A number of strategies were used to motivate the communities (Horwich *et al.* 2010). They included: 1) setting up a series of small community meetings requesting help from the communities to protect their unique forests and wildlife; 2) involving communities in creating forest protection committees and Self Help Groups for economic development; 3) holding more formal seminars that included communities, government agencies and NGOs; and 4) organizing four Manas Biosphere Celebrations reaching 8,000, 5,000, 20,000 and 35,000 participants across the biosphere reserve.

The NGOs worked with individual communities, and the interaction of the Forest Protection Committees and the Self Help Groups evidently empowered the communities in their actions. Over the years, the GLCP encouraged communities in the biosphere reserve and other reserve forests to form groups, and an atmosphere of conservation contagion (Horwich *et al.* 2012) developed around the projects as community conservation groups formed and joined the conservation activities of the project.

Results

In 2004, with an accord signed between one Bodo militant group and a second group declaring a ceasefire, and the

establishment of the Bodoland Territorial Council (BTC) to administer the area under the state of Assam, it seemed that there was renewed hope for the forests. The illegal loggers, however, still logged the forests with impunity, threatening both villagers and the Forest Department staff alike. In an attempt to halt this situation, the late Rajen Islari of Green Forest Conservation approached Kampa Borgoyari of the BTC requesting support for a 100-man Forest Protection Force that he would appoint and organize to protect the western reserve forests. Other community groups followed suit and began to protect their adjacent forests. Currently, with support from BTC and grants from the United States Fish and Wildlife Service Asian Elephant Program, 18 community groups are protecting almost the entire biosphere reserve. Six groups working with the GLCP protect Kachugaon, Ripu, Chirang and Manas reserve forests, while another 11 groups are clustered around the Manas National Park working with their staff, and an 18th group protects an area of the biosphere reserve east of the national park. The Golden Langur Conservation Project by 2005 had evolved into the Manas Elephant Protection Project that has catalyzed the formation of these community forest protection forces into the Unified Forest Conservation Network, which collectively, with the Bodoland Territorial Council and the Assam Forest Department, protects almost the entire Manas Biosphere Reserve.

In addition, Kakoijana Reserve Forest (17 km²) has become a model project, and 34 villages created two federations (Green Conservation Federation and Nature Guard) to jointly protect their forest. Their work has increased the canopy from 5% to 80%, which has been accompanied by an increase in the golden langur population from less than 100 to now over 500 animals (Horwich *et al.* 2012).

The actions of the GLCP generated over time a great deal of enthusiasm and community action that has resulted in a major revival of the Indian population of the golden langur. The most recent censuses in 2008–2012, which used project-trained villager researchers, of the entire Indian golden langur population have shown a major increase from 1,500 langurs in 1997 (Srivastava 2001b) to over 5,600 langurs in 2008–2012. This was thanks to the communities who have helped protect the forests in western Assam.

Table 1 and Figure 3 show the results of these censuses. The 1960 estimate is from Gee (1964), that for 1996 was from Mukherjee and Southwick (1997), the 2000 estimate from Choudhury (2002), the 2008 estimate from Ghosh (2008a, 2008b) and the 2009 estimate and Table 1 are a collective estimate from a compilation described in the *Assam Tribune* (2009; J. Biswas pers.comm.; Ghosh 2008a; A. Bose unpubl. data 2007, 2012).

The results of the GLCP and what it has become has stimulated the delisting of the "in danger" label to the Manas Biosphere Reserve by UNESCO. Besides the increase in golden langurs there is also evidence that the elephant population (Ghosh 2008b) and the tiger population (India Ministry of Environment and Forests 2011) are remaining stable and perhaps even increasing as well.

Table 1. Recent censuses of golden langur. *Trachypitecus geei*, in India.

	Area censused	Population	Reference
1	1. Kachugaon Division	2,293	<i>Assam Tribune</i> (2009)
2	2. Haltugaon Division	1,461	<i>Assam Tribune</i> (2009)
3	3. Chirrang Division	263	<i>Assam Tribune</i> (2009)
4	4. Manas National Park	214	<i>Assam Tribune</i> (2009)
5	5. Kakoiijana Reserve Forest	507	Bose (2012)
6	6. Baumugaon Reserve Forest	30	Bose (2007)
7	7. Khoragaon Reserve Forest	36	Bose (2012)
8	8. Bhumeshwar Reserve Forest	53	Bose (2012)
9	9. Bhairab Reserve Forest	49	Bose (2007)
10	10. Nakati Reserve Forest	14	Bose (2007)
11	11. Chakrashila Wildlife Sanctuary	501	Ghosh (2008a)
12	12. Abahaya rubber garden	112	Ghosh (2008a)
13	13. Nadangiri Reserve Forest	66	Ghosh (2008a)
	Total	5,599	

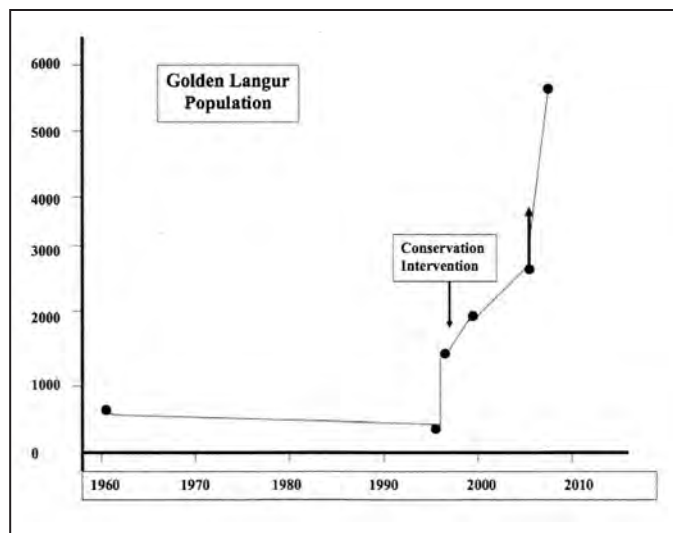


Figure 3. Golden langur population estimates for its geographic range in India (Gee 1964 for 1960; Srivastava *et al.* 2001b for 1997; Choudhury 2002 for 2000; Ghosh 2008a, 2008b for 2008; *Assam Tribune* 2009, Bose 2007, 2012, and Ghosh 2008a for 2009).

Community Conservation

Community conservation projects have been criticized by both biologists (Oates 1999; Terborgh 1999) and sociologists (Belsky 1999). This has mainly been due to lumping them with the large-scale Integrated Conservation and Development Projects (ICDPs), which are top down and treat rural community members as the problem rather than the solution (Horwich and Lyon 2007; Horwich *et al.* 2011, 2012). The GLCP has shown that communities when treated with respect and when asked for their help respond overwhelmingly positively. The GLCP developed slowly and had a contagious effect on the communities that often joined the project on their own and formed their own community conservation groups. They show what a powerful ally and partner local rural people can be, because they are on-the-ground partners. India seems to be particularly fertile ground for community conservation and a recent directory shows a minimum of

hundreds of community conservation projects throughout the country (Pathak 2009).

Because of the success of this project in motivating communities to protect their environment, we are now attempting to interest the Bodoland Territorial Council and the Assam State government in community co-management for the Manas Biosphere Reserve and experimental community village forests for Kakoiijana and other reserve forests (Horwich *et al.* 2010; Horwich 2011).

Golden Langur Population in Assam and Bhutan

The community reforestation and forest protection efforts showed how resilient the golden langur species is once they and their habitat are protected. Despite continued negative references by scientists about signs of a declining population (Srivastava *et al.* 2001b; Srivastava 2004, 2006; J. Biswas in the *Assam Tribune* 2009), the community protection efforts have led to a major increase in the golden langur population from an estimated 1,500 langurs in 1997 to currently over 5,600 langurs. The southern “island” fragmented populations, also thought to be not viable, have responded extremely well. Thirty-four communities surrounding Kakoiijana Reserve Forest have formed two federations that worked together to bring the 17 km² forest back from 5% to 80% canopy cover, and the langurs responded by increasing from less than 100 animals to now over 500. Despite the isolated appearance of these reserve forests, a few golden langurs have left Kakoiijana and found their way via existing corridors to Bhumeshwar Reserve Forest about 10 km away (Fig. 4).

It seems, therefore, that there are now four principle viable golden langur populations. 1) Kachugaon, Ripu and Manas reserve forests have continuity, although there is a major gap around Koila Moila. This is the largest population with 3,754 langurs. 2) A population of eastern Manas Reserve Forest. It is degraded but connected to the Manas National Park that has 477 langurs. Both of these populations have continuity with the Royal Manas National Park, Phipsoo Wildlife Sanctuary and Jigme Singye Wangchuck National Park in Bhutan. 3) A population centered on Chakrashila Wildlife Sanctuary, Nadangiri Reserve Forest and the rubber garden on the west side of the Champamati River, which has 679 langurs. 4) A population centered on Kakoiijana, Nakkati, Bhairab and Bhubeshwar Reserve Forests to the east of the Champamati River, which has 689 langurs. With continued protection and added reforestation, the two biosphere reserve populations may continue to increase, and, due to their being adjacent to the Bhutan populations, they may be considered connected to the population in the south of Bhutan (Wangchuk 2005).

The two southern populations although seemingly fragmented have been shown to have at least some viable corridors to each of their respective fragments on the west and east of the Champamati River. Given the langur response to the protection of Kakoiijana, it is probable that with increased community protection and reforestation these two populations

could increase to about 1,500–2,000 langurs each. The GLCP is now focusing on them for community protection and creating corridors between the island forests (Fig. 4). A long-term goal is to eventually connect these island populations to the Manas Biosphere Reserve with a corridor along the Cham-pamati River and beyond.

In Bhutan, Wangchuk (2005) conducted two censuses in 1994 and 2003 in the Mangde Chu Valley in Central Bhutan on ten 4-km transects on trails for convenience in the rugged terrain, over a period of seven days. He found 127 langurs in 58.5 km². From that census, he estimated the available habitat under 2,300 feet in altitude. The geographic range of the species in Bhutan is roughly 2,000 km² giving an estimate of 4,341 langurs. In 2003, he found 130 langurs in 60.5 km², almost the same density as in 1994. However, in 2003 he was able to survey more habitat and could make a better estimate of the total available. He adjusted it to 3089 km² and thus revised the estimate of the total golden langur population in Bhutan to 6,637. The substantial increase was largely due to a more accurate understanding of their distribution in Bhutan. Wangchuk (2005) noted that his estimate may be a bit high due to his censuses having been carried out in pristine habitat; some of the available habitat, especially, in the south may be more degraded due to human activities and consequently have lower densities. The density which Wangchuk noted as being relatively high, was 2.1 individuals/km². Wangchuk

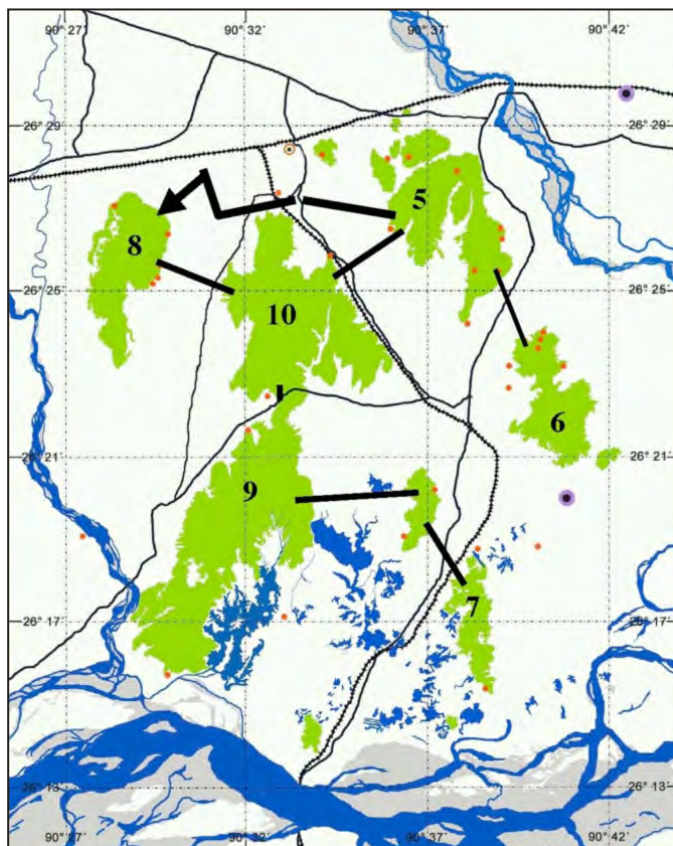


Figure 4. Map showing the 10-km path traveled (thick black line with arrow) by a subgroup leaving Kakoijana Reserve Forest (5), bypassing Nakkati (10) due to rice fields and ending in Bhubeshwar (8) about four months later. Thinner black lines between reserve forests indicate the plan for potential corridors.

(2005) differentiated two subspecies of golden langurs: the northern subspecies that has more gray in the arms and tail and the southern (nominate) subspecies that is similar to the langurs in Assam. He notes that the habitat available to the northern subspecies, which is higher in altitude and is a more temperate forest, is almost three times the area of the southern subspecies that inhabits a subtropical habitat as in Assam.

In Assam, using the absolute count in this paper of now over 5,600 golden langurs and Srivastava's estimate of the golden langur range to be 1506.47 km², the density has increased from 1 individual/km² in 1997 to about 3.71/km² in 2009; almost twice the density in Assam compared to Bhutan. This high population density in Assam may be due to the difference in the altitude between the southern and northern subspecies. With the increase of golden langurs in Assam due to our community conservation intervention, and the Bhutan estimate, the total golden langur population could be estimated now at over c.12,000.

There is an area of overlap of golden langurs with the capped langur, *Trachypitecus pileatus*, in Bhutan where hybridization has occurred (Wangchuk 2005; Choudhury 2008). Choudhury (2008) saw these langurs in Zhemgang District. He noted that they resemble more the golden langur with some non-uniform variations from the capped langur including blackish horn-like tufts and grayish forearms, thighs, and tails. Wangchuk noted that the contact zone was adjacent to the Chamkhar River valley in Zhemgang District. He felt that the Chamkhar River complex had acted as a restrictive boundary until recently constructed bridges allowed the langurs to cross the river and interbreed.

Community Conservation in Bhutan

Wangchuk (2005) discussed the historical and current land practices in Bhutan. He noted that customary law in the past was overlain by government laws when the forests were nationalized in 1952 but that customary law is maintained still in its historical practices as evidenced by interviews he conducted. Landowners continue to make decisions about their lands and its management. They have left the environment in its natural state and thus have things to teach us. This is most important since the Bhutan government capacity to protect the forests is limited.

Wangchuk (2005) examined the various structures of the Bhutan governmental agencies in regard to their abilities and the terrain and makes the case that they are not in a position for complete command and control of all of the country's forests and wildlife. This is emphasized by the fact that 80% of Bhutan's population is rural and depends on forest resources and can thus "make or break Bhutan's biodiversity conservation objectives." He showed that the village systems are basically democratic while the government system is more bureaucratic and authoritarian. In golden langur habitat there is inadequate enforcement, yet the golden langur population remains stable. Wangchuk thus suggests that the solution to his extensive analysis is to enlist support of the main stakeholder—the rural

populace—as forest protectors. The long-term recommendation is to return the traditional community forests to the local people with a system of monitoring and evaluation in place. He developed a plan for gradual transition from government to village control. This is similar in function to the community co-managed system we have recommended to the Bodoland Territorial Council for the Manas Biosphere Reserve. Perhaps, if Wangchuk's concepts can come to fruition, the possibility may arise for villager groups from Assam and Bhutan to ensure important protection of the transnational boundary to jointly protect the contiguous border forests and the wildlife that does not recognize such political boundaries.

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Habitat Use and Documentation of a Historic Decline of Western Hoolock Gibbon (*Hoolock hoolock*) in Dampa Tiger Reserve, Mizoram, India

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Abstract: The western hoolock gibbon *Hoolock hoolock* is globally threatened because of a combination of habitat loss, fragmentation and hunting. Most remaining populations are isolated and contain few individuals. We studied a small population of western hoolock gibbons in Dampa Tiger Reserve in Mizoram, India, an area with a deep-rooted tradition of hunting wildlife. We quantified differences in 35-ha sites that were used by gibbons ($n = 9$) with those that were not ($n = 6$). We found no statistical differences with respect to canopy continuity, distance to the nearest village or levels of habitat disturbance, but were not able to quantify levels of hunting. Interviews with local villagers ($n = 53$) from seven villages distributed throughout the reserve suggests that gibbon numbers declined progressively from the early 1970s to the present day, possibly from >100 to <50 individuals. Interviewees pointed at the combined result of fire (from, for instance, slash-and-burn cultivation), reduction of the available habitat, and hunting, exacerbated by an increase in the human population, as possible causes. This corresponds well with our analysis of vegetation maps of the reserve based on satellite imagery, that show a decline in suitable habitat from 63% of the reserve in 1978, to 59% in 1989, and 50% in 2005. We strongly recommend a greater emphasis on quantifying the impacts of hunting on the distribution and persistence of hoolock gibbons.

Key words: Disturbance, habitat, hunting, interviews, population numbers, status assessment

Introduction

The western hoolock gibbon (*Hoolock hoolock*) is the westernmost of the 16 species of smaller apes of South-east Asia (Geissmann 2007). Its range is restricted to the monsoon evergreen and semi-evergreen rainforests of north-east India and eastern Bangladesh, south and east of the Brahmaputra River, and north-west Myanmar, west of the Chindwin River (Brockelman *et al.* 2008). East of the Chindwin River it is replaced by its congener, the eastern hoolock gibbon (*H. leuconedys*) (Brockelman and Geissmann 2008). The western hoolock gibbon is predominantly frugivorous (Ahsan 2001; Islam and Feeroz 1992), confined to tall forest with continuous canopies (Choudhury 1991), and is instantly recognized by its loud and characteristic songs. Groups generally comprise an adult pair with their (dependent) offspring. Average group sizes are between three and four individuals (Das *et al.* 2009). The combined effects of habitat loss, fragmentation (especially in India driven partially by slash-and-burn or 'jhum' cultivation) and hunting (for food and because of

alleged medicinal properties) have led to the species being categorized as Endangered according to IUCN threat criteria (Brockelman *et al.* 2008). With a population of about 300 individuals in Bangladesh and about 2,500 in India, and an unknown number in Myanmar, the species has been on the of the World's 25 Most Endangered Primates list since 2006 (Walker *et al.* 2009). It is protected throughout its range, and included on Schedule I of the Indian Wildlife (Protection) Act of 1972. It is a protected species in Myanmar through the Protection of Wildlife and Wild Plants and Conservation of Natural Areas Law of 1994, and in Bangladesh it is protected under the Bangladesh Wildlife (Preservation) (Amendment) Act, 1974.

Because of continued destruction of its habitat through commercial logging, fragmentation and degradation, coupled with hunting pressures, most populations of western hoolock gibbons are isolated and small, with about 80% of those assessed in India and Bangladesh harboring fewer than 20 individuals, and over half having fewer than 10 (Walker *et al.* 2007). An important determinant of the populations of

gibbons, or primates in general, in the region is the level and intensity of hunting. Even with a good and intact habitat, gibbons are often absent as hunting leads to direct extermination of individuals. In the range of the western hoolock gibbon, the level and intensity of hunting differs from one tribal community to the other and from one region to the next. Even in India, where the majority of the people have a sacred reverence towards primates in general (Lee and Priston 2005), in parts of the country such reverence is absent. This could be attributed to the cultural and religious differences of the different individual tribes inhabiting the different states. We conducted research on western hoolock gibbons in Mizoram State, north-eastern India. Throughout the state the majority of the people have an intricate and deep history of hunting. While national laws prevent them from doing so, the remoteness of the area, and possibly lack of awareness, results in less than optimal enforcement of these non-hunting laws (see Gupta and Sharma 2005).

We set out to study the responses of western hoolock gibbons towards various disturbances in terms of their habitat use in the westernmost part of Mizoram State. We took the opportunity to make an attempt to retrace the population decline by prompting tribal villagers to recount population size. We show that structural measurements of the forest and distance to human habituation do not adequately explain the presence or absence of gibbons in different parts of the reserve, and report that the decline of gibbons as experienced by the tribal villagers does correspond well with the observed decline in gibbon habitat.

Study Area

The Dampa Tiger Reserve (23°20' to 23°47'N and 92°15' to 92°30'E) in western Mizoram was chosen as the study area after a preliminary investigation on the protected areas of Mizoram (Raman *et al.* 1998; Gupta and Sharma 2005). The main considerations were that this reserve has a mosaic of habitats with varying degrees of anthropogenic influence, from open *jhum* fallow lands to primary undisturbed forests, resulting in a high degree of forest fragmentation (Fig. 1). The 500-km² reserve was notified as a sanctuary in 1985 and subsequently afforded a Tiger Reserve status in 1994. Its westernmost border follows the Khawthlangtuipui River, which forms the international border with Bangladesh. The reserve covers mountainous terrain with elevations from 250 to 1100 m above sea level. Situated on the Tropic of Cancer, Dampa experiences a seasonal climate with relatively mild winters (December to February, average temperature of 15°C), a warm summer and a distinct rainy season from May to October.

The area is covered in tropical evergreen and semi-evergreen forests, as well as tropical moist deciduous forests, and at higher elevations, above 700 m above sea level, sub-montane forests. The low, moist valleys have evergreen vegetation characterized by species like *Michelia champaca*, *Dipterocarpus turbinatus*, and *Terminalia chebula*. The higher slopes are

characterized by species such as *Castanopsis indica*, *Schima wallichii*, *Mesua ferrea* and occasionally *Quercus* sp. On the steep western slopes the forest is more open with many deciduous species (for example, *Lannea coromandelica*, *Sterculia villosa* and *Gmelina arborea*) and large patches of *Dendrocalamus longispathus* bamboo, and expanses of open grassland on rocky surfaces on the highest slopes. Dampa's natural vegetation thus contains a cross-section of habitats from grasslands, successional habitats, and open forests to dense, lofty, primary evergreen forest.

The reserve provides a habitat for several endangered species such as tiger (*Panthera tigris*), clouded leopard (*Neofelis nebulosa*) and Asiatic elephant (*Elephas maximus*). It is especially rich in primates with, apart from the western hoolock gibbon, capped langurs (*Trachypithecus pileatus*), Phayre's langur (*T. phayrei*), rhesus macaque (*Macaca mulatta*), Assamese macaque (*M. assamensis*), northern pig-tailed macaque (*M. leonina*), stump-tailed macaque (*M. arctoides*), and Bengal slow loris (*Nycticebus bengalensis*).

Methods

Habitat use

The senior author, with the aid of two assistants, worked in the reserve from 10 December 2010 until 7 May 2011. It was not possible to visit the south-western parts of the reserve as logistical and linguistic reasons prevented us from working closer to the Bangladesh border. We set up 35-ha plots in 15 locations (35 ha is approximately the home range size of western hoolock gibbons: Feeroz and Islam 1992) to assess the presence of gibbons, and measure vegetation. Each plot was surveyed for five days, for a median duration of 18.8 hrs (interquartile range (IQR) 17.7–19.7 hrs) and covering 19.9 km (19.3–20.3 km). We established whether or not gibbons were present, and if so, their group sizes and composition. In each plot we estimated the canopy continuity (as a percentage) and, using a handheld GPS, the distance from the plot to the nearest village. We quantified disturbance by assessing the presence of trails (none or a single trail was scored as 0, multiple trails, up to 7, scored as 1), presence of cut tree stumps (absent 0, present 1), traces of non-timber forest collection (absent 0, present 1), presence of forest camps (absent 0, present 1), presence of *jhum* (absent 0, present adjacent to the plot 1). These were summed to arrive at a disturbance level score (from 0 to 5). We compare plot characteristics (canopy continuity, distance to village and disturbance levels) between plots with and without gibbons. We use non-parametric statistics, reporting medians and interquartile ranges (IQR), and accepting significance when $P < 0.05$ in a two-tailed test (Siegel 1956).

Historic decline

We conducted village interviews in two forest management ranges—Teirei on the western and northern side (three villages: Teirei, Damparengpui, Tuipuibari) and Phuldungsei in the south-east (four villages: Phuldungsei, Lallen, Saithah,

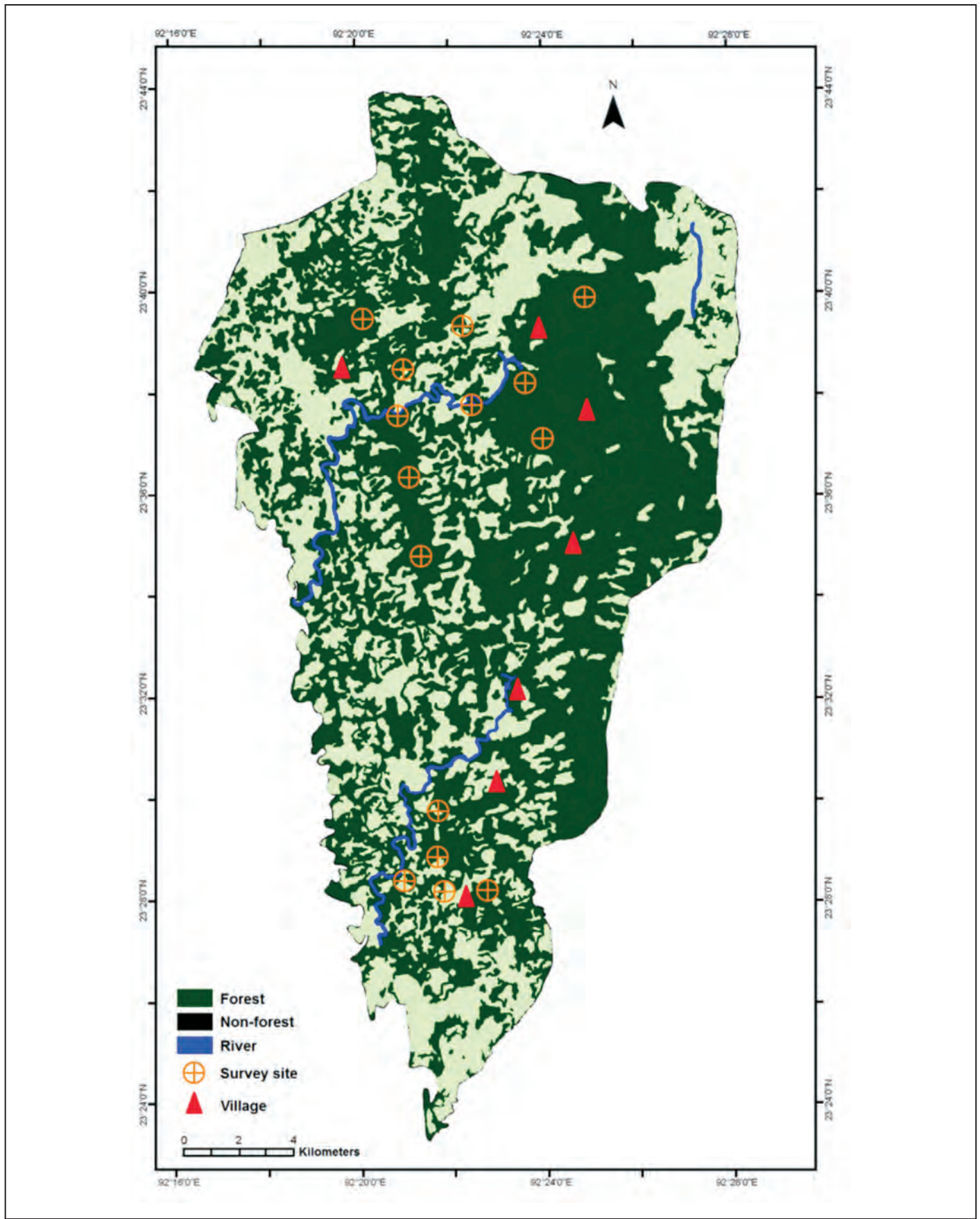


Figure 1. Dampa Tiger Reserve, in western Mizoram, India, showing habitat types suitable for gibbons (closed evergreen and semi-evergreen forest, open evergreen and semi-evergreen forest, closed and open moist deciduous forest) in dark green, habitats unsuitable for gibbons (bamboo, shrub forest, current and abandoned *jhum*, and villages) in pale green, and the survey sites (circles for gibbon surveys and triangles for village surveys). Modified from satellite imagery provided by the Mizoram Remote Sensing Application Centre.

and Kawnmawi)—with the aim of establishing changes in gibbon abundance over time (for locations of the villages inside the reserve see Figure 1). In Teirei, the villagers belong to the Bru tribe whereas in Phuldungsei they were Mizo. We focused on the elder villagers with, by their own account, a good knowledge of the forest and its fauna, resulting in a clear bias towards males. In both ranges a similar number of interviewees were selected with similar ratios of men to women, and of a similar age (Teirei, 25 men and one woman, median age 56 years; Phuldungsei: 23 men and four women, median age 55 years). Interviews were conducted in the Mizo language (Phuldungsei and parts of Teirei) by the senior author, or, with the aid of an assistant, in the Bru language (parts of Teirei). Each interview was conducted in the interviewee's house, and to ensure independence, each interviewee was questioned separately (Lammertink *et al.* 2003).

We asked interviewees about their perception of the population sizes of gibbons within their management ranges, in three predefined classes: >100, between 50 and 100, and <50, in three time periods. As the most distant time period we selected the late 1960s to early 1970, this being defined (and remembered by most interviewees) by a period of civil unrest (*Buai Kum*); the second time period covered the mid-1970s to the mid-1990s, defined as the period after the civil unrest but prior to the declaration of the area as a Tiger Reserve in 1994; the third time period was the present, defined as 2011 or one or two years prior. Interviewees born between 1966 and 1980 were only asked about possible changes between the latter two time periods.

We obtained detailed vegetation maps of the study area prepared by North Eastern Space Applications Centre (NESAC), Meghalaya and Mizoram Remote Sensing Application Centre (MRSAC), Mizoram, based on satellite images from the years 1978, 1989, and 2005 (MRSAC 2008). The satellite images covered over 80% of the reserve (416 km² of about 500 km²), including all of the regions we worked in. For analysis we distinguished all forest types used by gibbons into two major categories: Closed Evergreen/Semi-Evergreen Forest, Open Evergreen/Semi-Evergreen Forest, Closed Moist Deciduous Forest and Open Moist Deciduous Forest as *Forest* (habitable by gibbons) and deforested and regenerating forest in early stages of succession (Forest Blank; Scrub Forest), slash-and-burn agriculture (Current *Jhum*; Abandoned *Jhum*), bamboo patches (Bamboo) and Villages as *Non-Forest* (Inhabitable by gibbons)

Results

Habitat use

We found nine groups of gibbons, with a median group size of 3 (range 2–4), with no more than one group present in each of the plots. Each group contained one adult male, one adult female, and up to two young and the total number of gibbons we recorded was 31. There was a clear relationship between the distance from the plot to the nearest village and canopy continuity (Kendall's $T = 0.44$, $n = 15$, $P < 0.05$)

and disturbance level ($T = 0.60$, $n = 15$, $P < 0.005$) as well as between canopy continuity and disturbance level ($T = 0.85$, $P < 0.001$): plots at greater distances from villages had more continuous canopies and significantly less signs of human disturbance.

While most of the gibbons were found in what appeared to be 'good' forest, there was no significant difference between plots with or without gibbons in terms of nearest distance to the village (gibbons present: median distance 4.0 km (IQR 3.0–4.5 km, $n = 9$), gibbons absent: median distance 3.3 km (IQR 3.0–3.5 km, $n = 6$; Mann Whitney U, $P = 0.33$), canopy continuity (gibbons present: median canopy continuity 80% (IQR 70–90%, $n = 9$), gibbons absent: median canopy continuity 68% (IQR 60–70%, $n = 6$; Mann Whitney U, $P = 0.27$) or disturbance levels (gibbons present: median disturbance level score 2 (IQR 0–4, $n = 9$), gibbons absent: median disturbance level score 4 (IQR 2–5, $n = 6$; Mann Whitney U, $P = 0.34$).

Historic decline of gibbons and their habitat

The perceived change in abundance of gibbons was very similar in the two forest ranges. Referring to the period of civil unrest in the late 1960s and early 1970s, in Teirei 20 out of 24 and in Phuldungsei 18 out of 19 of the elder interviewees thought the population of gibbons in their area was larger than 100 individuals, with the remaining five estimating it at somewhere between 50 and 100 individuals. For the period after the civil unrest but prior to gazettement as a Tiger Reserve all interviewees were unanimous in their belief that the population was somewhere between 50 and 100 individuals. In Teirei eight and in Phuldungsei nine interviewees considered the present population to number between 50 and 100 individuals, but the remaining interviewees all were of the opinion that there were less than 50 gibbons in their part of the reserve. While the majority of interviewees agree that the population of gibbons in Dampa had declined, in four of the villages at least half of the interviewees were of the opinion that the population size had stabilized since gazettement as a Tiger Reserve in 1994.

Regarding the time period during which the decline took place, more interviewees pointed at a decline prior to gazettement of the area as a Tiger Reserve than after, with however, the majority of interviewees indicating a continuous decline (Table 1). No clear pattern emerged with respect to the spatial distribution of the decline with, for instance, no apparent differences between reports from villages in the two forest ranges (western and northern vs. south-east). While few interviewees were able to identify a single cause for the decline of gibbons in the reserve, the overall consensus was this was the combined result of fire (from *jhum* cultivation or other causes), reduction of the available habitat, and hunting, exacerbated by an increase in the human population.

The perceived decline of the gibbon population by the villagers living in the different parts of Dampa Tiger Reserve matched well with the recorded decline in gibbon habitat as

Table 1. Perceived changes in population sizes of hoolock gibbons (*Hoolock hoolock*) in seven villages in two forest ranges in Dampa Tiger Reserve (gazetted in 1994) between the late 1960s to the present. “Elder interviewees” were those born before 1965, and “All interviewees” included 10 interviewees born between 1966 and 1980.

Forest range	Village	Elder interviewees (43)		All interviewees (53)		
		(Elder, all)	Continuing decline	Early decline (<1996)	Late decline (>1996)	Stable since gazettelement
Teirei	Teirei (7, 7)		57	0	43	0
	Tuipuibari (10, 10)		40	50	10	50
	Damparengpui (7, 10)		85	15	0	30
Phuldungsei	Phuldungsei (7, 9)		85	0	33	0
	Kawnmawi (6, 6)		33	67	0	67
	Lallen (2, 6)		100	0	33	33
	Saithah (4, 6)		25	75	33	50

calculated from the vegetation maps. The extent of forest used by gibbons decreased from 63% of the mapped parts of the reserve in 1978 to 59% in 1989 to 50% in 2005.

Discussion

We recorded nine groups of western hoolock gibbons, totaling 31 individuals, in different sections of the Dampa Tiger Reserve. The only other population estimates available for the reserve were provided by Gupta and Sharma (2005) who, based on a 15-day survey mostly in the Teirei forest range, estimated Dampa to contain 10 groups, and by Molur *et al.* (2005) who estimated 20 individuals (about six groups) to be present. However, judging from Figure 1, which shows the combined total habitat available for gibbons in the reserve, and taking into account the extent of forest we actually managed to survey, we consider it more than likely that additional groups occur there. We expect the largest number of undetected groups to be present in the westernmost part of the reserve, close to the border with Bangladesh, as there the forest fragments are amongst the most remote and are not accessible by road.

We found no clear differences in terms of canopy continuity, distance to the nearest village or habitat disturbance levels between plots with and without gibbons. Indeed, while some of the groups inhabited some of the best forests in the reserve, others were found in the more disturbed sections, and conversely, we failed to detect gibbons in one part of the reserve where the forest appeared to be perfectly suitable for gibbons. We find this pattern consistent with a reduction in numbers (and local extinction) of gibbons due to habitat loss but especially hunting (either current or in the recent past).

We found a high degree of concordance between the decline in gibbon habitat recorded through satellite imagery (from 63% suitable for gibbons in 1978 to 50% in 2005, in a more or less linear fashion) and the decline in gibbon numbers as recalled by tribal villagers. Soliciting information from villagers can lead to a better understanding of the attitudes and perceptions of people towards biodiversity conservation including complex issues researchers may miss when conducting an ecological study. When interpreted with care, interview data can provide good insights regarding the (local)

status of threatened primates (Parker *et al.* 2008; Meijaard *et al.* 2011). Our approach was a general one, without giving precise time periods but referring to key events (civil unrest, establishment of the area as a Tiger Reserve) instead. Nor did we force interviewees to estimate numbers they have no ability or authority to estimate (see Asquith 2001). We purposely laid a greater emphasis on the elder villagers, as they indeed have experienced the decline first-hand, and by targeting seven villages from distinctly different parts of the reserve, we managed to capture the spatial component of the decline as well. A general consensus emerged that several decades ago, gibbons numbered over one-hundred individuals after which they experienced a decline that was either progressive until the present day, or that may have ceased with the gazettelement of the areas as a reserve. All interviewees agreed that at present the total gibbon population is less than 50 individuals. While this may be more a reflection of the gibbon population in the part of the reserve they were familiar with than true gibbon numbers, they do in fact match well with findings of the present study and that of Gupta and Sharma (2005). While local traditions of hunting and *jhum* cultivation are slowly fading, pressures in this regard are still felt especially from the southern parts of the reserve and from across the Bangladesh border. One of the positive outcomes of our study is that gibbons in Dampa are indeed able to persist outside the most pristine sections, and that a large number of villagers felt that the decline of western hoolock gibbons had ceased since the gazettelement of the area as a reserve.

Based on our study, we argue that future work with western and eastern hoolock gibbons needs to be interdisciplinary, focusing not only on the biology of the gibbons, but also addressing human-wildlife interactions, understanding the dependency of people on forests, and trying to charter conservation objectives. We do need more quantitative data on the dynamics of human disturbances, in particular hunting, as this may be fundamental to understanding the current distribution patterns and explaining the local extinction of gibbons in the region. In the absence of these data, we would argue against relying too much on community-based forest preservation initiatives as opposed to strict nature reserves, as here it is more challenging to control hunting.

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Identifying Habitat Connectivity for Isolated Populations of Lion-tailed Macaque (*Macaca silenus*) in Valparai Plateau, Western Ghats, India

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Abstract: The endangered and endemic lion-tailed macaques (*Macaca silenus*) of the rainforest fragments of Valparai plateau in the Western Ghats Hotspot (India) are facing serious threats to their survival due to anthropogenic pressures and habitat degradation. In this study, we identify potential wildlife corridors between the rainforest fragments and adjacent more extensive forest areas so as to connect isolated lion-tailed macaque populations. Satellite datasets were used to delineate the forest fragments and assess the conditions of the surrounding landscape. The corridors were selected on the basis of minimal impact on human settlements, agricultural areas and other infrastructure, as well as to enhance ecosystem services. The results show that a minimum area of 156 ha is required to connect three isolated lion-tailed macaque populations to the adjacent forest area. This includes 54 ha of seasonal stream beds (low human-use areas), 99 ha of cultivated area (medium human-use areas) and 3 ha of roads, settlements and built-up areas (high human-use areas). This methodology for identifying wildlife corridors in highly fragmented landscapes of the Western Ghats can also be applied to other human-dominated landscapes, including biodiversity hotspots.

Key Words: Habitat fragmentation, lion-tailed macaque, wildlife corridors, remote sensing, India.

Introduction

Habitat fragmentation is a serious concern in landscape ecology and conservation (Pimm and Raven 2000), influencing numerous ecological patterns and processes, and negatively affecting most taxonomic groups, including birds and mammals (Andr en 1994; Zuckerberg and Porter 2010), reptiles (Richard and Jean-Baptiste 2006), amphibians (Stuart *et al.* 2004), invertebrates (Leidner and Haddad 2011) and plants (Hobbs and Yates 2003). Studies show that the persistence of populations is lower in fragmented than in intact habitats (Tilman *et al.* 1994). Populations are susceptible to demographic extinction as well as environmental stress (Quinn and Hastings 1987). In addition, large-scale movements of species such as seasonal migration or range shifts in response to climate change may also be affected by habitat fragmentation (Soul e *et al.* 2004).

The Western Ghats is one of the eight major biodiversity hotspots (Myers *et al.* 2000). While less than 6% of India's landmass, more than 30% of all plant and vertebrate species of the Indian sub-continent are found there. The Western Ghats are also listed as one of the 200 globally most-important ecoregions (Olson and Dinerstein 1998). In spite of these

striking statistics, biodiversity in the Western Ghats is threatened. Menon and Bawa (1997) estimated that forest cover in the Western Ghats declined by 40% from 1920 to 1990, resulting in a four-fold increase in the number of fragments and an 83% reduction in the size of the remaining forest patches. This is unsurprising given that this region has the highest human population density of any of the biodiversity hotspots (Cincotta *et al.* 2000; Shi *et al.* 2005). The Valparai plateau in the southern region of the Western Ghats has undergone extensive fragmentation from the early 1900's because of forest clearance for tea, coffee, cardamom and eucalyptus plantations, and the associated infrastructural development (Joseph *et al.* 2009). Isolated remnants of rainforest in the middle of these plantations are known for their rich biodiversity. They harbour many endemic and endangered species including, for example, murid rodents and shrews (eight species), amphibians (40 species) and reptiles (40 species) (Umapathy and Kumar 2000).

The lion-tailed macaque (*Macaca silenus*) is an endangered primate endemic to the rainforests in the southern part of the Western Ghats. It has a scattered distribution of 49 subpopulations across eight locations (Molur *et al.* 2003; Kumara and Singh 2004). The total number of mature

lion-tailed macaques is estimated to be less than 2,500, with no subpopulation of more than 250 individuals (Kumar *et al.* 2008). It has been estimated that the species will experience a population decline of 20% over the next 25 years (Kumar *et al.* 2008). The lion-tailed macaque is listed in Appendix I of CITES, and in Schedule I, Part I, of the Indian Wildlife (Protection) Act, 1972. The rainforest fragments in the Valparai plateau harbour subpopulations that are on the verge of extinction due to demographic and environmental stochasticity, diseases, natural catastrophes, and inbreeding depression (Umapathy and Kumar 2000; Singh *et al.* 2001, 2009; Husain *et al.* 2013). Chapron *et al.* (2010) identified the preservation of the forests fragments of Valparai as one of the 100 top conservation priorities in Asia.

There are extensive studies on the ecology and behaviour of lion-tailed macaques in these rainforest fragments. Demographic studies have revealed reduced birth rates with correspondingly reduced numbers of immature individuals in the groups, an increase in the number of adult males, and considerable variation in group sizes and adult sex-ratios (Umapathy and Kumar 2000). Menon and Poirier (1996) pointed out that lion-tailed macaques in forest fragments spend more time ranging, and less time resting and feeding than is typical for groups in other regions. Forest-fragment size affects the demography, ranging patterns, feeding habits and reproductive rates of lion-tailed macaques (Kumar *et al.* 1995; Singh *et al.* 2001; Kumara and Singh, 2004). A recent study recommended the creation of dispersal corridors to facilitate male migration (Umapathi *et al.* 2011). Here we attempt to identify potential corridors between the fragments and the adjacent large forest area or nearby protected area.

Methods

Indian Remote Sensing Satellite data (IRS P6 LISS III and LISS IV) procured from the National Remote Sensing Centre was used to delineate the rainforest fragments in the Valparai plateau. LISS III images have a spatial resolution of 23.8 m, while for those of LISS IV it is 5.8 m. The LISS III data were geometrically corrected with respect to Enhanced Thematic Mapper (ETM+) satellite data based on 1st order polynomial regression between ground control points (RMSE<0.5 pixel) to compute the coefficients for two co-ordinate transformation equations, and registered to the UTM projection. Further to this, geometric correction of the LISS IV data was carried out with respect to LISS III data. The other data pre-processing techniques applied have been detailed elsewhere (Joseph *et al.* 2009, 2010, 2012). The rainforest fragments were delineated at a scale of 1: 10,000. Field surveys were conducted to find which isolated fragments harbored lion-tailed macaques. They were found in three, all owned by private enterprises in the Puthuthottam Estate, the Korangumudi Estate and the Tata Estate (hereafter the fragments are named with the name of the estate).

The following criteria were used in designing the corridors: 1) minimum impact on existing human settlements,

agricultural areas, and infrastructure such as motorable tarred roads, tea factories, and other built up areas; 2) favorable logistics for afforestation; 3) the provision of ecosystem services to the area; and 4) maximizing potential for harnessing the natural capacity of the areas to revegetate.

Considering the above factors, we favored relatively intact, seasonal stream beds and their riparian zones for the construction of wildlife corridors, anticipating that the land-owners would perceive the advantages of the preservation and provision of additional ecosystem services such as improvement in ecohydrology, water purification, biodiversity conservation (specifically freshwater biodiversity), and enhancement of carbon stock. Afforestation along the banks of streams is less expensive compared with other areas in the landscape.

The seasonal streams that connect the rainforest fragments with the adjacent main forested land were delineated. A 50-m buffer was generated over the delineated corridors, assuming that a 100-m wide corridor would be sufficient for dispersal. A strip of 100 m also minimizes impingement by agricultural land. Settlements, built-up areas, and other forms of land use by humans were identified in the strip to identify the least used and disturbed seasonal streams to connect the fragments.

Results

Lion-tailed macaques were found in three isolated rainforest fragments. 1. Puthuthottam fragment (123 ha) has evergreen trees in its upper storey and coffee plantations in its lower storey. A major road connecting the towns of Pollachi and Valparai passes through this fragment. The Puthuthottam fragment is surrounded by tea plantations, and Valparai is less than a kilometer from the fragment. The existing lion-tailed macaque population is highly exposed to human presence. 2. Tata Finlay fragment (78 ha) consists of relatively undisturbed evergreen forest, surrounded by tea and coffee plantations. 3. Korangumudi fragment (238 ha) is connected to another rainforest fragment (Pannimedu fragment) through a scattered chain of trees, and appears as a single fragment in the satellite data of 5.8-m resolution. It is surrounded by Upper Sholayar Reservoir on one side and extensive stretches of tea plantations on the other. The vegetation is relatively undisturbed evergreen forest.

Wildlife corridors: Puthuthottam fragment

There are two seasonal streams connecting the Puthuthottam fragment to the main forest area, the Indira Gandhi Wildlife Sanctuary (Fig. 1). The area statistics of delineated corridors along the streams are given in Table 1. The total area required for the first corridor is 38.82 ha while the second is 24.89 ha. In the first corridor, the cultivated area contributed 21.09 ha and motorable tarred roads 0.49 ha. There were no settlements and built up areas. In the second corridor, these land use categories represented 12.09 ha, 0.71 ha and 0.43 ha, respectively.

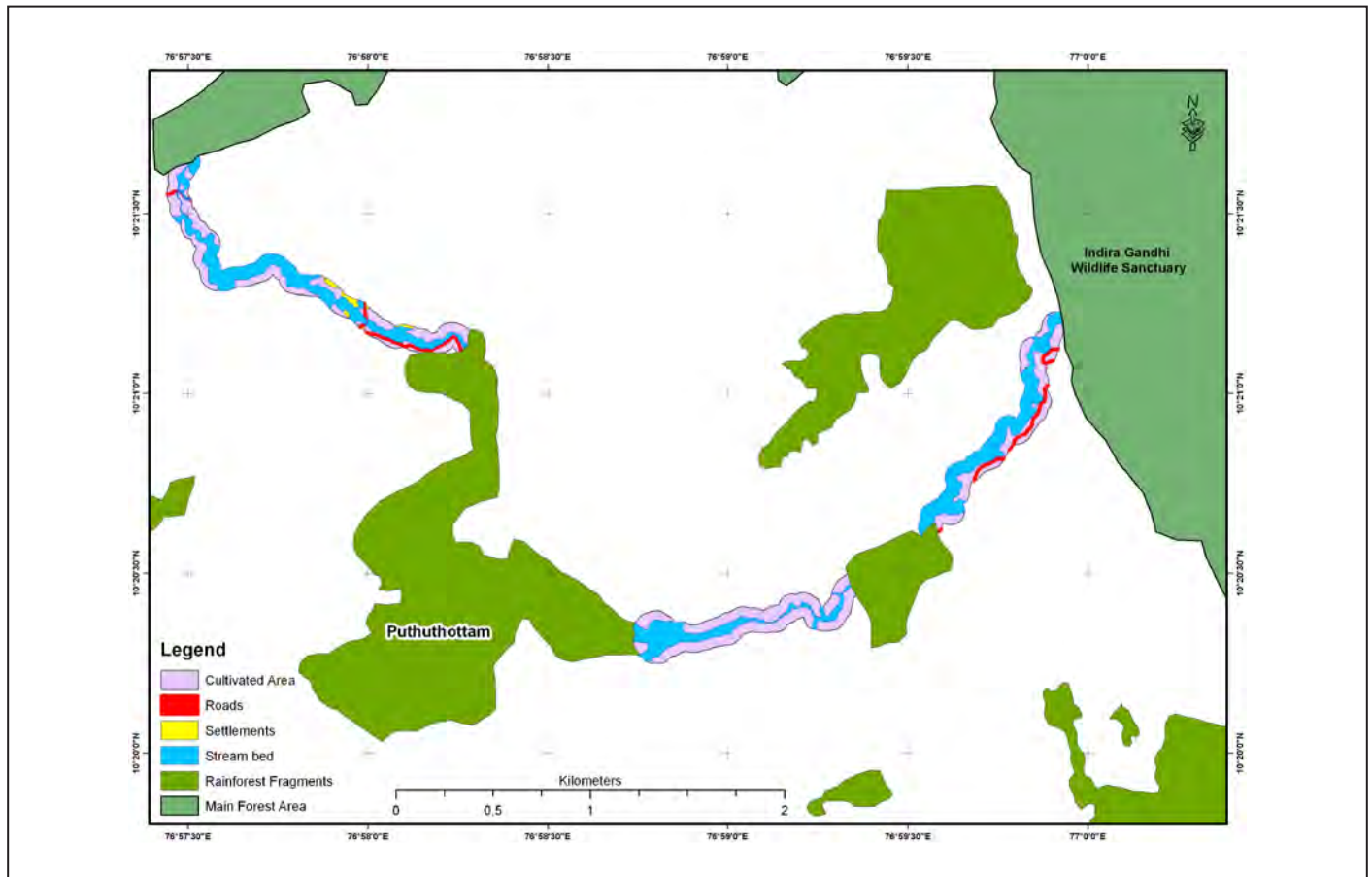


Figure 1. Potential wildlife corridors for the Puthuthottam rain forest fragment in the Valparai Plateau, Western Ghats, India.

Wildlife corridors: Tata Finlay fragment

A stream connects the Tata Finlay fragment with the Indira Gandhi Wildlife Sanctuary. The buffering and subsequent land-cover classification showed that an area of about 43.36 ha is required for the corridor, which includes 25.64 ha of cultivated land, 0.73 ha motorable roads, and 16.98 ha of stream bed (Table 2). There are no settlements or built-up areas in the assigned portion of the corridor (Fig. 2).

Wildlife corridors: Korangumudi Fragment

The Korangumudi fragment is almost in the center of the Valparai Plateau. A connection with the Indira Gandhi Wildlife sanctuary is not possible because of the Upper Sholayar Reservoir. The only way to connect this fragment with the main forest area is to link it with forest in the neighbouring state of Kerala. The area required for establishing this connection is 88.16 ha, which includes 61 ha of cultivated land, 0.51 ha of roads, 0.83 ha of settlements, and 25.82 ha of seasonal stream beds (see Fig. 3 and Table 3).

Discussion

Lion-tailed macaques are the most threatened of the primates endemic to the Western Ghats (Easa *et al.* 1997); habitat fragmentation is the primary reason (Singh *et al.* 2009).

Table 1. Area required for establishing the wildlife corridors between the Puthuthottam rain forest fragment and the Indira Gandhi Wildlife Sanctuary in the Western Ghats, India.

No.	Land use	Corridor 1 (ha)	Corridor 2 (ha)
1	Cultivated land	21.09	12.03
2	Motorable asphalt roads	0.49	0.43
3	Settlements and built-up areas	0.00	0.71
4	Seasonal streambed	17.24	11.72
	Total	38.82	24.89

Many studies carried out in the Valparai Plateau have highlighted that habitat fragmentation and the subsequent increase in human disturbance could lead to the local extinction of the species (Umapathy and Kumar 2000; Kumara and Singh 2004; Kumar *et al.* 2008; Hussain *et al.* 2013). Two practical solutions to avoid such biological insults are to connect the fragments through corridors or translocate the primates into other contiguous areas. The success rate of the latter method is found to be low (Marsh 2003), and therefore less preferred. We therefore explored the possibility of establishing conservation corridors between fragments and adjacent larger forest areas.

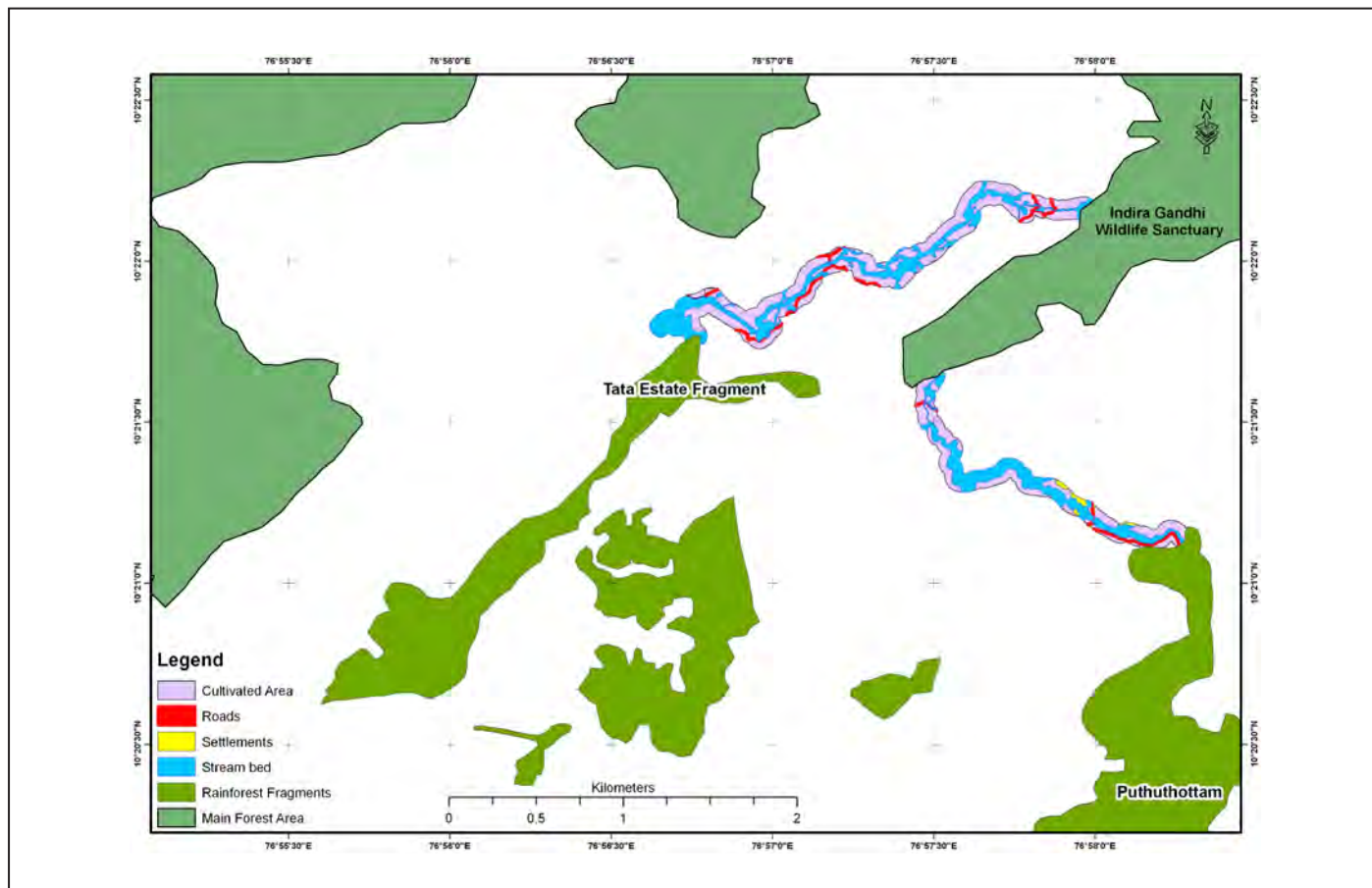


Figure 2. Potential wildlife corridors for the Tata Finlay Estate rainforest fragment in the Valparai Plateau, Western Ghats, India.

The next step we considered was to understand ways and means to acquire the land if we propose to establish a conservation corridor in private entities. The landscape is occupied mainly by plantations of tea, coffee, and cardamom, and the associated infrastructure (factories, settlements, motorable and muddy roads, community halls, religious places and small townships). Sharing land seems a challenging task in such areas, as seen from the recent increase in land disputes across India (Saikia 2011). Moreover, the overall terrain is highly undulating; elevations range from 550 to 1800 m, with intermittent steep slopes. We decided, therefore, to harness the utility of geological structures, and relatively less human-used areas such as the seasonal stream beds.

Our argument to use seasonal stream beds is valid in many ways. First of all there is no cultivation in the seasonal stream beds and their riparian zones, as they are saturated or filled with water at least half of the year. Revegetation of the riparian zones will enhance the water-holding capacity of the soil, and thereby increase water availability for agriculture. The riparian zones also act as water purifiers; the water is contaminated by the extensive use of pesticides and fertilizers in the plantations. Revegetation of riparian zones and the provision of better-quality water will further enhance the local biodiversity of the area, especially that of freshwater. The streams are part of the Chalakudy River basin, which harbors

Table 2. Area required for establishing a wildlife corridor between the Tata Finlay rain forest fragment and the Indira Gandhi Wildlife Sanctuary, Western Ghats, India.

No.	Land use	Corridor (ha)
1	Cultivated land	25.64
2	Motorable asphalt roads	0.73
3	Settlements and built-up areas	0.00
4	Seasonal streambed	16.98
	Total	43.36

an exceptional diversity of fishes, including several endemic and threatened species (Raghavan *et al.* 2008). Biophysically, the revegetation and afforestation programs could contribute to climate change mitigation by enhancing the carbon stock of the area (Joseph *et al.* 2012).

Though there are several such direct and indirect benefits, we have limited our proposal for revegetation to 50 m either side of the stream beds, in order to minimize pressure on cultivated land and other areas occupied by people. A marginally good portion of the land is available from the stream bed itself (47% in the case of Puthuthottam, 39% for Tata Finlay and 29% for Korangumudi). We have not carried

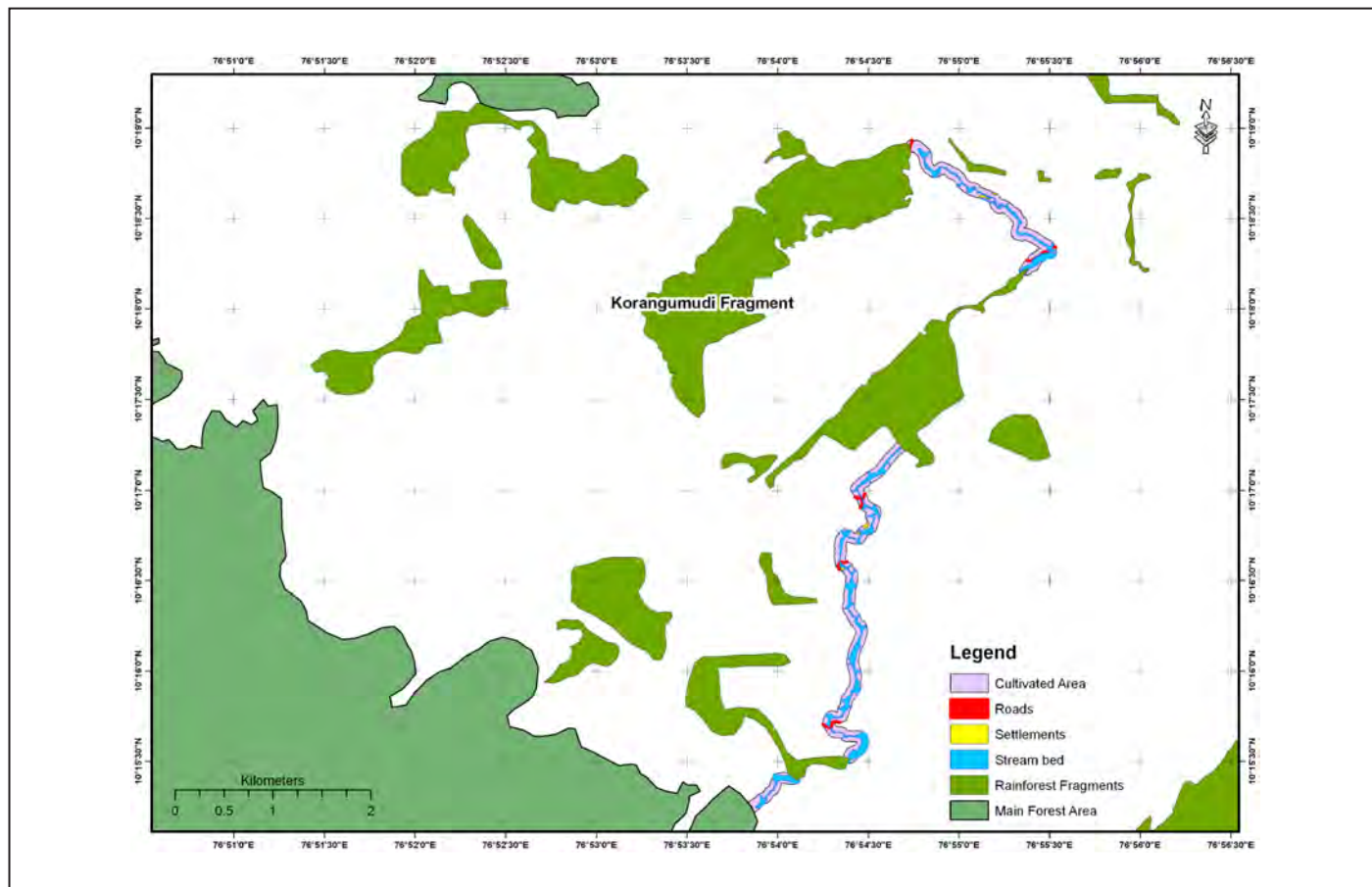


Figure 3. Potential wildlife corridors for Korangumudi rain forest fragment in the Valparai Plateau, Western Ghats, India.

out any simulation study to assess whether a 100-m stretch would be enough for lion-tailed macaques to disperse through these corridors. Our previous field experience, however, and the literature (Kumar *et al.* 2001; Joseph *et al.* 2010) reveal that lion-tailed macaque populations have undergone drastic behavioral changes in these fragments, and are not shy to human presence, unlike those observed in protected areas.

With respect to addressing the concerns of the various stakeholders, our studies showed that all the three fragments, especially Puthuthottam fragment, support the local communities existing around them in terms of provisioning firewood and other minor forest products (Kanagavel *et al.* 2013). Our previous study also indicated that the key stakeholders (plantation owners) may be unwilling to participate in legally-binding initiatives such as a Conservation Reserve on their leased lands, due to a perceived devolvement of power over their leased land that could curb further land use. Plantation enterprises such as Tata and Parry Agro, however, have already collaborated with local NGO’s to restore disturbed rainforest fragments (<<http://www.ncf-india.org/restoration>>). Hence informal, non-binding collaboration may be the way forward in this landscape, especially as some of the land owners are interested in supporting conservation. Further work is required to identify funding opportunities in the context of recent financial mechanisms such as REDD+

Table 3. Area required for establishing a wildlife corridor between the Korangumudi rain forest fragment with the main forest area in Kerala, Western Ghats, India.

No.	Land use	Corridor (ha)
1	Cultivated land	61.00
2	Motorable asphalt roads	0.51
3	Settlements and built-up areas	0.83
4	Seasonal streambed	25.82
	Total	88.16

(Reducing Emissions from Deforestation and Degradation) (Estrada and Joseph, 2012).

The present study has wider implications for global conservation efforts. Habitat fragmentation is a serious issue, happening on a day-to-day basis for numerous reasons with varying impacts from a minor change in habitat quality to local extinction. Our study used a simple methodology for corridor construction after considering the biophysical and socio-economic conditions of the landscape. The scope of the study was limited to geospatial analysis only as there was extensive information available on the ecology, behavior, distribution, and population size and structure of the lion-tailed

macaque groups in these forest fragments (Menon and Poirier 1996; Umaphathy and Kumar 2000; Kumar *et al.* 2001; Singh *et al.* 2001, 2002, 2009; Umaphathy *et al.* 2011; Hussain *et al.* 2013). We therefore project the present study as a model to highlight the implementation of simple conservation practices with minimal impact on existing conditions and the livelihood profiles of local communities.

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Food Habits and Habitat Use Patterns of Sri Lanka's Western Purple-faced Langur

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Abstract: Sri Lanka's western purple-faced langur (*Semnopithecus vetulus nestor*) is Critically Endangered, mainly because of habitat loss due to deforestation. Reforestation to expand the langur's natural habitat became feasible when the present government mandated the use of native plants to increase the country's forest cover. For reforestation to benefit langur populations, however, the re-created habitat needs to be similar to the natural forest that provides food and space for their survival. This monkey's diet and the manner in which it uses its natural habitat are, therefore, being investigated as the first step. The diet and habitat use patterns of two groups, Tikira and Appu, were studied for 13 and 14 months respectively (n = 1695 hours). Scan sampling (with ten-minute sample periods) was used to record all activities observed in the groups and the trees on which these activities were performed. The plant parts eaten were also noted. Our results showed that Tikira used more species than Appu to perform all of its daily activities. Additionally, while the Tikira group used *Dipterocarpus zeylanicus* most frequently during most months, the Appu group had six species occupying the top rank during different months. Of the ten most frequently used species, only five were common to both groups, and the frequency of use of these plants was sometimes quite variable as well. With respect to diet, Appu used at least 27 species while Tikira fed on more than 41. The top-ranking food plants of the two groups were different, and among the top ten only four were the same. The top fifteen food plants of both groups constituted over 85% of their feeding records. Nineteen species eaten by Appu and 29 eaten by Tikira were exploited for less than two months, and the two groups ate no more than five species for more than seven months of the study. Although the two groups relied on different plants for much of their nutrition, nearly 86% and 74% of feeding observations of Appu (n = 422) and Tikira (n = 685), respectively, were of them feeding on leaves. Blossoms, fruits and petioles made up the remainder of the groups' diets. While these items contributed variable amounts to the monthly diet of both groups, none was exploited more frequently than leaves. The above results are compared to information from other non-human primates, and discussed with respect to reforestation. Two points are emphasized. One is that the langur living in its natural habitat is a typical folivore, unlike those living around home gardens. The other is that while field research is essential to reforest degraded habitats it must be conducted in conjunction with conservation education and other initiatives that are designed to dissuade people from destroying restored and intact natural habitats.

Key words: Western purple-faced langur, *Semnopithecus vetulus*, diet, habitat, conservation, Sri Lanka

Introduction

Sri Lanka's western purple-faced langur (*Semnopithecus vetulus nestor*) has been listed among the 25 most endangered primates in the world since 2006 (Mittermeier *et al.* 2006, 2009, 2012). A field survey was conducted to investigate the cause of the langur's population decline (Rudran 2007). This survey and another study by Nahallage *et al.* (2008) indicated that the decline of this highly arboreal langur was mainly due to deforestation. Hence reforestation was evidently a logical

step to increase the extent of the langur's habitat and reverse its decline. This step was also in line with the current government's economic development policy, which mandated the planting of native species to increase Sri Lanka's forest cover from 27% to 36% (Rajapakse 2010; Yatawara 2011). Reforestation was therefore considered a feasible approach to help ensure the langur's future survival. For reforestation to benefit langur populations, however, the re-created habitat needs to be similar to the natural forest that provides the food and space for their survival, and here we report on a study of

the habitat use and diets of two groups—the first ecological study of the western purple-faced langur in its natural habitat.

Study Site and Methods

Our study site was located about 50 km southeast of Colombo, Sri Lanka's capital, in the most deforested region of the country (Fig. 1). The site was, however, relatively undisturbed because it was in the water catchment forest for two reservoirs crucial to the well-being of about one million residents of the capital. Besides being protected because of its function, this forest is the largest patch of undisturbed natural habitat (about 21 km²) occupied by the langur, and as such has the population with the best chance of survival over the long term in its highly fragmented range. We therefore decided to study this population's diet and habitat use patterns, in order to obtain a better understanding of its needs for plans to expand its natural habitat and enhance its long-term survival. For added security against deforestation, we established our study site in the Indikada Forest Reserve in the catchment forest, legally protected by Sri Lanka's Forest Department.

Our study site was close to a village called Waga (Fig. 1), and consisted of relatively flat areas and gently undulating terrain where dense-canopied trees rose to heights of about 40 m. These habitat conditions made focal animal sampling

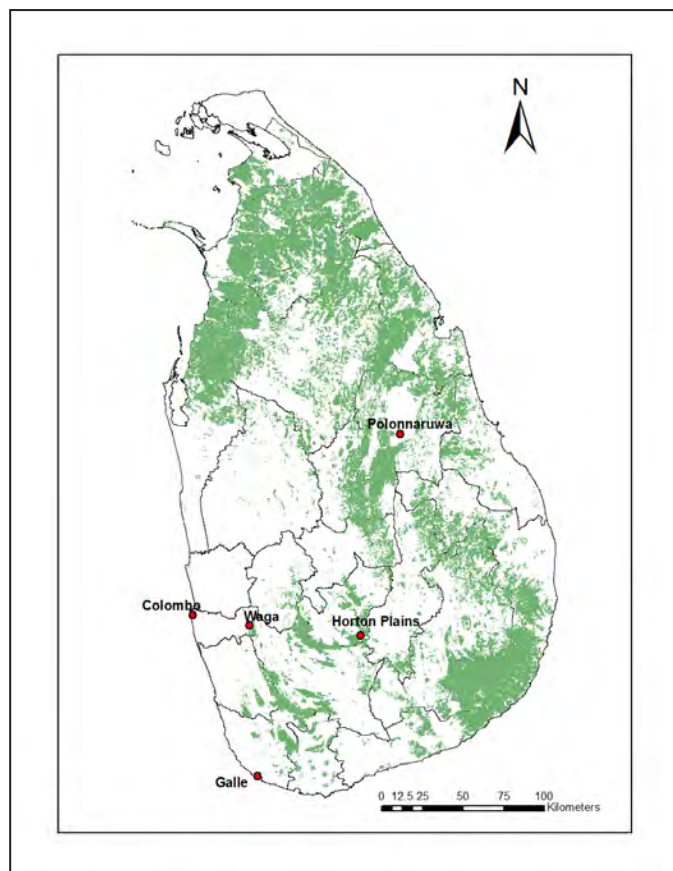


Figure 1. Forest cover of Sri Lanka (2010) showing extensive deforestation in the west, which includes the range of the western purple-faced langur (*Semnopithecus vetulus nestor*). Courtesy of V.A.P. Samarawickrama.

unfeasible, requiring as it does relatively long unbroken periods of observations (Altmann 1974). We, therefore, used scan sampling to collect data on the langur's diet and patterns of habitat use. The data were collected from two habituated groups that lived in adjacent home ranges. One group, named Tikira, consisted of eight members and occupied a home range on undulating terrain. The other, named Appu, was made up of seven individuals, and ranged over a relatively flat area.

Data from the two groups were collected for three to five days each month between June 2009 and December 2010. Daily observations usually lasted for 8–12 hours but were shorter when the project's conservation-related community activities demanded attention. Data on diet and habitat use patterns were collected during ten-minute sample periods separated by five-minute breaks (Rudran 1978; Struhsaker 1975). During each sample period the first activity performed for at least five seconds by a group member visible to the observer was recorded, along with the tree used to perform that activity. Each group member was sampled for activity only once during a sample period, and if feeding was observed the plant parts eaten were recorded as well. Each sample period started and ended at fixed times during each hour of observation to enable direct comparisons of activity data collected on different days of field work.

The permit we received from the Forest Department to work in its reserve specifically prohibited us from collecting plant specimens. We therefore could not use a herbarium to confirm the identity of plants used by the study groups, and were limited to assigning scientific names only to those that were very familiar to us. Unfamiliar species and even doubtful ones used by the study groups were assigned field names, some of which have been used in this paper.

Results

During our 18-month study, the Tikira group was observed for 793 hours over 13 months and the Appu group for 902 hours over 14 months (Table 1). Delays in the renewal of our reserve entry permits meant that we were unable to observe them in certain months.

Habitat use

The groups used a minimum of 69 plant species during their daily activities—feeding, resting, moving, and social interactions with other group members—but neither used all of them. Appu (observed for a longer period than Tikira) used 36 species, while Tikira used 52 (Table 2). Seventeen species used by Appu (47%) were not used by Tikira, and 33 species used by Tikira (62%) were not used by Appu. Only 19 species were used by both groups. Two species were cultivated varieties—*Hevea brasiliensis* (cultivated rubber) and *Pinus* sp. Appu used both, but they accounted for much less than 1% of the group's habitat use records ($n = 3527$). Tikira used *Hevea brasiliensis*, which contributed about 3% to the group's total habitat use records ($n = 3795$).

Table 1. Durations of monthly observations of study groups.

Month	Appu group	Tikira group	Total (hrs)
June 2009	67	-	67
July 2009	91	90	181
August 2009	75	63	137
September 2009	157	-	157
November 2009	103	109	212
December 2009	106	110	216
January 2010	37	67	104
February 2010	48	48	96
May 2010	-	37	37
June 2010	24	49	73
July 2010	73	73	146
August 2010	48	49	97
September 2010	48	35	83
October 2010	14	30	45
December 2010	11	33	44
Total	902	793	1695

Just one species, *Dipterocarpus zeylanicus*, was exploited during all months of observations (Tables 3 and 4). This species was Tikira's most frequently used plant in 12 of 13 months, but was top ranking in use by Appu in only five of 14 months. While Tikira concentrated on a single species during most months, Appu favored the use of six species in the different months (Table 3). Of the 10 species most frequently used by the two groups, only five were common to both (Table 2). The frequency of use of these five species also varied, sometimes quite appreciably. The other five species either occupied ranks below ten or were used only by one group (for example, *Mangifera zeylanica*).

Since *Dipterocarpus zeylanicus* was a dietary item in all months of the study, its total frequency of use by both groups was higher than for any other species. Its use constituted, however, only 14% of Appu's habitat use records, but nearly 41% of Tikira's (Table 2). While the two groups differed substantially in their use of one species, the collective use of their fifteen most frequently exploited species did not differ by much; they represented 92% and 89% of the habitat use records obtained from Appu and Tikira, respectively

Table 2. Intergroup comparisons of habitat use of two groups of the western purple-faced langur (*Semnopithecus vetulus nestor*).

Comparison	Appu group	Tikira group
# months of observations	14	13
# habitat use records	3527	3795
# species used	>36	>52
# species used only by one group	17	33
# species used by both groups	19	19
Use of species (% of records)		
Rank #1	<i>Dipterocarpus zeylanicus</i> (14.2)*	<i>Dipterocarpus zeylanicus</i> (40.5)*
Rank #2	<i>Litsea decanensis</i> (13.0) T	<i>Stemonurus apicalis</i> (9.3)*
Rank #3	<i>Albizia lebbek</i> (10.6) T	RBSL (5.6) X**
Rank #4	<i>Alstonia macrophylla</i> (10.4) T	<i>Mangifera zeylanica</i> (5.3) X
Rank #5	<i>Bridelia retusa</i> (8.9) T	<i>Persea macrantha</i> (5.0) A
Rank #6	<i>Melia azedarach</i> (7.6)*	<i>Melia azedarach</i> (3.7)*
Rank #7	<i>Artocarpus nobilis</i> (4.8)*	<i>Artocarpus nobilis</i> (3.7)*
Rank #8	<i>Dillenia retusa</i> (4.5) T	UI (3.4) A**
Rank #9	<i>Stemonurus apicalis</i> (3.0)*	<i>Hevea brasiliensis</i> (3.1) A
Rank #10	<i>Horsfeldia iryagedhi</i> (2.9)	<i>Bridelia retusa</i> (2.8) A
% use of top five species	57	64
% use of top ten species	80	82
% use of top fifteen species	92	89
Use of species (# months)		
≤2 months	14	22
≤7 months	24	41
≥8 months	12	11

* species used by both groups

** Field name and number. Species not identified.

(T) species used by Appu and also found in Tikira range

(A) species used by Tikira and also found in Appu range

(X) species not used by the other group

(Table 2). Nearly 39% and 42% of all species used by Appu and Tikira, respectively, were exploited for two months or less. Only 33% of the species in the diet of Appu and 21% of those in the diet of Tikira were used for more than eight months of the study. Hence, both groups used an appreciable number of species only for short periods.

Diet

At least 27 species were exploited by Appu, while Tikira fed on more than 41 (Table 5). Tikira's diet included 22 species that were absent from that of the Appu group, while eight species in the diet of Appu were absent from that of Tikira. Thus both groups exploited a minimum of 49 species for food. Just one of these was a cultivated plant, *Hevea brasiliensis*, which was eaten only by the Tikira group. During the study, 382 and 567 feeding records were collected from Appu and Tikira, respectively. Appu fed most frequently on *Albizia lebbek* (35.8% of feeding records) and Tikira on *D. zeylanicus* (22.6% of feeding records). Intergroup dietary differences were also evident in the top ten species used for food (Table 5). Only four of these were common to both groups, and sometimes their frequency of use was quite variable as well.

The top fifteen food plants of Appu and Tikira constituted about 95% and 86% respectively of the feeding records obtained from them. Nineteen of the 27 species eaten by Appu and 29 of the 41 species eaten by Tikira were included in the diet for less than two months. Similarly, only four species in Appu's diet and five species in Tikira's were exploited for more than seven months. Similar to the patterns found in their habitat use patterns, both groups relied on relatively few but different species for most of their nutritional requirements.

Although the two groups relied on different food plants for much of their nutrition, nearly 85% and 74% of the observations from Appu (n = 422) and Tikira (n = 685), respectively, were feeding on leaves (Tables 6 and 7). Blossoms, fruits and petioles made up the remainder of the diet of both groups. These items

contributed variable amounts to the monthly diet of both groups, but none were exploited more frequently than leaves.



Figure 2. Adult female western purple-faced langur (*Semnopithecus vetulus nestor*). Photo by N. L. Dhangampola.

Table 3. Western purple-faced langur (*Semnopithecus vetulus nestor*) – Appu group. Monthly frequency of use of top ten species for all activities.

Species	2009						2010						Total		
	Jun	Jul	Aug	Sep	Nov	Dec	Jan	Feb	Jun	Jul	Aug	Sep		Oct	Dec
<i>Dipterocarpus zeylanicus</i>	18	34	33	32	27	68	58	36	22	54	42	2	50	26	502
<i>Litsea decanensis</i>	27	38	35	30	15	48	21	17		84	9	35	26	75	460
<i>Albizia lebbek</i>	21	40	44	25	26	67	6	24	7	31	3	31	32	17	374
<i>Alstonia macrophylla</i>	27	79	43	23	20	51	35	12		31	18	9	12	8	368
<i>Bridelia retusa</i>		9	31	39	22	28	81	16	10	31	6	1	22	18	314
<i>Melia azaderach</i>	16	48	13	20	11	37	31	6	20	10	19			37	268
<i>Artocarpus nobilis</i>	0	18		4	15	16	17	12	32	9	4	1	13	27	168
<i>Dillenia retusa</i>	11	37	21	3	3	17	16	7		20		22			157
<i>Stemonurus apicalis</i>							34	28	7	22	4	5	3	4	107
<i>Horsfeldia iryaghedhi</i>							23	23	16	14	20	8			104

Highest monthly frequency of use each month is in **bold**.

Diversity of diet and habitat use

To compare monthly variations in the diversity of diet and habitat use of the two groups we used the Shannon index (Lloyd and Ghelardi 1964; Pielou 1966), which is given as:

$$H = -\sum p_i \log p_i$$

where p_i is the proportion (n_i/N) of the i th species used

by a group during a particular month. We preferred this index to species richness measures or other diversity indices (for example, Menhinick 1964) because it takes into account the number of species used by a group each month as well as their individual frequencies of use, and produces a single value to compare diet or habitat use diversity of the two groups. Indices calculated for twelve of the fifteen months of our study provided such comparisons (Table 8).

Table 4. Western purple-faced langur (*Semnopithecus vetulus nestor*) – Tikira group. Monthly frequency of use of top ten species for all activities.

Species	2009				2010									Total
	Jul	Aug	Nov	Dec	Jan	Feb	May	Jun	Jul	Aug	Sep	Oct	Dec	
<i>Dipterocarpus zeylanicus</i>	30	111	226	304	189	41	22	88	170	56	95	80	126	1538
<i>Stemonurus apicalis</i>			46	61	63	3		10	66	21	29	21	34	354
RBSL*			34	9	15	9		1	47		3	53	42	213
<i>Mangifera indica</i>				39	22	10		10	29	8	6	14	63	201
<i>Persea macrantha</i>			1	32	5	4			51	19	6	32	41	191
<i>Artocarpus nobilis</i>	3	5	12	22	14	1	33	5	13	1	8	6	20	143
<i>Melia azedarach</i>	30		14	26	16	3		2	1	4	9	27	10	142
UI*			105	14		7		3	3					132
<i>Hevea brasiliensis</i>			14	28	31	1		4	17	2	6	12	1	116
<i>Bridelia retusa</i>	1		14		11		7		4	15	2	15	38	107

Highest monthly frequency of use is listed in **bold numbers**

*Species not identified.

Table 5. Intergroup comparisons of food habits of two groups of the western purple-faced langur (*Semnopithecus vetulus nestor*).

Comparison	Appu group	Tikira group
# diet records	382	567
# species used	27	41
# species used only by one group	8	22
# species used by both groups	19	19
Use of species (% records)		
Rank #1	<i>Albizia lebbek</i> (35.8)T	<i>Dipterocarpus zeylanicus</i> (22.6)*
Rank #2	<i>Pothos scandens</i> (12.7) T	<i>Stemonurus apicalis</i> (12.3)*
Rank #3	<i>Litsea decanensis</i> (10.2) T	<i>Pothos scandens</i> (8.5) A
Rank #4	3-leaf vine (6.5)T	<i>Hevea brasiliensis</i> (8.0) A
Rank #5	<i>Artocarpus nobilis</i> (5.2)*	<i>Persea macrantha</i> (7.6)A
Rank #6	<i>Dipterocarpus zeylanicus</i> (3.4)*	<i>Mangifera zeylanica</i> (7.1) X
Rank #7	<i>Alstonia macrophylla</i> (2.6) T	RBSL (5.0) X**
Rank #8	<i>Hopea-L</i> (1.8) X**	<i>Artocarpus nobilis</i> (3.7)*
Rank #9	<i>Bridelia retusa</i> (1.8) T	<i>Melia azedarach</i> (4.3)*
Rank #10	<i>Melia azedarach</i> (1.3)*	<i>Bridelia retusa</i> (2.3) A
% use of top five species	78	59
% use of top ten species	89	80
%use of top fifteen species	95	86
Use of species (# months)		
≤2months	19	29
≤7 months	23	35
≥8 months	4 (ranks 1–4)	5 (ranks 1–3, 6 and 7)

*species consumed by both groups

(T) species eaten by Appu and also found in Tikira range
(X) species not used by the other group

** Field name and number. Species not identified.

(A) species eaten by Tikira and also found in Appu range

The monthly values showed that Appu had lower diet diversity indices than Tikira for 10 of the 12 months of the study. In contrast, it had higher diversity indices for habitat use than Tikira during all twelve months. These results suggest that Tikira's monthly diet was consistently more diverse than Appu's because it exploited more species as food. On the other hand, Appu probably distributed its use of different species in the habitat more equitably, and therefore, had higher monthly values for habitat use diversity than Tikira.

Table 6. Monthly frequency of use of food items; western purple-faced langur groups (*Semnopithecus vetulus nestor*) – Appu group.

Month	Leaves	Blossom	Fruit	Petiole	Seed	UI*	Total
June	22				8	2	32
July	32		1			3	36
August	11					13	24
September	11					1	12
November	18	1				3	22
December	44		1	2		4	51
January	7						7
February	30		5				35
June	9						9
July	24						24
August	51						51
September	22	6					28
October	32		7	1			40
Dec	40						40
Total	360	7	14	3	8	30	422

*Unidentified items

Table 7. Monthly frequency of use of food items; western purple-faced langur groups (*Semnopithecus vetulus nestor*) – Tikira group.

Month	Leaves	bl	Fruits	Petiole	Seed	UI*	Total
July	4		1			2	7
August	19		1			1	21
November	65	30	1			1	97
December	118	1	23			1	143
January	50	14		6			70
February	13		8				21
May	11	1		3			15
June	10	2		2			14
July	54	7		8			69
August	10						10
September	22			1			23
October	23	1	3	5			32
November	65	21	10			1	97
December	37	3	5	13			58
Total	509	80	52	38		6	685

*Unidentified items

Discussion

Similarities between the groups

Despite numerous differences in diet and habitat use patterns, the two groups were similar in many ways. They maintained a highly folivorous diet as is typical of colobines (Horwich 1972; Oates 1977; Hladik 1978; Stanford 1988; Gupta and Kumar 1994; Saj and Sicotte 2007; Struhsaker 2010; Choudhury 2012; Vandercone *et al.* 2012). The proportion of leaves in the monthly diet of our study groups was quite similar to that in the diet of two other subspecies of purple-faced langurs (*S. v. philbricki* and *S. v. monticola*) that were studied in the dry zone forests of Polonnaruwa and the cloud forests of Horton Plains (Fig. 1). In the dry zone forest the average monthly diet of purple-faced langurs consisted of 53% leaves, while in the cloud forest leaves contributed nearly 80% (Rudran 1970, 2012).

The folivorous diet of the langurs in natural habitats is in marked contrast to that of groups living around human habitations and rubber plantations, which have been found to rely mainly on cultivated (human edible) fruits (Dela 2007). The extensive exploitation of cultivated fruits has been interpreted to mean that these langurs are adapting to changing environmental conditions and preferentially selecting and feeding on fruits rather than leaves (Dela 2007, 2012; but see Setchell 2012). As a result, it was recommended that the langur's dietary switch be considered when formulating effective action for its conservation. There are several reasons why we feel this recommendation is untenable.

First, like other colobines, purple-faced langurs have evolved numerous adaptations over several millennia to satisfy their nutritional requirements mainly through a leafy diet. For instance, they harbor numerous symbiotic bacteria in the stomach to ferment the structural carbohydrates in leaves; and the end products become the langur's primary source of energy (Bauchop and Martucci 1968; Bauchop 1975). Second, the stomach is large and sacculated (Hill 1934) to reduce the speed at which it fills up with food and the rate at which the ingesta moves out. The slow passage of ingesta out of the stomach increases the time available for microbial action (Milton 1999). Third, to further improve bacterial action and fermentation efficiency, the langurs rest for long periods between feeding bouts; a behavior characteristic of other colobines (Struhsaker 1975; Oates 1987). Fourth, the symbiotic bacteria can also convert the host's urea into microbial protein, and contribute a valuable supplement to protein derived from leaves. Fifth, bacterial action on the ingesta leads to manifold increases in vitamins that makes the langurs virtually independent of dietary sources of all vitamins except A and D (Bauchop 1975). These morphological, kinetic, physiological and behavioral adaptations clearly show that langurs have evolved highly specialized traits to exploit a leafy diet for their energetic and nutritional requirements.

Langurs do, of course, eat fruits, but the amount consumed in the wild is much less than around home gardens. Cultivated fruits are generally lower in protein, fiber and mineral content

than wild fruits (Milton 1999), and are unlikely to provide the langurs with adequate nutrition over the long term. Hence groups that rely on cultivated fruits for extended periods may run the risk of dying of malnutrition (if they do not meet their end before then, through other outcomes of human-monkey conflicts such as electrocution, attacks by village dogs, or parasitic infestations; Ekanayake *et al.* 2006; Rudran 2007; De Silva *et al.* 2012). Furthermore, Nijman (2012) analyzed Dela's (2012) selection ratios and found that there was no statistically significant basis for the claim that the langur was selecting cultivated fruits over leaves. In fact, he showed that some of the plants with the highest selection ratios were used mainly for their leaves.

It is unreasonable, therefore, to assume that the langur is adapting to environmental changes by switching its diet to cultivated fruits and recommend that its conservation be based on this assumption. Nevertheless, this recommendation is already being mentioned by others as a strategy for langur conservation (De Silva *et al.* 2012). Before this notion gains further traction we hope the information in this paper will convince local conservationists to think differently.

Our findings have also indicated that both groups relied on relatively few species for much of their nutritional requirements (Table 5). This feeding pattern is quite widespread among colobines (Hladik 1978; Gupta and Kumar 1994; Struhsaker 2010; Vandercone 2012) and other non-human primates (Rudran 1978; Miller 1998; Watts *et al.* 2012); and likely the result of intergroup differences in food species selection and food plant density differences between home ranges. Furthermore, long-term studies on the red colobus of Kibale National Park have shown that the species most frequently exploited for food can vary between years (Struhsaker 2010) because of naturally occurring events such as tree

regeneration or mortality resulting from disease. This suggests that monkeys are to some extent capable of adapting to changes in their natural environment.

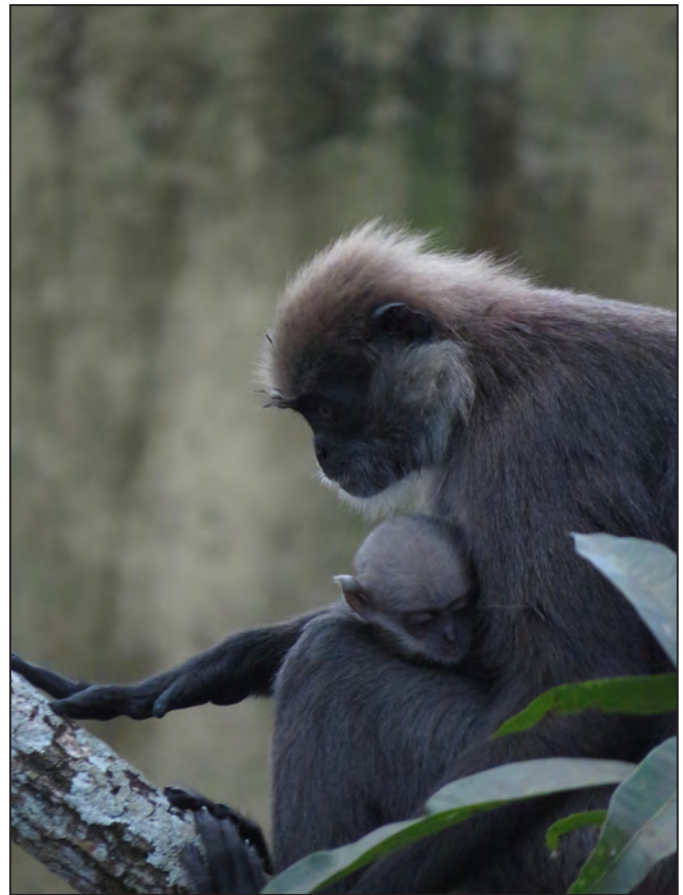


Figure 3. Mother and offspring western purple-faced langur (*Semnopithecus vetulus nestor*). Photo by N. L. Dhangampola.

Table 8. Intergroup comparisons of diet and habitat-use diversity in two western purple-faced langur groups (*Semnopithecus vetulus nestor*).

Month	Appu group		Tikira group	
	Habitat use diversity	Diet diversity	Habitat use diversity	Diet diversity
June 2009	0.916	0.405		
July 2009	0.855	0.515	0.702	0.959
August 2009	0.909	0.444	0.405	0.217
September 2009	0.910	0.747		
November 2009	0.900	0.603	0.617	0.831
December 2009	0.855	0.904	0.671	0.849
January 2010	0.834	0.555	0.659	0.883
February 2010	0.921	0.832	0.763	0.867
May 2010			0.928	0.840
June 2010	0.913	0.620	0.650	0.929
July 2010	0.850	0.564	0.714	0.937
August 2010	0.883	0.560	0.747	0.881
September 2010	0.795	0.573	0.668	0.829
October 2010	0.825	0.845	0.771	0.931
December 2010	0.864	0.734	0.737	0.903

While the groups obtained most of their nutritional requirements from a few species, they also exploited an appreciable number at low frequencies. The additional food intake from several plants may have satisfied a group's total energy and nutritional needs (Struhsaker 2010). However, Freeland and Janzen (1974) have suggested that infrequent feeding on a large number of species helps folivores to maintain metabolic pathways for detoxifying secondary compounds found in plant material. Keeping these pathways open may have been necessary for langurs to exploit alternative food plants without suffering any ill-effects, when food from its most frequently exploited species are in short supply.

Another point related to the langur's heavy dependence on just a few species for feeding and other activities is that only a small number of species may be needed to re-create forests that are optimal for its survival. Detailed investigations of habitat variables (for example, species composition, density and plant phenology), however must be conducted before final decisions could be made about the species most suitable for reforestation. If these investigations are conducted, they would help ensure that reforested areas have adequate amounts of food and space throughout the year for the langurs to thrive.

Differences between the groups

The differences we found are remarkable because the two groups lived in adjacent home ranges. Despite the close proximity of home ranges, data collection in Appu's home range was considerably more difficult than in Tikira's. This was because unlike Tikira's home range, that of Appu was located on relatively flat ground where collecting data by looking straight up into the dark and dense canopies often proved difficult. However, we do not believe that the intergroup differences documented in our study were the result of observation conditions, because such differences have also been found in other field studies of non-human primates. For instance, dietary differences between groups living in adjacent home ranges or close proximity have been reported in capuchin monkeys (*Cebus capucinus*) of Costa Rica (Chapman and Fedigan 1990) and blue monkeys (*Cercopithecus mitis stuhlmanni*), red colobus (*Piliocolobus rufomitratus tephrosceles*), and chimpanzees (*Pan troglodytes schweinfurthii*) in Kibale National Park, Uganda (Rudran 1978; Chapman and Chapman 1999; Struhsaker 2010; Watts *et al.* 2012).

Several reasons have been proposed to explain the above mentioned differences. Fairgrieve and Muhumuza (2003) indicated that dietary differences between blue monkey groups inhabiting Budongo Forest Reserve, Uganda, were the result of logging. This could not have been the case at our study site with its long history of habitat stability. In their study of *Cebus capucinus*, Chapman and Fedigan (1990) asked if intergroup dietary differences were the result of differences in food abundance between home ranges, and found no evidence for it. They were also unable to determine if group specific diets were due to intergroup differences in foraging strategies (Schoener 1971) or the result of group specific

traditions (McGrew 1983). Nevertheless, Perry (2011) argued that intergroup differences in foraging in the Costa Rican *Cebus capucinus* were the result of group specific social traditions. Struhsaker (2010), on the other hand, showed that intergroup dietary differences in red colobus monkeys in Kibale National Park, Uganda, were the result of differences in tree species composition between sites and also due to the extent to which groups fed selectively on different species. These differences could also have arisen from intraspecific differences in nutrient content of plants growing in different home ranges (Chapman *et al.* 2003). It is possible that plant density differences between home ranges (habitat heterogeneity) and selective feeding are the underlying reasons for differences in foraging strategies and social traditions that ultimately lead to group specific diets.

Although we were unable to determine the exact reason for group specific diets in these langurs, the fact that they were real, presented a novel way of relating the langur's lifestyle to that of local human communities, where dietary differences between neighbors were quite common. We drew similarities between human families and langur groups with respect to their food habits and composition of social units, to create public empathy for the endangered folivore and discourage the destruction of its natural habitat (Batahira Kaluwandura 2011). In this manner, our field research became an invaluable tool to promote public awareness of the precariousness of the langur's future.

Promoting public awareness of the langur's plight has been an important component since the project's inception. It included workshops to identify the critical needs of the community's adults, which turned out to be focused on employment opportunities, improvement of health services, and the need for vocational training. To address the need for employment opportunities, a home gardening program was launched (Anonymous 2011) to help augment household income and improve nutrition. This program also gave villagers opportunities to grow seedlings of plants important to the langur in backyard nurseries, to satisfy future reforestation needs, and to take pride in helping to conserve the endangered folivore. The other two needs of adults were addressed through an eye-care clinic, and training in making cloth bags for sale to locals and tourists. Activities for young people have included conservation-oriented classroom lectures, nature walks, competitions, and public exhibitions of children's artwork and essays. Because of these activities the local people now view us as people who are not only concerned about monkeys but also genuinely interested in their welfare. We hope this change in attitude will help garner support from local communities to protect the langurs over the long term.

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Of Concern Yet? Distribution and Conservation Status of the Bonnet Macaque (*Macaca radiata*) in Goa, India

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Abstract: The bonnet macaque *Macaca radiata*, endemic to peninsular India, is typically ignored in conservation initiatives as it is considered a common species, ubiquitously present across its distribution. Recent studies in southern India, however, report drastic declines in its populations. From March to May, 2012, we carried out a study to investigate its conservation status at the northern end of its range, in the state of Goa on the western coast of India. We recorded bonnet macaques in less than 40% of the locations we visited where they had previously been reported, and found that local residents living in close association with bonnet macaque populations were largely intolerant of the species' presence. Surveys and studies to monitor the population dynamics of such 'common' species as the bonnet macaque are urgently needed so that we can be better informed about their actual conservation status.

Key Words: Bonnet macaque, India, survey, distribution, human-primate conflict, conservation

Introduction

Primate species that are characterized as 'common' on account of their wide distribution and ability to adapt to a range of habitats occupy the lowest position on the totem pole of primate conservation. Since conservation efforts by necessity are based on prioritization, this is understandable. However as Eudey (2008) warned in her assessment of the conservation status of *Macaca fascicularis*, excluding common primate species from protection initiatives has proved detrimental to their future. Although these species are highly adaptable, they are not neutral to various anthropogenic pressures such as habitat degradation, forest fragmentation and poaching. In addition, perceptions regarding their widespread distribution provide impetus for their indiscriminate use in biomedical research and commercial trade, further endangering their existence (Molur *et al.* 2003; Eudey 2008; Radhakrishna and Sinha 2011).

Of the eight macaques found in India, the rhesus macaque *Macaca mulatta*, the bonnet macaque *M. radiata* and the long-tailed macaque *M. fascicularis* are categorized as Least Concern as they are presumed to be widespread, tolerant to a range of habitats and found in large populations (IUCN 2012). Recently however, concerns have been raised regarding the

conservation status of the bonnet macaque in southern India (Kumar *et al.* 2011; Singh and Rao 2004; Singh *et al.* 2011). Studies monitoring demographic changes in their populations in southern India between 1989 and 2009 showed drastic reductions in both the number of groups (54 to 31) as well as the number of individuals (1207 to 697) (D'Souza and Singh 1992; Sharma 1998; Singh and Rao 2004; Singh *et al.* 2011). The bonnet macaque is also reported to be locally extinct in many regions in southern India (Kumara *et al.* 2010a). These studies reiterate the importance of collecting even baseline data on the present distribution and populations of the species in order to obtain a more accurate picture of its current conservation status.

The geographic range of the bonnet macaque extends across peninsular India, but studies on its ecology and distribution have largely focused on the species at the southern end. There have been surveys to assess the northern distributional limit of the bonnet macaque (Fooden 1981; Koyama and Shekar 1981; Kumar *et al.* 2011), but there has been no systematic attempt to study the distribution of the bonnet macaque in the northern part of its range (Sinha 2001). As a start, the aim of our study was, therefore, to obtain an understanding of the distribution of *M. radiata* in the state of Goa, towards the northern end of its distributional range. Trapping

and hunting of bonnet macaques as retaliatory measures against macaque crop-raiding has been a major factor for the decline of bonnet populations in India (Kumara *et al.* 2010a; Singh and Rao 2004). People's tolerance for crop-raiding and attitudes towards bonnet macaques therefore crucially determine the conservation status of the species. Hence an important secondary aim of our study was to document the attitudes and tolerance levels of the local residents towards *Macaca radiata* with the view that such knowledge aids in the appraisal of the actual human-macaque conflict situation (Nahallage *et al.* 2008), and thereby in formulating management strategies that aid the conservation of the species.

Methods

We carried out our study in the state of Goa on the western coast of India (14°53'N to 15°40'N and 73°40'E to 74°21'E). Goa has 33.06% of its area under forest cover (Forest Survey of India 2011). Of this, 20.67% and 69.04% are classified as reserve forests and protected areas, respectively (forests under the jurisdiction of the State Forest Department), and 10.29% of the forest cover is considered “unclassified” (not under the aegis of the State Forest Department). The Protected Area network includes a national park (Mollem National Park) and six wildlife sanctuaries (Mhadei, Bhagwan Mahaveer, Netravali, Cotigao, Bondla, and Salim Ali) (FSI 2011).

We conducted surveys in both the districts of the state—North and South Goa—from March to May, 2012. Bonnet macaques occupy both forest and anthropogenic habitats, and we surveyed protected areas, privately owned plantations and estates, and roadsides. The protected areas included Mollem National Park, and Mhadei, Bondla, Cotigao and Netravali wildlife sanctuaries. The private estates were located in the villages of Shigao, Kalay and Nayawada (bordering the Mollem National Park and Bhagwan Mahaveer Wildlife Sanctuary) and the town of Poinguinim (near the Cotigao Wildlife Sanctuary). The roadside surveys were conducted along (a) national highways NH 4A, between the towns of Ponda and Anmod, and NH 17, between the towns of Canacona and Poinguinim, and (b) other roads that run between the town of Valpoi and Mollem village, between the villages of Collem and Mollem and in Siolim village (Fig. 1).

We selected the survey sites based on reports of bonnet macaques being present as stated by local people or forest department personnel, and that confirmed the presence of bonnet macaques based on direct sightings (Kumar *et al.* 2011; Singh *et al.* 2011). In the protected areas, we used paved roads, beat paths, and cattle trails for surveying the areas. We typically travelled these paths on foot twice a day; between 06:00 h and 10:00 h, and between 15:00 h and 18:00 h. Whenever a group was observed, the location was recorded using a hand held GPS unit, and note was made of the number of visible individuals in the troop. The same methods were employed for surveying the private plantations or estates. We used vehicles, driven at speeds of approximately 10 km/hr, to conduct our surveys of roadside macaque populations (Singh

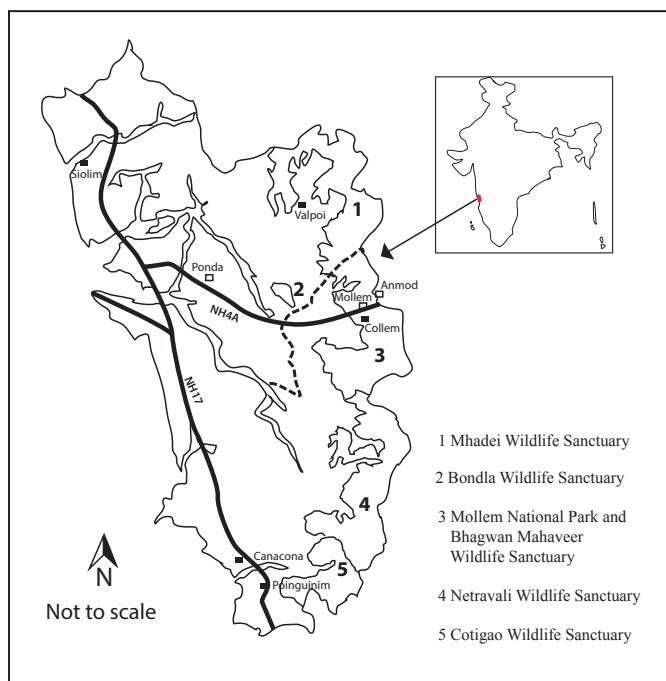


Figure 1. Locations surveyed in Goa (inset: Location of Goa within India).

and Rao 2004; Singh *et al.* 2011). We also used vehicles to conduct our surveys on some roads in the protected areas. We calculated the encounter rates of bonnet macaques as the number of macaque groups/km (Singh *et al.*, 2011).

During the course of our survey, we identified several bonnet macaque populations in the Mollem National Park and Bhagwan Mahaveer Wildlife Sanctuary. Hence in the second phase of our study, we conducted a questionnaire survey of the attitudes of local villagers towards bonnet macaques in 10 randomly chosen villages that are situated on the fringes of the Mollem National Park. The villages selected for the survey were Shigao, Kalay, Matojen, Souzamol, Bharipwada, Nayawada, Maidawada, Kumarmol, Tambdimol and Kondemol. As these villages are small with an average of 10 households each, we selected at random a total of five households from each for our questionnaire survey. We questioned respondents on the extent of wildlife-caused crop damage, frequency of macaque crop-raiding, their reactions to macaque crop depredations, and their attitudes towards retaliation measures against macaques. We also collected basic demographic and socio-economic data on all respondents, such as gender, religion, length of residence, level of education, and present employment status. We were accompanied by a local forest guard on all our interview visits. He introduced the purpose of our study to the villagers and acted as translator/interpreter when necessary.

We analyzed encounter rates of *M. radiata* groups in the different kinds of sites and percentages of participant responses to survey questions, and used non-parametric statistical tests to check if various categories were significantly different from each other (Zar 2010).



Figure 2. Bonnet macaque (*Macaca radiata*) mother and young, Goa, India. Photo by Asmita Sengupta.

Table 1. Bonnet macaque (*Macaca radiata*) groups seen in Goa. PA = protected area, RS = roadside, PE = private estate.

Location	Location type	No. of groups	Group no.	No of individuals seen
Mollem NP & Bhagwan Mahaveer Wildlife Sanctuary	PA	6	1	17
			2	16
			3	147
			4	12
			5	10
			6	26
Bondla Wildlife Sanctuary	PA	1	1	16
Mhadei Wildlife Sanctuary	PA	1	1	5
Cotigao Wildlife Sanctuary	PA	2	1	15
			2	5
Netravali Wildlife Sanctuary	PA	1	1	12
Kalay village	PE	1	1	12
Shigao village	PE	1	1	15
Poinguinim town	PE	1	1	5
National highways and other roads	RS	5	1	9
			2	12
			3	16
			4	4
			5	10

Results

We traversed a total of 334.47 km during the course of the study—protected areas (PA) 148.81 km; roadsides (RS) 103.3 km; private estates (PE) 82.26 km—and surveyed 46 locations in all. We obtained direct sightings of *M. radiata* in only 18 locations. Of the 18 groups sighted, 11 were in protected areas, five along roadsides and two in private estates. We found two groups of *M. radiata* in North Goa district and 16 groups in the South Goa district (Table 1). In protected areas, the major vegetation type was moist, mixed-deciduous forest. All the private estates where bonnet macaques were encountered had banana, coconut, and rubber plantations. The encounter rate of bonnet macaques for the entire state was 0.05 group/km and encounter rates did not vary significantly across the different kinds of sites (PA: 0.07 groups/km, PE: 0.02 groups/km, RS: 0.04 groups/km; G-test: $G = 3.007$, $df = 2$, $\alpha = 0.05$, $p = 0.22$). The average group size of *M. radiata* was 19 (range: 5–147). The largest group (147 individuals) was found at the Dudhsagar waterfall and the smallest groups, each with five individuals, were encountered in Mhadei Wildlife Sanctuary and in a private plantation in Poinguinim.

We interviewed 50 villagers regarding their perceptions of macaque crop-raiding and on their attitudes towards *M. radiata*. They included 30 men and 20 women, and were predominantly in the age categories of 21–40 years (46%) and 41–60 years (48%), respectively. The large majority of the respondents practiced Hinduism (94%); very few of them were Christians. The larger majority of them were farmers (24%), housewives (24%) or employed in government services (24%); a smaller minority worked as laborers (16%) or were small-scale businessmen (6%).

Respondents listed six wildlife species as crop depre-dators – gaur (*Bos gaurus*), bonnet macaque (*Macaca radiata*), Malabar sacred langur (*Semnopithecus hypoleucos*), fox (*Vulpes bengalensis*), jungle cat (*Felis chaus*), and wild pig (*Sus scrofa*). They rated *M. radiata* as the second most frequent, destructive, and feared species, after *S. hypoleucos* (Table 2). People identified summer as the season when bonnet macaques visited their farms most often (87%), and most of them reported that macaque crop-raids were a daily occurrence (67%). Apart from crop-raiding, bonnet macaques were also reported to cause damage to household structures such as roofs, cowsheds and granaries (53%), and also raid kitchens (7%). All respondents claimed that they never killed any macaques in retaliation. While a significantly high percentage (73%) attributed this to fear of Forest Department officials, 13% claimed they would never kill macaques as they regarded them as God. The remainder reported that they had never felt the need for retaliatory killing (Chi-square test: $\chi^2 = 10.8$, $df = 2$, $p = 0.004$). More than half the respondents (53%) reported that they shot stones at macaques from catapults to chase them from their farms; others chased away the macaques themselves (27%) or used their pet dogs (7%). A small percentage (13%) did not take any action.

Table 2. Comparative ranking of crop-raiding species.

Species	Most frequent		Most destructive		Most feared	
	Respondents (%)	Rank	Respondents (%)	Rank	Respondents (%)	Rank
Malabar sacred langur	54	1	50	1	60	1
Bonnet macaque	30	2	28	2	26	2
Wild pig	6	4	14	3	10	3
Gaur	2	5	-	-	2	4
Fox	8	3	-	-	-	-
Jungle cat	-	-	8	4	2	4

Most of the respondents (67%) opined that macaques should be translocated to the forests and never be allowed on their farms or into their households. A very small percentage (7%) responded that they had no objection to macaques raiding their farms if the Forest Department provided adequate compensation. About 26% reported that they had no problems with macaques being in their vicinity despite the damage they cause. Following Campbell-Smith *et al.* (2010), we classified the first group of respondents as “intolerant,” the second group as “moderately tolerant” and the third group as “highly tolerant.” The number of intolerant individuals was significantly higher than the other two categories (Chi-square test: $\chi^2 = 8.4$, $df = 2$, $p = 0.01$).

Discussion

It is generally believed that the bonnet macaque is ubiquitously present throughout its geographic range in India (Krishnan 1972; Roonwal and Mohnot 1977). In reality, however, little is actually known about the population status of the species in a number of parts of its range, particularly the northern (Sinha 2001). Goa is in the northern part of the Western Ghats mountain range, towards the north-west of the range of the bonnet macaque. Existing literature lists only Bhagwan Mahaveer Wildlife Sanctuary, Mollem National Park and Bondla Wildlife Sanctuary as locations where they are found in Goa (Kumar 2012; Molur *et al.* 2003). The results of our study indicate that bonnet macaques are found in many more locations and in diverse habitat types in Goa. We found more bonnet macaque populations in the southern part of Goa than the northern, but this is more likely due to a bias in our choice of survey locations than a reflection of the true status of bonnet macaques in Goa.

In the present study, the encounter rate of 0.05 groups/km is comparable to the encounter rate of 0.021 groups/km found in Karnataka (Kumara *et al.* 2010a). Although bonnet macaques are usually found at higher densities in marginal and unprotected habitats (Kumara *et al.* 2010b), in our study, the group encounter rate was higher (though not statistically significant) in protected forest areas. Also, the total number of bonnet macaque individuals was highest in forests, whereas in Karnataka, the number of bonnet macaques sighted was

highest in areas of human habitation and lowest in forests. The largest bonnet macaque group was found at the Dudhsagar waterfall in the Mollem National Park where they are provisioned by the numerous tourists visiting this region. This supports earlier observations on the bonnet macaque that they are typically found in higher numbers in areas where they live in close contact with humans and are provisioned regularly (Sinha 2001).

We saw bonnet macaques in less than 40% of the locations where they were reported to be present. In comparison, we encountered a total of 36 Malabar sacred langur (*Semnopithecus hypoleucos*) groups (range: 5 to 15 individuals) in the areas surveyed, and the encounter rate was twice that of *M. radiata* (0.1 groups/km). Although it is possible that we have underestimated bonnet macaque abundance in protected areas due to lower visibility, the larger number of bonnet macaque individuals sighted in protected areas (compared to the other habitats) suggests that our findings accurately reflect the current distribution status of the bonnet macaque in Goa. It is also instructive that we did not find any bonnet macaque groups along the western coast of the state. Unfortunately this appears to be a part of the definite pattern in the general decline of bonnet macaque populations in southern India. In the neighboring state of Karnataka, Kumara *et al.* (2010a) noted that nearly 91% of the coastal populations of *M. radiata* have been extirpated. Secondary information collected during the course of our study also indicated that bonnet macaques have long been extirpated from coastal towns such as Siolim.

Crop damage by various wildlife species, including primates, in areas in the vicinity of forests is a common occurrence in many parts of India (Chhangani and Mohnot 2004). The people in the villages bordering the Mollem National Park and Bhagwan Mahaveer Wildlife Sanctuary reported the bonnet macaque to be the second most feared, most destructive and most frequent of the crop depredating species, second only to Malabar sacred langurs. Previous studies have shown that the actual extent of damage by primates may be much less than what is perceived, the negative perceptions of people being driven mainly by the large body and large group sizes of the primate species (Nahallage *et al.* 2008). For example, in the northern periphery of the Dja Faunal Reserve, Cameroon, farmers perceived primates such as chimpanzees and

mangabeys as the worst pests in contrast to actual observations which showed squirrels and antelopes to be the animals inflicting most of the damage (Arlet and Molleman 2010). So in reality, the scale of damage by bonnet macaques may actually be comparable to the other crop-damaging wildlife species. It is also likely that respondent accounts of the heavy damage caused by bonnet macaques may have been exaggerations, influenced by the presence of the Forest Department guard who accompanied us on our visits, as villagers felt that the monetary compensation routinely awarded by the Forest Department for wildlife-caused crop damage was low and insufficient.

A study in Bangladesh on people's attitudes towards the Bengal sacred langur (*S. entellus*) documented that, despite the extensive and considerable crop and orchard damage that the species inflicts, 90% of the people interviewed were supportive of langur conservation (Khatun *et al.* 2012). In Goa, however, tolerance to primates was found to be generally very low as a significantly large proportion of the respondents demanded relocation of the animals. Interestingly, despite the lack of tolerance, all the respondents claimed that they had never hunted or killed bonnet macaques. This could be due to the Hindu belief in the sanctity of monkeys (as has been noted in other studies, for example, that of Sharma *et al.* [2011]) or, as suggested by our study responses, was more likely due to the presence of the Forest Department guard who accompanied us.

As evidenced in many parts of India (Mishra 1997; Ogra 2008, 2009; Radhakrishna and Sinha 2010), our study also throws light on the underlying tension between farmers and the Forest Department in contexts of wildlife-caused damage and the responsibility for conserving wildlife species, and reiterates the urgent requirement for studies that systematically estimate the exact amount and rate of damage by wildlife species. We also indicate the need for studies that regularly monitor the population dynamics of the bonnet macaque in other parts of its range as it clearly, even as a common species, stands in need of measures to preserve its 'Least Concern' status.

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The Eastern Hoolock Gibbon (*Hoolock leuconedys*) in Eastern Arunachal Pradesh, India

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Abstract: Lowland tropical forest in the Lower Dibang Valley district in the state of Arunachal Pradesh is the prime habitat in India of the eastern hoolock gibbon (*Hoolock leuconedys*). The present study was conducted to assess the threats to the population of *H. leuconedys* in the unprotected forest fragments of the Lower Dibang Valley district, Arunachal Pradesh. Besides field observations, we employed a structured questionnaire survey method to interview villagers, including the *Gaon Burha* (village head) and local hunters. Based on our observations and the interviewees' responses, we recorded that gibbons were occasionally hunted for bushmeat, and that dogs, the Mountain Hawk Eagle (*Nisaetus nipalensis*), and monitor lizards (*Varanus*) were evidently predators of, particularly young, gibbons in the fragmented forest patches. Indirect threats included habitat destruction and fragmentation, deforestation, extraction of non-timber forest products, livestock grazing, road construction, selective (illegal) logging, shifting cultivation, commercial cash-crops, and permanent human settlement. A conservation action plan is urgently needed to protect and conserve the eastern hoolock gibbon from extinction in the region.

Key Words: Anthropogenic threats, forest fragmentation, Lower Dibang Valley, population, hunting, predation.

Introduction

Hoolock gibbons are the only apes found in India. There are two species: the western hoolock gibbon (*Hoolock hoolock*) and the eastern hoolock gibbon (*Hoolock leuconedys*) (Mootnick and Groves 2005; Das *et al.* 2006). They are confined to the rain forests of Southeast Asia, Northeast India, and Bangladesh (Preuschoft *et al.* 1984; Leighton 1987). *Hoolock leuconedys* was earlier believed to be restricted to the east of the Chindwin River to the Salween River in Myanmar and south-western Yunnan Province in China at elevations of 1067 m to 1219 m (Groves 1971). It was reported from the state of Arunachal Pradesh, India, by Das *et al.* in 2006, between the Lohit River in the north and the high mountains of Dafa bum in the south. Further, it was also reported in the Sadiya Division, the easternmost part of the state of Assam, on the south bank of the Dibang-Brahmaputra River system (Chetry and Chetry 2010).

Lowland tropical forest of the Lower Dibang Valley district in Arunachal Pradesh is the prime habitat of *H. leuconedys*. Surprisingly, only a few population surveys have been carried out in India (Das *et al.* 2006; Chetry *et al.* 2008,

2010; Sarma *et al.* in press). The species has otherwise been surveyed in China (Lan 1994; Tian *et al.* 1996; Zhang 1998; Zhang *et al.* 2002; Fan *et al.* 2011) and Myanmar (Geissmann 2007), and just recently in Arunachal Pradesh, covering a significant portion of the species' population and highlighting the anthropogenic threat faced by the species in matrix habitats in India (Sarma *et al.* in press).

Forests in the foothill areas of the Lower Dibang Valley district are being heavily exploited, destroying and fragmenting the gibbon's habitats (Sarma *et al.* 2011, 2013). The forest fragments are subjected to logging, fuelwood collection, grazing, and poaching (Umopathy and Kumar 2000). In this paper we report on threats to the long-term survival of *H. leuconedys* in the agricultural matrices (unclassified state forests) in Arunachal Pradesh, India.

Methods

Study area

The study was conducted in protected and unprotected forest fragments in the Lower Dibang Valley district in the state of Arunachal Pradesh, India. The only protected area

there is the Mehao Wildlife Sanctuary, which is surrounded by numerous unprotected forest patches forming a matrix of habitats. The Mehao Wildlife Sanctuary lies roughly between 93°30' and 95°45'E and 28°05' and 28°15'N. The terrain is entirely hilly, ranging in altitude from 400 m to 3568 m above sea level. The forest types of the study area vary with altitude, and consist of tropical evergreen forest (up to 900 m), subtropical and temperate forest (above 900 m to 1800 m), temperate broadleaf forest (1800 m to 2800 m), and temperate conifer forest (2800 m to 3500 m) (Rawat *et al.* 2009). There are a number of perennial streams and lakes draining the sanctuary. The biological importance of the sanctuary is due to the fact that the area has a combination of Palaearctic, Indo-Chinese and Indo-Malayan floral and faunal elements (Rawat *et al.* 2009). The climate is cool throughout the year, with 2680 mm average annual rainfall. The richness and diversity of the flora provides for a highly diversified fauna.

The main tribes inhabiting the area are the *Idu-Mishimis* and *Adis*. They cultivate mainly rice, maize, and millet; their staple foods apart from bushmeat. They also eat wild vegetables, roots, tubers, and fruit. Agriculture is the mainstay of the economy, and traditional shifting cultivation 'Jhum' is the most common farming practice.

Survey techniques

A field survey was conducted from November 2010 to October 2012. Using a questionnaire, we interviewed the *Gaon Burha* (village heads), hunters, and other local people to assess the present threats to the gibbons in unclassified state forests (USF) near the Mehao Wildlife Sanctuary. We selected four of the 14 villages in and around the sanctuary—Horupahar, Koronu, Injuno and Delo—based on the extent to which the people depended on natural resources, their proximity to the forest patches, and the occurrence of the eastern hoolock gibbon (Fig. 1). We selected 20 interviewees from each village using a stratified random sampling technique. The primary data were collected through structured and open-ended questionnaires; secondary data were collected from published and unpublished reports, research papers and articles, as well as through interviews of forest department officials. We also interviewed 20 hunters.

Results

Human demography

There are 14 villages in and around the Mehao Wildlife Sanctuary (Table 1). A total of 937 households were reported in the 14 villages, comprising a population of *c.* 4238 people. The largest village was Koronu (population 793) and the smallest was Simari (population 12); both located on the southern boundary of the sanctuary. These people are extending their agricultural activities into the unclassified forest areas, where they also extract non-timber forest products to meet their day-to-day needs. This intrusion affects the gibbon population by reducing the extent of pristine forest cover.

Livelihood options: emergence of anthropogenic threats

Agriculture was the primary occupation for 85% of the hunters interviewed (*n* = 20). Cultivation, listed as the highest source of income by nearly 80% of interviewed villagers (*n* = 80), was followed by non-timber forest products (13%), and hunting (7%) (Fig. 2). Seventy per cent of the respondents listed subsistence use, 19% indicated trade and 11% listed human-wildlife conflict as the main reason for hunting.

Anthropogenic threats

A number of threats were recorded in the study area, based on field observations, questionnaires, personal interviews and discussions with village heads, hunters and local people. These threats were grouped into two categories—direct and indirect—based on their impact on the population of *H. leuconedys* and their habitats (Fig. 3).

Direct threats: hunting and predation

Hunting was found to be the major activity posing a direct threat to the gibbons. Although the local tribe *Idu Mishmi* do not hunt gibbons due to a cultural taboo, the *Adi* hunt the species for bushmeat. Another alarming direct threat recorded was predation by dogs reared by local villagers to protect them from wild animals. About 20–25 gibbons were reported killed by dogs in the last seven years. Other predators recorded were the Mountain Hawk Eagle (*Nisaetus nipalensis*) and monitor lizards (*Varanus*), which target mainly infant gibbons. Ten attacks by Mountain Hawk Eagles and monitor lizards on immature gibbons were recorded during the period of the study.

Indirect threats: habitat destruction and fragmentation

Indirect threats were subdivided into habitat destruction and fragmentation. Fragmentation is due to selective logging and road construction, whereas habitat destruction was driven by a number of activities, such as extension of agricultural land, encroachment, tree felling for commercial purposes, and

Table 1. Villages in and around Mehao Wildlife Sanctuary and their population status. (Source: Department of Statistics, Arunachal Pradesh).

	Name of village	No. of households	Population
1	Bhismaknagar	34	128
2	Koronu	175	793
3	Dello	59	240
4	Injuno	137	707
5	Abango	115	506
6	Simari	3	12
7	Balek	93	442
8	Cheta I & II	74	300
9	Rayang	64	358
10	6 Kilo	57	193
11	Kebali	34	98
12	Horu Pahar	37	265
13	Chidu	38	129
14	Tewari Goan	17	67
	Total	937	4238

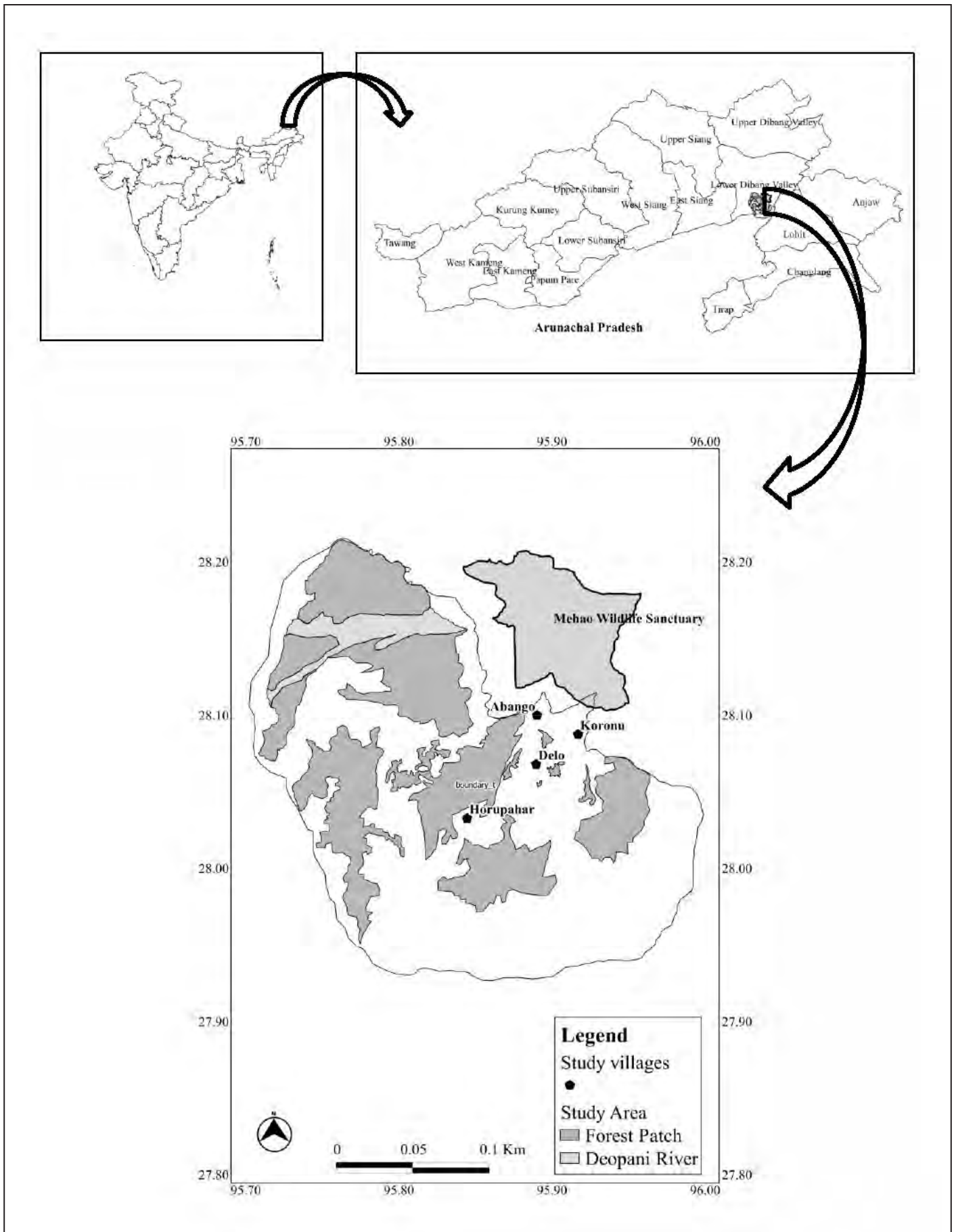


Figure 1. The study area showing the locations of the study villages and existing forest patches in the Lower Dibang Valley district, Arunachal Pradesh, India.

shifting cultivation. We recorded 33 plant species selectively logged for fuelwood, timber, and housing materials during the study period. Twenty-two of them were used by gibbons for food (Table 2). Other indirect threats damaging the habitat of the gibbons include livestock grazing, over extraction and over exploitation of non-timber forest products (including wild vegetables, leaves of many medicinal plants, fuelwood and small poles and boles for building houses).

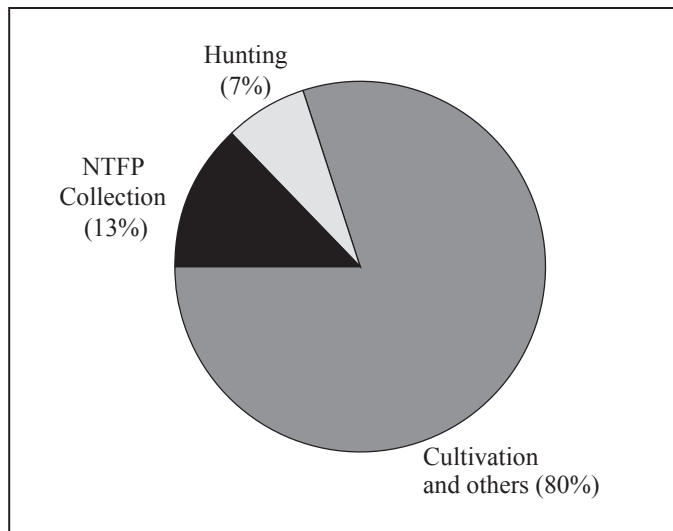


Figure 2. Percentage livelihood options of the villagers in the study area.

Discussion

Lowland tropical rain forests in Northeast India, particularly Arunachal Pradesh, are the most species rich terrestrial ecosystems harboring gibbons in India. Substantial degradation of these rain forests in and outside of protected areas has led to fragmentation and conflict, affecting the populations of both the western hoolock (*Hoolock hoolock*) and eastern hoolock (*Hoolock leuconedys*) gibbons. Populations in the wild have declined by more than 90% over the past three decades due to numerous anthropogenic threats (Walker *et al.* 2007). The western hoolock gibbon is the species most studied for anthropogenic threats in its range (Choudhury 1990, 1991; Mukherjee *et al.* 1992; Srivastava 1999; Ahmed 2001; Srivastava *et al.* 2001a, 2001b; Malone *et al.* 2002; Das and Bhattacharjee 2002; Das *et al.* 2004; Solanki and Chuita 2004; Das *et al.* 2006; Walker *et al.* 2007), and most of the threats apply also to the eastern hoolock gibbon in Arunachal Pradesh. These threats have affected the conservation status of the gibbons (Alfred and Sati 1990, 1994; Choudhury 1991; Islam and Feeroz 1992; Kakati 1997). However, for a species such as the eastern hoolock gibbon, the range of which has yet to be ascertained, this compilation might still be incomplete. Habitat loss and fragmentation have been reported as major anthropogenic threats for the eastern hoolock gibbon throughout its known range (Table 3).

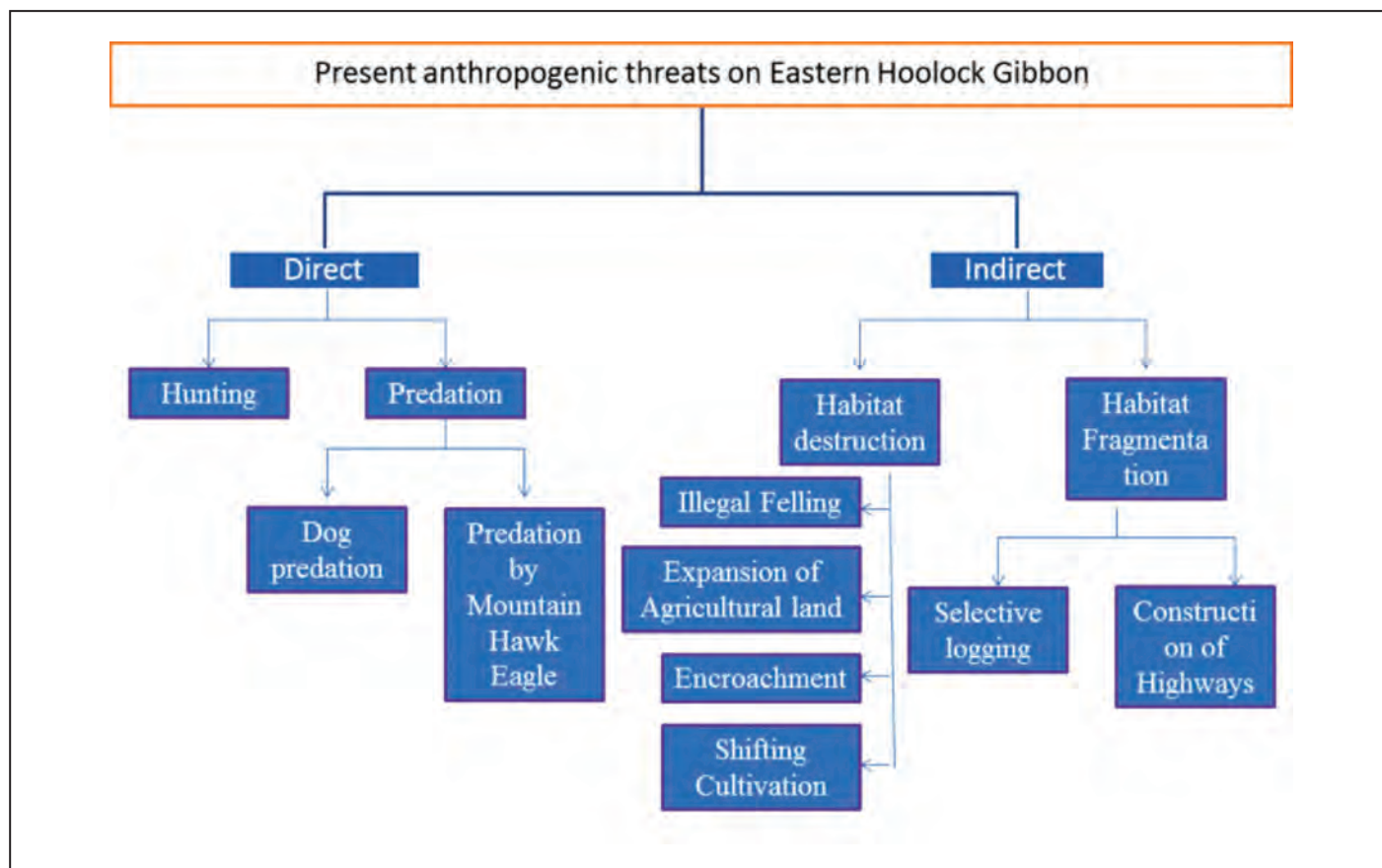


Figure 3. Flow chart of anthropogenic threats recorded for the eastern hoolock gibbon (*Hoolock leuconedys*) in the study area.

Table 2. Plant species selectively logged from the forest areas and their purposes.

	Species name (local name)	Family	Purposes	Food plant*
1	<i>Duabanga grandiflora</i> (Khokon)	Myrtaceae	Construction	X
2	<i>Morus laevis</i> (Bola)	Moraceae	Construction	X
3	<i>Terminalia myriocarpa</i> (Hollock)	Dipterocarpaceae	Construction	X
4	<i>Bambusa tulda</i> (Jati Bans)	Poaceae	Construction	X
5	<i>Bambusa hemiltonii</i> (Kako Bans)	Poaceae	Fuelwood	
6	<i>Bischofia javanica</i> (Uriam)	Euphorbiaceae	Construction/ Fuelwood	X
7	<i>Sterculia</i> spp. (Udal)	Malvaceae	Fuelwood	X
8	<i>Sterculia villosa</i> (Dewachali)	Malvaceae	Fuelwood	X
9	<i>Albizia procera</i> (Koroi)	Caesalpiniaceae	Fuelwood	X
10	<i>Delinia indica</i> (Otenga)	Dilleniaceae	Fuelwood	
11	<i>Bauhinia</i> spp. (Kanchan)	Verbenaceae	Fuelwood	
12	<i>Cinnamomum glaucescens</i> (Gonkorai)	Lauraceae	Construction	X
13	<i>Bombax ceiba</i> (Semal)	Bombacaceae	Construction / light work	X
14	<i>Alstonia scholaris</i> (Satiana)	Apocynaceae	Fuelwood	
15	<i>Pterospermum acerifolium</i> (Hatipoila)	Malvaceae	Fuelwood	
16	<i>Ailanthus integrifolia</i> (Borpat)	Simaroubaceae	Fuelwood	X
17	<i>Calamus</i> spp.	Arecaceae	Construction	
18	<i>Lannea coromandelica</i> (Jiapoma)	Anacardiaceae	Construction/Fuelwood	X
19	<i>Gmelina arborea</i> (Gomari)	Lamiaceae	Construction	X
20	<i>Chukarasia tabularis</i> (Bogipoma)	Meliaceae	Construction/ Fuelwood	X
21	<i>Calamus erectus</i> (Jeng patta)	Arecaceae	Construction	
22	<i>Michelia champaca</i> (Teeta Sopa)	Magnoliaceae	Construction	
23	<i>Melia azederach</i> (Gorat Neem)	Meliaceae	Fuelwood	
24	<i>Kydia glabrascence</i> (Pichola)	Malvaceae	Fuelwood	X
25	<i>Erythrina stricta</i> (Mader)	Fabaceae	Fuelwood	X
26	<i>Mesua ferra</i> (Nahar)	Meliaceae	Construction	
27	<i>Stereospermum chelonoides</i> (Paroli)	Bignoniaceae	Construction/ Fuelwood	X
28	<i>Shorea assamica</i> (Mekai)	Dipterocarpaceae	Construction	X
29	<i>Spondias pinnata</i> (Amora)	Anacardiaceae	Fuelwood	X
30	<i>Toona cilata</i> (Jatipoma)	Meliaceae	Construction/ Fuelwood	X
31	<i>Terminalia citrina</i> (Hilika)	Combretaceae	Construction	X
32	<i>Calamus tenuis</i> (jati bet)	Arecaceae	Construction	
33	<i>Neolamarckia cadamba</i> (Kadam)	Rubiaceae	Fuelwood	X

*Food plant data of Das *et al.* (2004) and Kakati *et al.* (2004).

In Arunachal Pradesh, human settlements and livestock grazing have resulted in a new threat in the form of attacks by free ranging dogs, associated also with roundworm (*Toxocara canis*) infestations. Firewood collection and extraction of non-timber forest products damage the forest canopy, forcing the gibbons to go to the ground (Sarma *et al.* 2013), making them vulnerable to dog predation, which will surely affect the survival rates of young gibbons, especially in the long run. An episode of dog predation was recorded by Panor (2011); a young female was rescued from the mouth of a dog.

The land use pattern is gradually changing; more and more local farmers are switching to short-duration, cash-crop cultivation for quick returns. The rate and extent of forest

encroachment, disturbance and depletion are determined by many factors, including the legal status and land ownership of each forest area (Baranga *et al.* 2009). Local people have no clear understanding of the existence of the Mehao Wildlife Sanctuary due to the lack of a well-marked boundary, and still think that the land belongs to their forefathers. As such they believe they have the right to hunt and to carry out their day-to-day activities there (Chetry *et al.* 2010). Occasional hunting and illegal selective logging and collection of timber are widespread in the area. There are many wood-based industries within a radius of 5 km from the boundaries of the sanctuary under the Mehao Forest Division. The forest inside the



Figure 4. Major threats to *H. leuconedys* in and around Mehao Wildlife Sanctuary: (a) and (b) male and female gibbons in a fragmented landscape; (c) pasture; (d) and (e) tea plantations; (f) and (g) selective logging (h) timber mill in the study area; and (i) skulls of hunted gibbons.

Table 3. Threats recorded for the eastern hoolock gibbon (*Hoolock leuconedys*) throughout its range.

Country	Forest status	Threats	Sources
Myanmar	Protected and non-protected habitat of Myanmar	<ul style="list-style-type: none"> – Habitat loss and degradation caused by shifting cultivation and timber extraction. – Hunting for trade and subsistence was also recorded as a serious threat – Poaching by villagers. 	Lwin <i>et al.</i> (2011)
China	Protected and non-protected habitat of China	<ul style="list-style-type: none"> – Commercial logging both historical and current leading to habitat destruction – Hunting was also recorded as threat. – Agricultural encroachment and population fragmentation 	Fan <i>et al.</i> (2011)
India	Namsai Reserve Forest	<ul style="list-style-type: none"> – Habitat destruction and fragmentation. – Change in land use pattern. – Shifting cultivation. – Conversion of forest area to tea garden. – Expansion of road network. – High tension electric wires. 	Das <i>et al.</i> (2006)
	Mehao Wildlife Sanctuary and Koronu circle	<ul style="list-style-type: none"> – Forest loss and fragmentation due to expansion of agriculture. – The conversion of forest for the commercial cultivation of orange, ginger and cardamom – Construction of high-tension electric power line – Encroachment of the forest for human settlements and for small-scale agriculture. 	Chetry <i>et al.</i> (2008, 2010)
	Reserve forests of Assam	<ul style="list-style-type: none"> – Forest loss and fragmentation due to the expansion of agricultural activities, encroachment by human settlements, selective illegal felling – Collection of firewood and grazing. 	Chetry & Chetry (2010)

sanctuary is still dense, but timber mafias are now targeting felling for commercial purposes inside the sanctuary.

The forests in the foothills are suffering from considerable exploitation, which leads to the destruction and fragmentation of the habitat, adversely affecting the survival of the gibbons. Besides the protected areas, the unclassified state forests, particularly in the southwestern vicinity of the sanctuary that hold a significant portion of the total gibbon population in the state, are facing serious threats in terms of encroachment for agricultural and horticultural practices and logging (Panor 2011; Sarma *et al.* in press). This is evidenced by the number of stumps in the study area. An average of 1.85 stumps per kilometer was reported from the four villages around the Mehao Wildlife Sanctuary (Krishna *et al.* 2012). As the tree densities of all the four study sites are very low, the gibbons are facing difficulties in dispersing. These threats are common in Assam and Arunachal Pradesh.

Based on these anthropogenic threats, the gibbon population is believed to be declining rapidly. Immediate interventions are needed to conserve this vulnerable species; through a captive breeding program for restocking of the wild population and reintroducing the species into protected areas. Two major conservation actions have already been undertaken in Arunachal Pradesh. The Wildlife Trust of India (WTI) in collaboration with the Forest Department has translocated a few isolated groups to the Mehao Wildlife Sanctuary, although they have not been monitored. Moreover, the Biological Park, Itanagar, under the guidance of the Central Zoo Authority (CZA), has taken up the initiative for a conservation breeding program with the ultimate goal of releasing captive-bred individuals into the wild. However, habitat improvement through multipurpose tree plantations and the construction of canopy bridges to connect the remnant forest patches for future survival of the species in the wild is the prime necessity in its fragmented habitats. Local awareness and involvement of the native communities are also needed for the conservation of this species.

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Conservation Prospects for the Lion-tailed Macaque (*Macaca silenus*) in the Forests of Sirsi-Honnava, Western Ghats, India

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Abstract: The lion-tailed macaque (*Macaca silenus*) is one of the most threatened of the primates of the Western Ghats. Confirmation of its large metapopulation in a relatively unprotected area (a reserve forest) of Karnataka has marked an important step for the future of this population. The number of lion-tailed macaques estimated was 638 in 31 groups with an average group size of 20.6, excluding lone males. A review of the literature confirms that this is one of the larger known populations in the wild. This reserve forest faces a number of threats because of anthropogenic activities such as habitat fragmentation, encroachment and developmental projects. In an attempt to save and restore the northernmost habitat of the lion-tailed macaque, we proposed that the forests where they live be declared a wildlife sanctuary or conservation reserve, using them as an umbrella species for conservation. In response to this, the forest department of the Government of Karnataka notified the proposed area, with only minor modifications to the boundary, as the ‘Aghanashini Lion-tailed Macaque Conservation Reserve’. We suggest some immediate management interventions to minimize further pressure on this highly threatened habitat.

Key words: Primates, protected area, umbrella species, conservation reserve, management intervention, Karnataka, lion-tailed macaque

Introduction

The hill ranges of the Western Ghats cover less than 6% of India’s landmass but harbor more than 30% of the world’s plant and vertebrate species (Das *et al.* 2006), and are thus considered a global biodiversity hotspot (Myers *et al.* 2000). About 12% of the mammal species present in the Western Ghats is endemic (Das *et al.* 2006). The IUCN Red List ranks the lion-tailed macaque (*Macaca silenus*) as Endangered (IUCN 2013); endemic to the narrow ranges of the southern and central Western Ghats. Molur *et al.* (2003) projected a total lion-tailed macaque population of about 3,500 individuals in 49 sub-populations in eight locations in the Western Ghats. They are locally threatened in most of the protected areas and reserve forests of the state of Karnataka (Kumara and Sinha 2009). Karanth (1985) reported about 3,000 individuals in 123 groups in 19 locations in Karnataka from the northernmost Kumta range to southern Brahmagiri Wildlife Sanctuary. Since then, however, there have been declines in numbers of about 69% to 90% in 14 of these forest reserves due to habitat loss and fragmentation and hunting, and hunting

has eliminated them entirely from five reserves (Kumara and Sinha 2009).

In a study based largely on secondary information, Karanth (1985) reported few lion-tailed macaque groups in the forests of Sirsi-Honnava. A short survey by Kumara and Singh (2004a), however, indicated a population of more than 250 individuals in the same forests; among the few large populations of this species in the entire Western Ghats (Kumara and Singh 2004a). The Sirsi-Honnava lion-tailed macaques are, however, facing severe threats from encroachment of the forests and valleys for agriculture, developmental activities such as construction of roads, transmission lines, dams, hydroelectric power plants, and hunting (Kumara and Singh 2004a; Kumara *et al.* 2008). The problem is that reserve forests are not part of the protected area network. The forests are contiguous, and a conservation strategy is urgently needed for the lion-tailed macaques there (Kumara *et al.* 2008; Kumara and Sinha 2009). This region also harbors many endemic and endangered species, including plants such as *Semecarpus kathalekanensis* (Anacardiaceae), *Madhuca bourdillonii* (Sapotaceae), and *Syzygium travancoricum* (Myrtaceae) (Chandran *et al.* 2008),

about 26 amphibians endemic to the Western Ghats (Kumara *et al.* 2008), 17 globally threatened large mammals (Kumara and Singh 2004b), and also unique ‘*Myristica* swamps’ (Chandran *et al.* 2008). The study by Kumara and Singh (2004b) that reported on the large population of lion-tailed macaques there stressed the need for their conservation, but did not provide conservation measures, maps, boundaries or a protocol for population monitoring. In this study, we reassessed the current status of lion-tailed macaques using a sweep sampling method, mapped the population based on their locations, and developed the boundaries (based on village boundaries) for the management of the area. We discuss strategies for the conservation of the area, indicating the lion-tailed macaque as an umbrella species for the region.

Methods

Study area

The study site is in the central Western Ghats, in the district of Uttara Kannada, state of Karnataka, south India (Fig. 1), 14°23'N to 14°23'38"N and 74°48'E to 74°38"E. The legal status of the forest is “Reserve Forest,” with mosaics of revenue lands interspersed around them (Kumara and Singh 2004a). The study site falls under the administrative

jurisdiction of Kanara Forest Circle, represented by the Kyadagi and Siddapur forest ranges in the Sirsi Territorial Forest Division, and the Kumta, Honnavara and Gersoppa forest ranges in the Honnavara territorial forest division. The altitude ranges from 300 m to 800 m above sea level. The wet season is from May to October. It rains most in July; an average of 3,000 mm. Even though the region does not receive the north-east monsoons, the total annual rainfall is about 6,000 mm. The study site forms the northern limit of the evergreen forests of plains and low elevations (Pascal 1988). Forest in the study site has been classified as “west coast tropical evergreen forest” with low-level type floristics (Champion and Seth 2005). The vegetation type is *Persea macarantha* – *Diospyros* spp. – *Holigarna* spp., which has been replaced by the dominance of *Dipterocarpus indicus* – *Diospyros condolleana* – *Diospyros oocarpa* because of human interference (Pascal 1988). The major ethnic community in the area includes *Naika*, *Vokkaliga*, *Gowda*, *Harijana* and *Brahmin*.

Data collection and analysis

The survey was conducted from January to April 2008 for 63 days, using the sweep sampling method for total counts (NRC 1981) appropriate for rare and patchily distributed

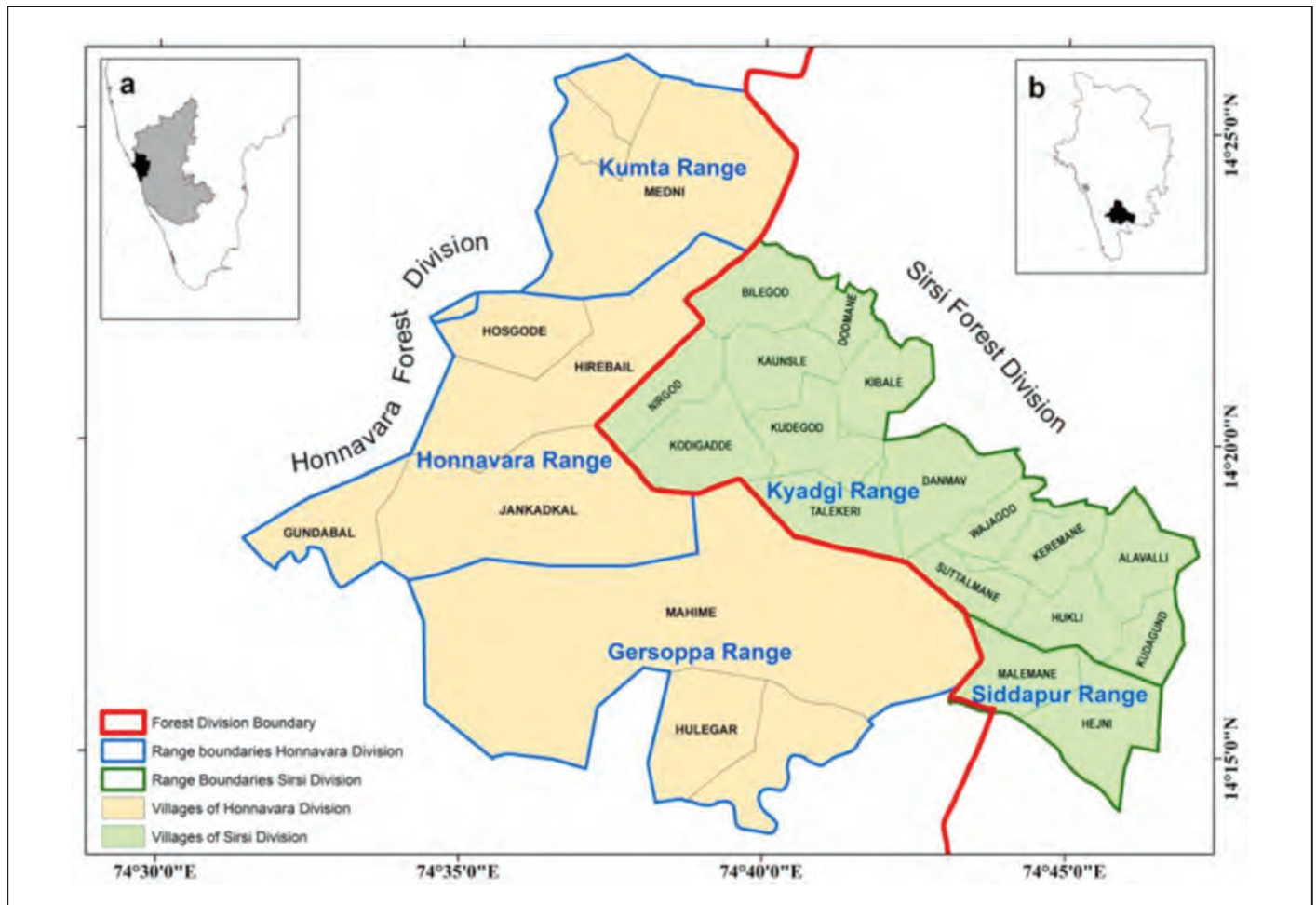


Figure 1. Different forest ranges of Sirsi-Honnavara divisions with village boundaries.

species (Whitesides *et al.* 1998; White and Edwards 2000). This method increases the probability of seeing the species, but counting the same group twice can be an issue if the group spread is large and they are moving. We avoided double counting by noting the sighting distance, location, time of the sighting and the direction of movement. Suitable habitat for the lion-tailed macaque was considered based on earlier surveys and group locations (Kumara and Singh 2004a). We plotted those group locations on a map, with a 2-km radius presumed to be equal to the maximum home range size of a lion-tailed macaque group (Green and Minkowski 1977). We considered those plots as the sampling area for sweep sampling. We set up predetermined lines for sampling in each such sampling area. A team of three trained observers walked each line for 3–4 consecutive days. The observers walked parallel to each other, maintaining an inter-individual distance of 100 m, to maximize the chances of finding lion-tailed macaque groups. We assumed that neither visibility nor detectability factors would bias the data since they remained constant throughout the study site, and all the observers were familiar with the species and its habits.

We conducted the surveys from 05:30 h to 12:00 h and 15:00 h to 18:00 h. Lion-tailed macaques are active and vocalize throughout the day (Kumar 1987). Geo-coordinates were recorded using handheld Garmin GPS60 and GPS72 for each group sighting, as were group sizes, generally from counts at common cross-over points, by spending sufficient time with the group (maximum 30 minutes). Previous studies have documented the home range of a group to be about 5 km² (Green and Minkowski 1977; Kumar 1987; Umapathy 1998). Hence, we considered each group sighting within a range of a 1.5-km radius from another sighting to be the same group, unless 1) the two groups were seen one very soon after the other, or 2) the group size and identity of each were confirmed as different. The surveys were carried out in a relatively short period, in the pre-monsoon season, to eliminate any bias caused by changes in ranging across the seasons. The intergroup distance was extracted on a GIS platform using ArcView3.2. We walked 1,056 km to sample Sirsi-Honnava; including 546 km, 56 km, 87 km, and 354 km in the Kyadagi, Siddapura, Honnavara and Gersoppa ranges, respectively. We estimated the number of groups and the population size in the area on the basis of the location of sightings and group sizes. Complete group counts were used for calculating the average group size to estimate the minimum number of individuals. We also collected data on the human population and demography from all the villages, as well as other details on developmental activities in the area from the Karnataka Forest Department records (Kumara *et al.* 2008).

Results

Population estimate

We obtained a total of 49 sightings of lion-tailed macaque groups and, on three occasions, single lone males. The estimated number of groups for the region was 31 (Tables 1 and 2);

15, 2, 1, 2 and 11 groups in Kyadagi, Siddapura, Kumta, Honnavara and Gersoppa ranges, respectively (Fig. 2). Complete group counts were obtained for 24 groups, providing a mean group size of 20.5 individuals/group (Table 2). The group size varied from 12 to 35. About 63% of the groups had sizes of between 16 and 25 (Fig. 3). The estimated minimum population size in the study site was 638 monkeys in 31 groups, excluding the three lone males.

Boundary demarcation

Although the forests of Sirsi-Honnava include semi-evergreen forests, moist-deciduous forests and various plantations, 27 lion-tailed macaque groups were located in the evergreen forests and only four were found in the semi-evergreen and semi-deciduous forests (Fig. 4). All the groups were restricted to the highly undulating terrain of the Ghats with slopes of more than 35% (Fig. 5). Officially, the entire habitat range of the lion-tailed macaques in Sirsi-Honnava is a reserve forest. The boundary for the proposed protected area delimits 32,479 ha. However, the suitable habitat for lion-tailed macaques based on forest cover within the proposed area is rather less (27,519 ha). It encompasses 28 villages with a human population of about 15,041 (46.3 people/km²).

Discussion

The previous survey by Kumara and Singh (2004a) was based on single-observer sweep sampling, but the limitation of this method would be the possibility of an inflated abundance estimate due to counting the same groups twice (Struhsaker 2002). A conventional line-transect technique was difficult due to the hilly terrain. The multi-observer, sweep-sampling method was the best technique to estimate the abundance there. The limitations of this method include the fact that it may require many trained observers, and it does not overcome the problem of changes in group size when social groups are the unit of measure (Struhsaker 2002). To overcome these limitations, we had three observers trained before each survey to minimize error in recognizing and locating the macaques. Changes in group composition and size were few in the short period we surveyed.

Our survey demonstrated the persistence of a large population of lion-tailed macaques in the Sirsi-Honnava forests (Table 3). Lion-tailed macaques have been extirpated further north in the Anshi, Kumbarawada, Varahalli, Janmane, and

Table 1. Sampling effort, lion-tailed macaque (*Macaca silenus*) groups sighted and groups estimated in different forest ranges of Sirsi-Honnava.

Range	No. of km walked	No. of groups seen	No. of estimated groups
Kyadagi	546	27	15
Siddapura	56	2	2
Kumta	20	1	1
Honnava	67	3	2
Gersoppa	367	16	11
Total	1056	49	31

Table 2. Details of lion-tailed macaque (*Macaca silenus*) groups found, with names, group size, geo-coordinates and altitude.

No.	Group name	Group size	Geocoordinates	Altitude (m asl)
1	Sannamane gudde	6*	14°18'48.1"N, 74°39'24.5"E	437
2	Maavinmarada savalu	9*	14°20'14.8"N, 74°37'35.5"E	335
3	Hirebylu	12	14°24'18.5"N, 74°37'8.3"E	342
4	Chiksuli	17	14°21'4.2"N, 74°40'30.5"E	569
5	Krishnaghatta	24	14°21'10.9"N, 74°40'0.8"E	551
6	Hullingadde thota	7*	14°18'55.22"N, 74°34'28.0"E	433
7	Hosthota	17	14°20'18.4"N, 74°40'13.4"E	430
8	Sarvanthota	20	14°19'09.2"N, 74°39'53.6"E	436
9	Salikanu	30	14°20'26.8"N, 74°38'31.8"E	532
10	Dasur	26	14°19'19.9"N, 74°40'26.4"E	460
11	Kudegod	22	14°19'38.0"N, 74°41'5.2"E	515
12	Hapregoli	15	14°18'25"N, 74°42'56.4"E	750
13	Kalegadde	20	14°18'19.8"N, 74°42'3.0"E	690
14	Galmav	15	14°17'54.5"N, 74°42'8.5"E	530
15	Suthlumane	19	14°17'36.4"N, 74°44'0.9"E	625
16	Doddgudde kaanu	19	14°17'31.9"N, 74°43'37.2"E	698
17	Kotegudda	17	14°17'16.9"N, 74°43'32.0"E	703
18	Hukkali	22	14°17'17.6"N, 74°45'0.7"E	744
19	Tormay	21	14°18'44.5"N, 74°41'35.1"E	650
20	Hegdegadde halla	1	14°19'48.1"N, 74°38'57.0"E	460
21	Malemane	35	14°17'16.7"N, 74°43'20.4"E	644
22	Kathlekaanu	14	14°16'25.4"N, 74°44'16.9"E	502
23	Kodgi-kerigadde	25	14°18'23.4"N, 74°37'55.5"E	438
24	Kendikuli	9*	14°17'57.5"N, 74°40'26.9"E	544
25	Mahime	0*	14°17'16.7"N, 74°43'20.4"E	646
26	Sasiguli-1	13	14°17'15.1"N, 74°41'3.2"E	542
27	Sasiguli-2	33	14°17'25.2"N, 74°41'12.7"E	508
28	Dundmaav-1	14	14°17'11.1"N, 74°42'3.6"E	466
29	Dundmaav-2	23	14°17'45"N, 74°42'17.5"E	563
30	Matnigadde	21	14°16'53.9"N, 74°42'38.9"E	540
31	Vatehalla	17*	14°16'15.72"N, 74°42'56.8"E	542
32	Vatehalla	1	14°16'15.7"N, 74°42'56.8"E	542
33	Vatehalla	1	14°16'15.72"N, 74°42'56.8"E	542
34	Water falls	6*	14°16'36.4"N, 74°42'17"E	502

*Indicates groups for which complete group sizes could not be obtained

Honnavara ranges (Kurup 1978; Bhat 1982; Karanth 1985). The forests of Sirsi-Honnavara contain the northernmost population in its present range. The mean group size was slightly higher than in some populations but quite similar to those of others; for example, 16.3 in Indira Gandhi Wildlife Sanctuary (Singh *et al.* 1997), 19.6 in Silent Valley National Park (Joseph and Ramachandran 1998), and 33.2 in Theni (Kumara *et al.* 2011a). Sightings of several lone males in the present survey

Table 3. Number of estimated groups of lion-tailed macaques in the forests of Sirsi-Honnavara between different studies.

Range	Karanth (1985)	Kumara and Singh (2004a)	Kumara <i>et al.</i> (2008)
Gersoppa	4	10	11
Siddapura	2	2	2
Kyadagi	1	17	15
Honnavara	0	3	2
Kumta	1	0	1
Total	8	32	31

also indicate dispersal, ensuring gene flow. The mean group size of 20.5 is, however, lower than that found in the earlier survey of 2002–2003 (24.7 individuals/group) (Kumara and Singh 2004a). The difference in the group size between the study periods can be attributed to observer bias or to increased hunting, or mortality due to such as electrocution or snares set for other animals (Kumara and Singh 2004b). There is little direct hunting; >90% of the inhabitants are Hindus who believe in the monkey god Hanuman, and killing monkeys is taboo (Kumara and Singh 2004b). During our survey, however, we were told by many villagers that people from Kerala who have settled in neighboring *taluks* of the Shimoga district and people from coastal areas venture into the region using local hunters, to hunt primates, sambar and gaur. We deduce that this is increasingly damaging to the entire wildlife of the area, causing local extinctions of many of the large mammals. Such local extinctions and sharp declines in the lion-tailed macaque population have been reported in different parts of Karnataka (Kumara and Sinha 2009). Hunting should now be considered as one of the major threats prevalent in the area.

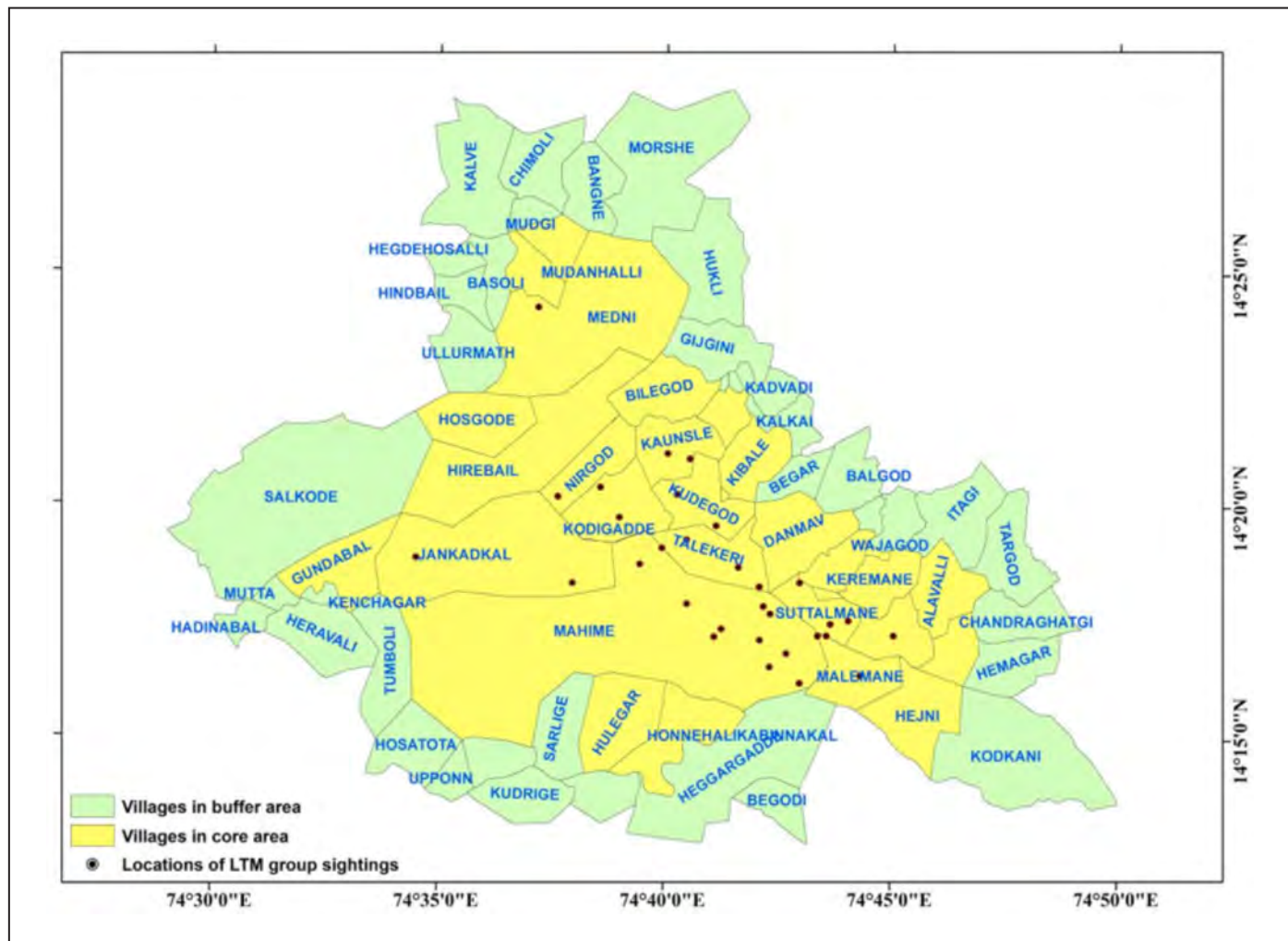


Figure 2. Proposed core and buffer areas on village boundaries based on the locations of lion-tailed macaque (*Macaca silenus*) groups in the forests of Sirsi-Honnavara.

The highly undulating terrain where there are evergreen forests is the most important habitat in the landscape for the lion-tailed macaques. The high human density has led local people to expand their agriculture and increase the area of settlements and villages. Forests are shrinking, especially

evergreen forest, at a rapid rate—1.9% yearly leading to the loss of 11.5% just in the last decade (Kumara *et al.* 2011b).

As a first step towards protection, we fixed the boundaries that should be notified as a protected area, considering the forests containing lion-tailed macaques with the village boundaries as core areas, and the adjoining village boundaries as buffer areas (Fig. 2) as was proposed by Kumara *et al.* (2008). In response to this, the forest department of the Government of Karnataka has notified the proposed area, with little modification, as the “Aghanashini Lion-tailed Macaque Conservation Reserve.”

Until the conservation management plan is prepared, we suggest a few immediate interventions, such as avoiding cutting monoculture plantations within the habitat, since they act as a link between most forest stretches and also avoid development activities (building roads or laying electricity lines) and prevent further fragmentation of the habitat. Extension of the existing farmlands and further honey-combing of valleys for agriculture, uncontrolled timber extraction, and leaf-litter and green-manure collection (Kumara *et al.* 2008, 2011b) are some of the activities that are detrimental to the forests.

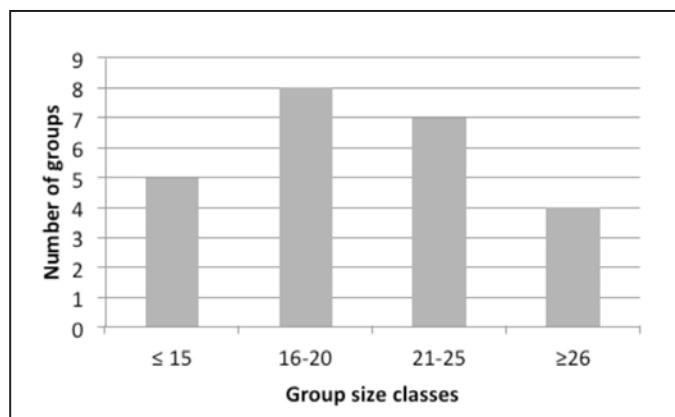


Figure 3. Number of lion-tailed macaque (*Macaca silenus*) groups in different group size classes in Sirsi-Honnavara forest divisions.

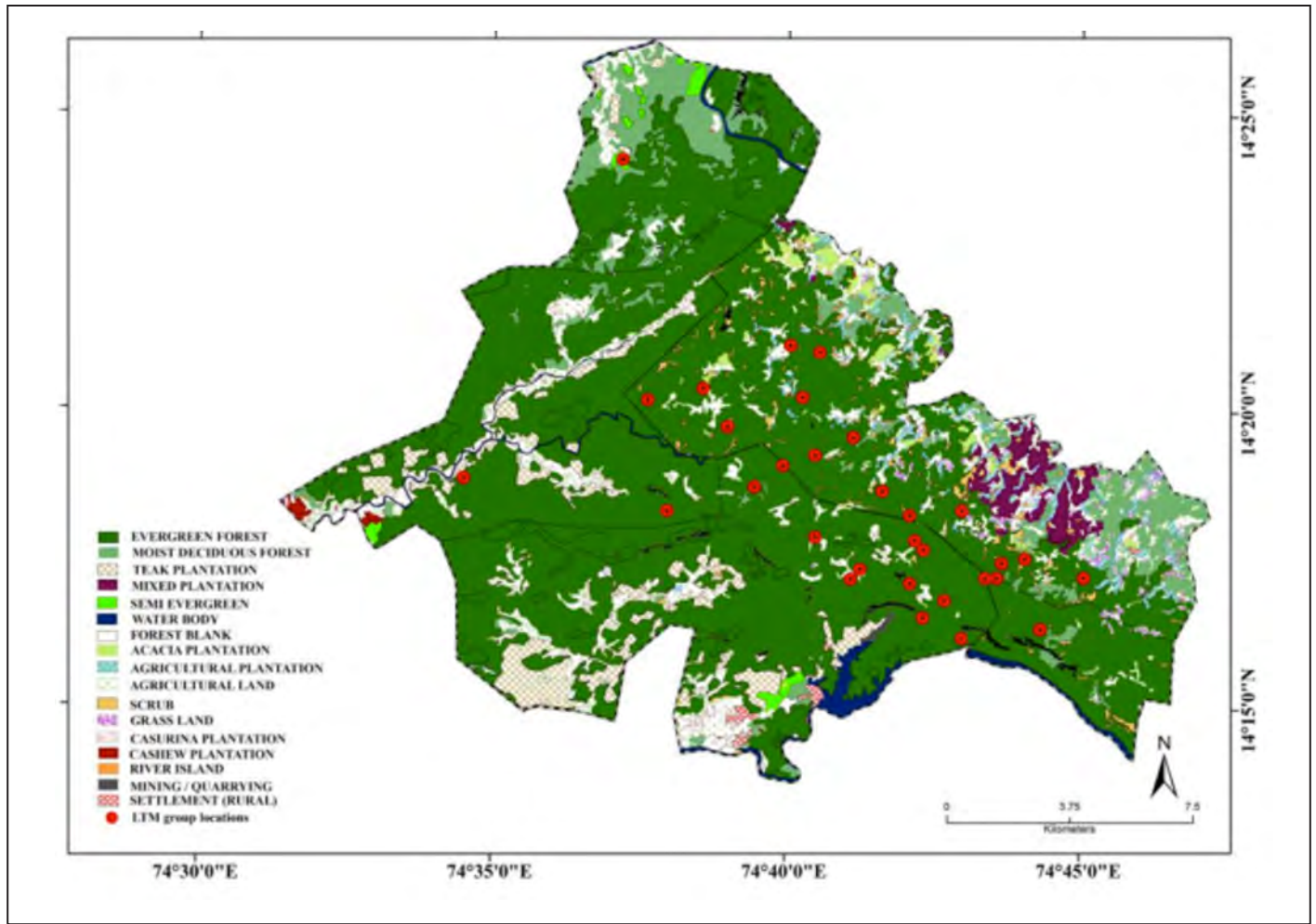


Figure 4. Forest types prevailing in the habitat of the lion-tailed macaque (*Macaca silenus*) in Sirsi-Honnavara forest divisions.

Management interventions against such threats should be taken seriously as an attempt to conserve the northernmost population of LTMs in its contiguous habitat.

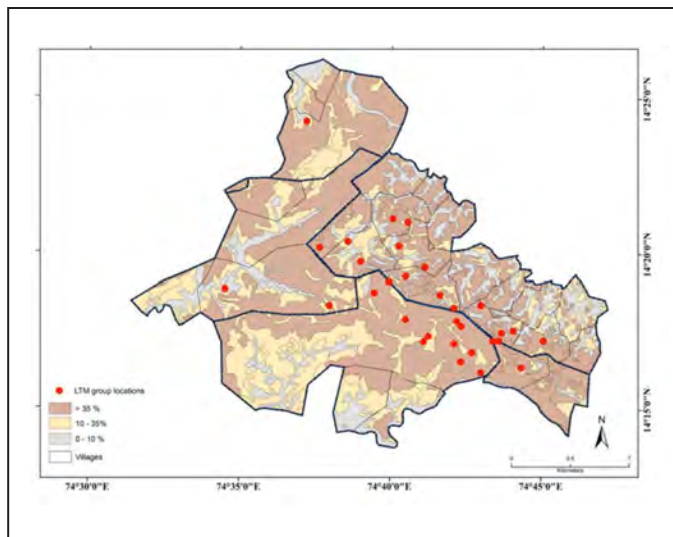


Figure 5. Slope gradients prevailing in Sirsi-Honnavara forest divisions with the locations of lion-tailed macaque (*Macaca silenus*) groups.

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