Primates in Peril

The World's 25 Most Endangered Primates 2014–2016

Edited by
Christoph Schwitzer, Russell A. Mittermeier, Anthony B. Rylands, Federica Chiozza, Elizabeth A. Williamson, Janette Wallis and Alison Cotton

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Illustrations by Stephen D. Nash

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Front cover photo: Chamba sacred langur (*Semnopithecus ajax*). Photo © WILD / Himalayan Langur Project

Back cover photo: Ecuadorian brown-headed spider monkey (*Ateles fusciceps fusciceps*). Tesoro Escondido, Esmeraldas, Ecuador. Photo © Juan de Dios Morales

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Acknowledgements

Here, we present the 2014–2016 iteration of the list of the World's 25 Most Endangered Primates, drawn up during an open meeting held during the XXV Congress of the International Primatological Society (IPS), Hanoi, 13 August 2014.

We have updated the species profiles from the 2012–2014 edition (Schwitzer *et al.* 2014) for those species remaining on the list, and added additional profiles for newly listed species.

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The World's 25 Most Endangered Primates: 2014–2016

Here we report on the eighth iteration of the biennial listing of a consensus of the 25 primate species considered to be among the most endangered worldwide and the most in need of conservation measures.

The 2014–2016 list of the world's 25 most endangered primates has five species from Africa, five from Madagascar, ten from Asia, and five from the Neotropics (Table 1). Madagascar tops the list with five species. Indonesia and Vietnam both have three, Brazil two, and Cameroon, China, Colombia, Côte d'Ivoire, the Democratic Republic of Congo, Ecuador, Ghana, India, Kenya, Nigeria, Peru, the Philippines, Sri Lanka, Tanzania and Venezuela each have one.

The changes made in this list compared to the previous iteration (2012–2014) were not because the situation of the eight species that were dropped (Table 2) has improved. In some cases, such as, for example, *Microcebus berthae*, the situation has in fact worsened, due to ongoing deforestation in this species' small distribution range in western Madagascar. By making these changes we intend rather to highlight other, closely related species enduring equally bleak prospects for their survival. One species for which the situation may have improved since it was first added to the list in 2008 is *Eulemur flavifrons*, Sclater's black lemur. While severe threats to this species remain in large parts of its range, some populations inside the Sahamalaza – Îles Radama National Park are now under more effective protection, mainly owing to a long-term research and monitoring programme that has been active in this protected area since 2004.

Eight of the primates were not on the previous (2012–2014) list (Table 3). Four of them are listed as among the world's most endangered primates for the first time. The Lac Alaotra bamboo lemur, Perrier's sifaka, the Hainan gibbon and the Sumatran orangutan had already been on previous iterations, but were subsequently removed in favour of other highly threatened species. The 2014–2016 list contains two members each of the genera *Piliocolobus*, *Trachypithecus*, *Semnopithecus* and *Ateles*, thus particularly highlighting the severe threats that large primates are facing in all of the world's primate habitat regions.

During the discussion of the 2014–2016 list at the XXV Congress of IPS in Hanoi in 2014, a number of other highly threatened primate species were considered for inclusion (Table 4). For all of these, the situation in the wild is as precarious as it is for those that finally made it on the list.

Table 1. The World's 25 Most Endangered Primates 2014–2016.

Africa		
Galagoides rondoensis	Rondo dwarf galago	Tanzania
Cercopithecus roloway	Roloway monkey	Côte d'Ivoire, Ghana
Piliocolobus preussi	Preuss's red colobus	Cameroon, Nigeria
Piliocolobus rufomitratus	Tana River red colobus	Kenya
Gorilla beringei graueri	Grauer's gorilla	DRC
Madagascar		
Cheirogaleus lavasoensis	Lavasoa Mountains dwarf lemur	Madagascar
Hapalemur alaotrensis	Lac Alaotra bamboo lemur	Madagascar
Varecia rubra	Red ruffed lemur	Madagascar
Lepilemur septentrionalis	Northern sportive lemur	Madagascar
Propithecus perrieri	Perrier's sifaka	Madagascar
Asia		
Carlito syrichta	Philippine tarsier	Philippines
Nycticebus javanicus	Javan slow loris	Indonesia (Java)
Simias concolor	Pig-tailed snub-nosed langur	Indonesia (Mentawai Is.)
Trachypithecus delacouri	Delacour's langur	Vietnam
Trachypithecus poliocephalus	Golden-headed or Cat Ba langur	Vietnam
Rhinopithecus avunculus	Tonkin snub-nosed monkey	Vietnam
Semnopithecus ajax	Chamba sacred langur	India
Semnopithecus vetulus nestor	Western purple-faced langur	Sri Lanka
Nomascus hainanus	Hainan gibbon	China
Pongo abelii	Sumatran orangutan	Indonesia (Sumatra)
Neotropics		
Ateles hybridus	Brown spider monkey	Colombia, Venezuela
Ateles fusciceps fusciceps	Ecuadorian brown-headed spider monkey	Ecuador
Cebus kaapori	Ka'apor capuchin	Brazil
Callicebus oenanthe	San Martín titi monkey	Peru
Alouatta guariba guariba	Northern brown howler	Brazil

Table 2. Primate species included on the 2012–2014 list that were removed from the 2014–2016 list.

Africa		
Piliocolobus pennantii	Bioko red colobus	Equatorial Guinea (Bioko Is.)
Madagascar		
Microcebus berthae	Madame Berthe's mouse lemur	Madagascar
Eulemur flavifrons	Sclater's black lemur	Madagascar
Propithecus candidus	Silky sifaka	Madagascar
Indri indri	Indri	Madagascar
Asia		
Tarsius pumilus	Pygmy tarsier	Indonesia (Sulawesi)
Pygathrix cinerea	Gray-shanked douc langur	Vietnam
Nomascus nasutus	Cao Vit or Eastern black-crested gibbon	China, Vietnam

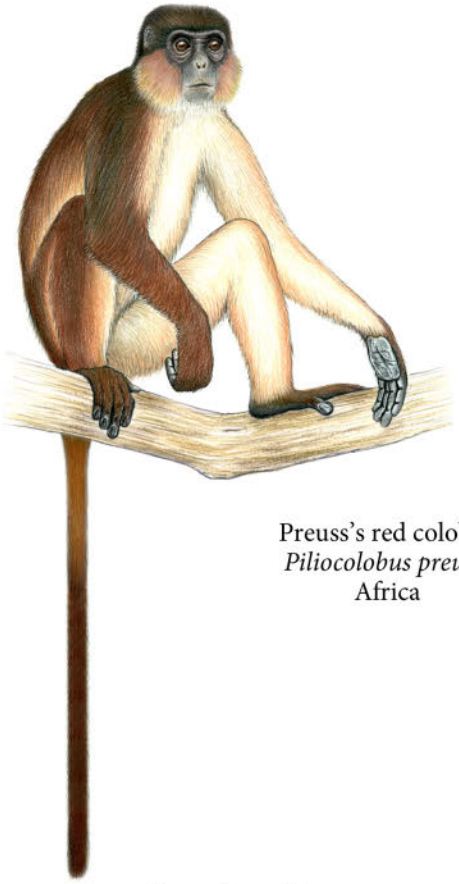
Table 3. Primate species that were added to the 2014–2016 list. The Lake Alaotra bamboo lemur, Perrier's sifaka, the Hainan black-crested gibbon and the Sumatran orang-utan were on previous lists. The other four species, marked with an asterisk, are new to the list.

Africa		
*Piliocolobus preussi**	Preuss's red colobus	Cameroon, Nigeria
Madagascar		
*Cheirogaleus lavasoensis**	Lavasoa Mountains dwarf lemur	Madagascar
Hapalemur alaotrensis	Lac Alaotra bamboo lemur	Madagascar
Propithecus perrieri	Perrier's sifaka	Madagascar
Asia		
*Carlito syrichta**	Philippine tarsier	Philippines
*Semnopithecus ajax**	Chamba sacred langur	India
Nomascus hainanus	Hainan gibbon	China
Pongo abelii	Sumatran orangutan	Indonesia (Sumatra)

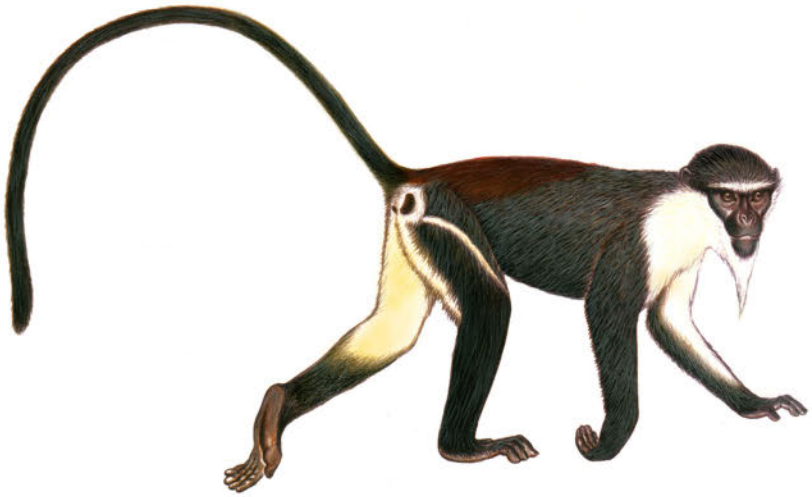
Table 4. Primate species considered during the discussion of the 2014–2016 list at the IPS Congress in Hanoi that did not make it onto the list, but are also highly threatened.

Africa		
Piliocolobus epieni	Niger Delta red colobus	Nigeria
Madagascar		
Cheirogaleus sibreei	Sibree's dwarf lemur	Madagascar
Lepilemur sahamalazensis	Sahamalaza sportive lemur	Madagascar
Daubentonia madagascariensis	Aye-aye	Madagascar
Asia		
Nycticebus coucang	Sunda slow loris	Indonesia, Malaysia, Singapore, Thailand
Loris tardigradus nycticeboides	Horton Plains slender loris	Sri Lanka
Trachypithecus hatinhensis	Hatinh langur	Lao PDR, Vietnam
Neotropics		
Cebus aequatorialis	Ecuadorian white-fronted capuchin	Ecuador, Peru

Photos of some of the Top 25 Most Endangered Primates. From top to bottom, left to right: 1. *Carlito syrichta* (photo by Russell A. Mittermeier); 2. *Nomascus hainanus* (juvenile)(photo by Zhao Chao); 3. *Ateles hybridus* (photo by Russell A. Mittermeier); 4. *Alouatta guariba guariba* (photo by Russell A. Mittermeier); 5. *Cercopithecus roloway* (photo by S. Wolters, WAPCA); 6. *Propithecus perrieri* (photo by Russell A. Mittermeier); 7. *Simias concolor* (juvenile)(photo by Richard Tenaza); 8. *Callicebus oenanthe* (photo by Russell A. Mittermeier); 9. *Varecia rubra* (photo by Russell A. Mittermeier); 10. *Lepilemur septentrionalis* (photo by Edward E. Louis, Jr.); 11. *Trachypithecus poliocephalus* (photo by Tilo Nadler); 12. *Gorilla beringei graueri* (photo by Russell A. Mittermeier); 13. *Pongo abelii* (photo by Russell A. Mittermeier); 14. *Galagoides rondoensis* (photo by Andrew Perkin); 15. *Hapalemur alaotrensis* (juvenile) (photo by Russell A. Mittermeier); 16. *Trachypithecus delacouri* (photo by Tilo Nadler).



Preuss's red colobus
Piliocolobus preussi
Africa

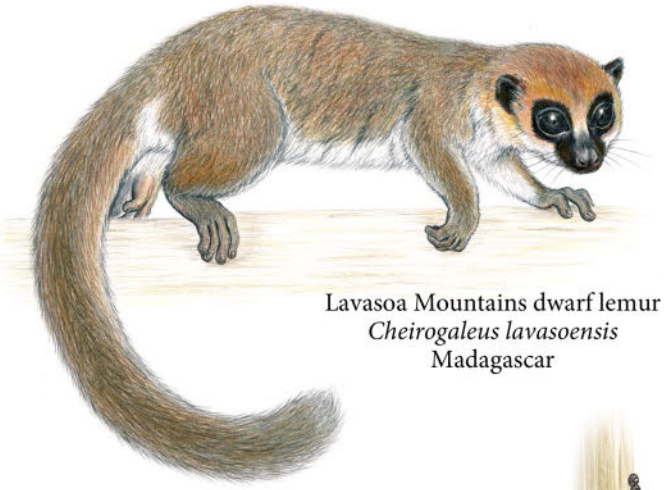


Roloway monkey
Cercopithecus roloway
Africa

Grauer's gorilla
Gorilla beringei graueri
Africa

Tana River red colobus
Piliocolobus rufomitratus
Africa

Rondo dwarf galago
Galagoides rondoensis
Africa



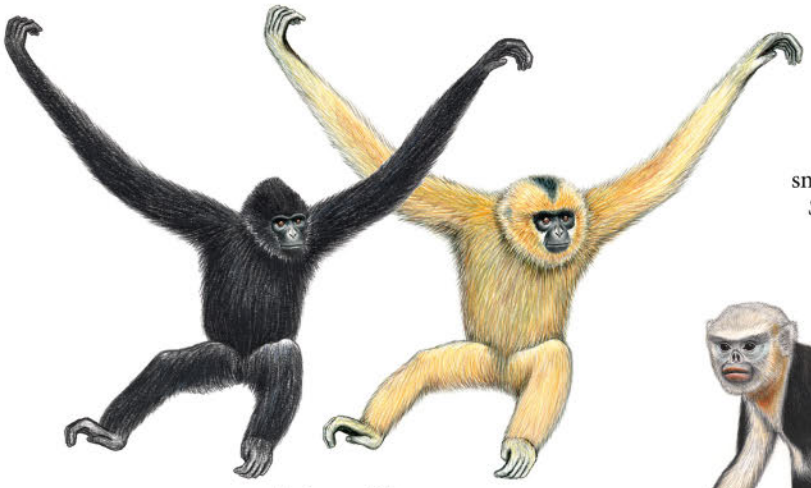
Lavasoa Mountains dwarf lemur
Cheirogaleus lavasoensis
Madagascar

Northern sportive lemur
Lepilemur septentrionalis
Madagascar

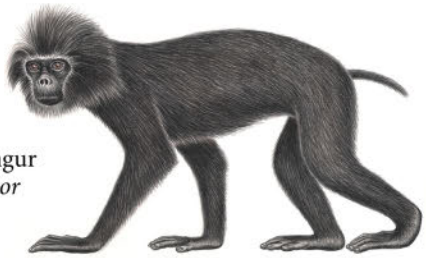
Perrier's sifaka
Propithecus perrieri
Madagascar

Lac Alaotra bamboo lemur
Hapalemur alaotrensis
Madagascar

Red ruffed lemur
Varecia rubra
Madagascar



Hainan gibbon
Nomascus hainanus
Asia



Pig-tailed
snub-nosed langur
Simias concolor
Asia

Tonkin snub-nosed
monkey
Rhinopithecus
avunculus
Asia

Chamba sacred langur
Semnopithecus ajax
Asia

Sumatran orangutan
Pongo abelii
Asia

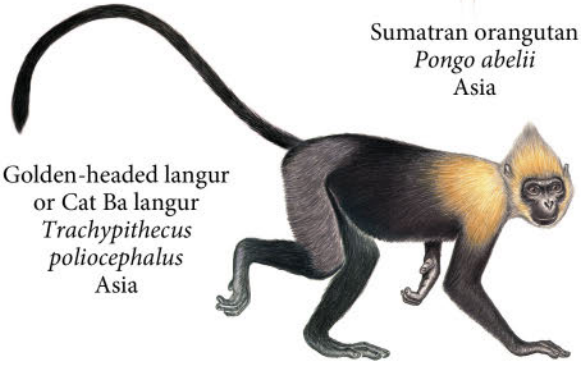
Javan
slow loris
Nycticebus
javanicus
Asia

Carlito
syrichta
syrichta

Philippine tarsier
Carlito syrichta
Asia

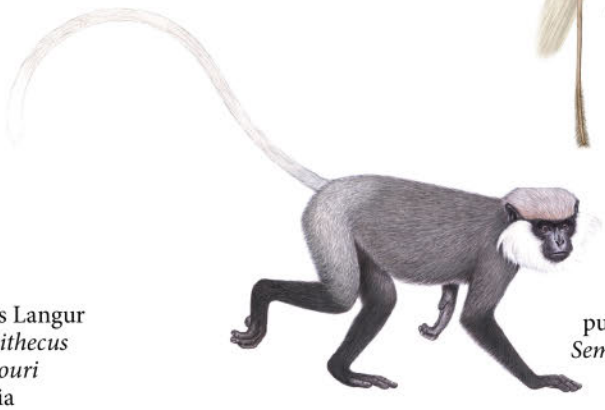
Carlito
syrichta
carbonarius

Carlito
syrichta
fraterculus



Golden-headed langur
or Cat Ba langur
Trachypithecus
poliocephalus
Asia

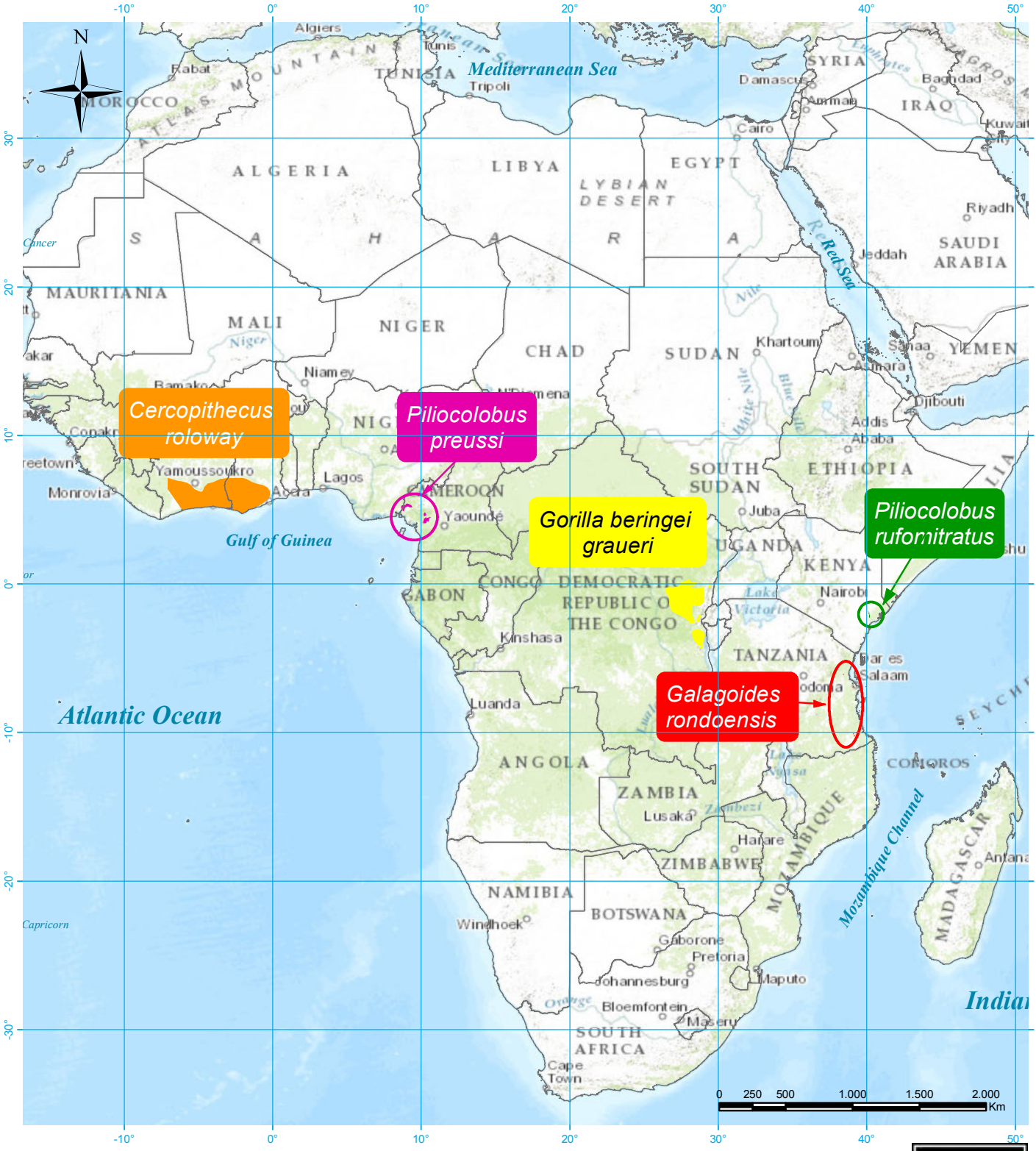
Delacour's Langur
Trachypithecus
delacouri
Asia



Western
purple-faced langur
Semnopithecus vetulus
nestor
Asia


Brown spider monkey
Ateles hybridus
Neotropics
light
variant
dark
variant
San Martín titi monkey
Callicebus oenanthe
Neotropics
dark variant
Northern brown howler
Alouatta guariba guariba
Neotropics
Ecuadorian brown-headed
spider monkey
Ateles fusciceps fusciceps
Neotropics
Ka'apor capuchin
Cebus kaapori
Neotropics

Africa



African Primates

- *Cercopithecus roloway*
- *Piliocolobus preussi*
- *Galagoides rondoensis*
- *Piliocolobus rufomitratus*
- *Gorilla beringei graueri*

IUCN/SSC PRIMATE Specialist Group

Rondo Dwarf Galago

Galagoides rondoensis Honess in Kingdon, 1997

Tanzania

(2012, 2014)

Andrew Perkin & Daphne Kerhoas

Weighing approximately 60 g, this is the smallest of all galago species (Perkin *et al*. 2013). It is distinct from other dwarf galagos in its diminutive size, a bottle-brush-shaped tail, its reproductive anatomy, and its distinctive "double unit rolling call" (Perkin and Honess 2013). Current knowledge indicates that this species occurs in two distinct areas, one in southwest Tanzania near the coastal towns of Lindi and Mtwara, the other approximately 400 km further north, above the Rufiji River, in pockets of forest around Dar es Salaam. One further population occurs in Sadaani National Park, approximately 100 km north of Dar es Salaam. Rondo dwarf galagos have a mixed diet of insects and fruit, often feed close to the ground, and move by vertical clinging and leaping in the shrubby understorey. They build daytime sleeping nests, which are often in the canopy (Bearder *et al*. 2003). As with many small primates, *G. rondoensis* is probably subject to predation by owls and other nocturnal predators. Among these, genets, palm civets and snakes invoke intense episodes of alarm calling (Perkin and Honess 2013).

Rondo dwarf galago (*Galagoides rondoensis*)
(Illustration: Stephen D. Nash)

Over the last decade, the status of *G. rondoensis* on the IUCN Red List has changed from Endangered in 2000 to Critically Endangered in 2008 (Perkin *et al*. 2008). In fact, based on a comparative ranking of the 27 primate species of Tanzania, the Taxon conservation score of *Galagoides rondoensis* was the second highest (7.13 out of 8; Davenport *et al*. 2014), thus, making this species one of particular conservation concern. It has an extremely limited and fragmented range in a number of remnant patches of Eastern African Coastal Dry Forest (*sensu* Burgess and Clarke 2000; p.18) in Tanzania, namely those at Zaraninge forest (06°08'S, 38°38'E) in Sadaani National Park (Perkin 2000), Pande Game Reserve (GR) (06°42'S, 39°05'E), Pugu/Kazimzumbwi (06°54'S, 39°05'E) (Perkin 2003, 2004), Rondo (NR) (10°08'S, 39°12'E), Litipo (10°02'S, 39°29'E) and Ziwani (10°20'S, 40°18'E) forest reserves (FR) (Honess 1996; Honess and Bearder 1996). New sub-populations were identified in 2007 near Lindi town in Chitoa FR (09°57'S, 39°27'E) and Ruawa FR (09°44'S, 39°33'E), and in 2011 in Noto Village Forest Reserve (09°53'S, 39°25'E) (Perkin *et al*. 2011, 2013) and in the northern population at Ruvu South Forest Reserve (06°58'S, 38°52'E). Specimens of *G. rondoensis*, originally described as *Galagoides demidovii phasma*, were collected by Ionides from Rondo Plateau in 1955, and Lumsden from Nambunga, near Kitangari, (approximately 10°40'S, 39°25'E) on the Makonde Plateau in Newala District in 1953. Doubts surround the persistence of this species on the Makonde Plateau, which has been extensively cleared for agriculture.

Surveys there in 1992 failed to detect any extant populations (Honess 1996). The areas most critical to their long-term conservation are Kazimzumbwi Forest Reserve (9 km²), Zaraninge Forest (20 km²) in Sadaani National Park, Pugu Forest Reserve (24 km²), and Rondo Forest Reserve (25 km²), eastern Tanzania (De Jong and Butynski 2012).

No detailed surveys have been conducted to assess population sizes of *G. rondoensis*. Distribution surveys have been conducted, however, in the southern (Honess 1996; Perkin *et al*. in prep.) and northern coastal forests of Tanzania (29 surveyed) and Kenya (seven surveyed) (Perkin 2000, 2003, 2004; Perkin *et al*. 2013). Absolute population sizes remain undetermined but recent surveys have provided estimates of density (3–6/ha at Pande Game Reserve (Perkin 2003) and 8/ha at Pugu Forest Reserve (Perkin 2004)) and relative abundance from encounter rates (3–10/hr at Pande Game Reserve and Pugu/Kazimzumbwi Forest Reserve (Perkin 2003, 2004) and 3.94/hr at Rondo Forest Reserve (Honess 1996)). There is a clear and urgent need for further surveys to determine population sizes in these dwindling forest patches.

In 2008, it was reported that the total area of forest in which *G. rondoensis* is currently known to occur does not exceed 101.6 km² (Pande GR: 2.4 km², Rondo FR: 25 km², Ziwani FR: 7.7 km², Pugu/Kazimzumbwi FR: 33.5 km², Litipo FR: 4 km², Zaraninge forest: 20 km², Chitoa FR: 5 km², and Ruawa FR 4 km²) (Minimum area data source: Burgess and Clarke 2000; Doggart 2003; Perkin *et al*. in prep.). New data on forest area change indicates that while two new sub-populations have been discovered; the overall area of occupancy hovers around 100 km². 2008 and 2014 forest-area estimations are as follows: Zaraninge 2008: 20 km², 2014: 15 km²; Pande 2008: 2.4 km², 2014: 2.4 km²; Pugu/Kazimzumbwe 2008: 33.5 km², 2014: 9 km²; Ruvu South 2008: 20 km², 2014: 10 km²; Ruawa 2008: 4 km², 2014: 4 km²; Litipo 2008: 4 km², 2014: 3 km²; Chitoa 2008: 4 km², 2014: 5 km²; Noto 2008: 21 km², 2014: 20 km²; Rondo 2008: 25 km², 2014: 25 km²; Ziwani 2008: 7.7 km², 2014: 1 km². The total forest area estimates are as follows: 2008 101.6 km²; 2014 94.4 km².

The major threat facing this species is loss of habitat. All sites are subject to some level of agricultural encroachment, charcoal manufacture and/or logging. All sites, except Pande (Game Reserve), Zaraninge (within Saadani National Park) and Rondo (Nature Reserve), are national or local authority forest reserves and as such nominally, but in practice minimally, protected. Since 2008, there have been changes resulting in the increase in protection of two forests. The Noto plateau forest, formerly open village land, is part of a newly created village forest reserve, and the Rondo Forest Reserve has now been declared a new Nature Reserve, both are important for Rondo galago conservation given their relatively large size. Given current trends in charcoal production for nearby Dar es Salaam, the forest reserves of Pugu and Kazimzumbwi have been predicted to disappear over the next 10–15 years (Ahrends 2005). Pugu/Kazimzumbwe as well as Ruvu South have seen continued and predicted losses to the rampant charcoal trade since Ahrends (2005) study. Pande, as a Game Reserve, is perhaps more secure, and Zareninge forest, being in a National Park, is the most protected part of the range of *G. rondoensis*. In the south, the Noto, Chitoa and Rondo populations are the most secure, as they are buffered by tracts of woodland. The type population at Rondo is buffered by woodland and *Pinus* plantations managed by the Rondo Forestry Project, and is now a Nature Reserve. Litipo, and Ruawa FRs are under threat from bordering village lands. Ziwani is now mostly degraded scrub forest, thicket and grassland.

Conservation action is urgently needed by monitoring rates of habitat loss, surveying new areas for remnant populations, estimating population size, reassessing the phylogenetic relationships of the sub-populations and increasing awareness. There is emerging data (vocal and penile morphology) that the northern and southern populations may be phylogenetically distinct with important taxonomic implications. As such the conservation of all sub-populations is important.

Across its known range, the Rondo galago can be found in sympatry with a number of other galagos, including two much larger species in the genus *Otolemur*: Garnett's galago *O. garnettii* (Least Concern, Butynski *et al*. 2008a), and the thick-tailed galago, *O. crassicaudatus* (Least Concern, Bearder 2008). The Rondo galago is sympatric with the Zanzibar galago, *Galagoides zanzibaricus* (Least Concern, Butynski *et al.* 2008b), in the northern parts of its range (for example, in Zaraninge forest, Pugu/Kazimzumbwi FR and Pande GR). In the southern parts of its range (for example, in Rondo, Litipo and Noto), the Rondo galago is sympatric with Grant's galago, *Galagoides granti* (Least Concern, Honess *et al*. 2008).

A new project to address these conservation and research issues has been implemented since 2012. Targeted conservation initiatives are taking place in Ruvu South FR, Chitoa FR and Noto VFR.

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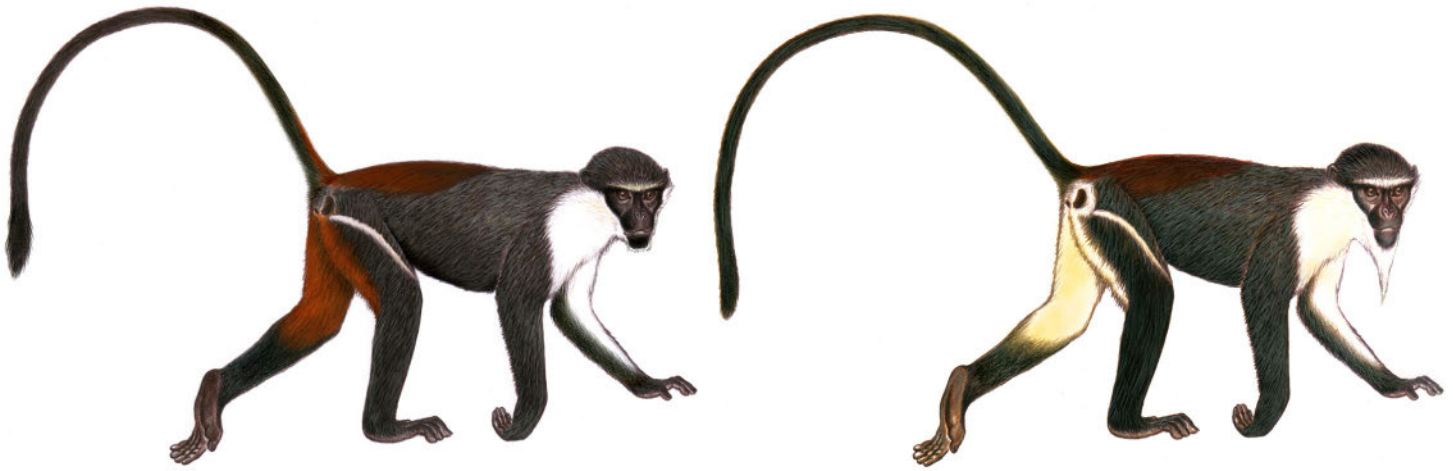
Roloway Monkey

Cercopithecus roloway (Schreber, 1774)
Ghana, Côte d'Ivoire
(2002, 2006, 2008, 2010, 2012, 2014)

W. Scott McGraw & John F. Oates



Roloway monkey (right) (*Cercopithecus roloway*) and Diana monkey (left) (*Cercopithecus diana*)
(Illustrations: Stephen D. Nash)

Considered by some to be subspecies of *Cercopithecus diana*, the Diana monkey and the roloway monkey are highly attractive, arboreal primates that inhabit the Upper Guinean forests of West Africa (Grubb *et al.* 2003). Groves (2001) considers the two subspecies to be sufficiently distinct to be regarded as full species. The roloway monkey is distinguished by its broad white brow line, long white beard and yellow thighs. Of the two forms, the roloway, which is known from Ghana and central and eastern Côte d'Ivoire, is more seriously threatened with extinction; it is rated as Endangered in the current IUCN Red List (Oates *et al.* 2008), but its status should be upgraded to Critically Endangered.

Roloway monkeys are upper-canopy specialists that prefer undisturbed forest habitat. Destruction and degradation of their habitat and relentless hunting for the bushmeat trade have reduced their population to small, isolated pockets. Miss Waldron's red colobus (*Procolobus badius waldroni*) once inhabited many of the same forest areas as the roloway, but is now almost certainly extinct (Oates 2011). Unless more effective conservation action is taken, there is a strong possibility that the roloway monkey will also disappear in the near future.

Over the last 40 years roloway monkeys have been steadily extirpated in Ghana. Several recent surveys have failed to confirm the presence of these monkeys in any reserves in western Ghana, including Bia National Park, Krokosua Hills Forest Reserve, Subri River Forest Reserve and Dadieso Forest Reserve (Oates 2006; Gatti 2010; Buzzard and Parker 2012; Wiafe 2013), although it is possible that the Ankasa Conservation Area still contains a few individuals (Magnuson 2003; Gatti 2010). Community-controlled forests along the Tano River (referred to as the "Kwabre Forest") in the far southwestern corner of the country are the only place in Ghana at which any roloways have been reported as seen by scientists or conservationists in the last decade. Kwabre consists of patches of swamp forest along the lower Tano River, adjacent to the Tanoé forest in Côte d'Ivoire. Surveys of these forests have been conducted under the auspices of the West African Primate Conservation Action organization since 2011, and several sightings of roloway groups have been made, along with mona monkeys, spot-nosed monkeys, white-naped mangabeys and olive colobus (WAPCA 2012, 2014; Dempsey 2014). WAPCA has launched a community-based conservation project with villages around these forests with the aim of establishing a Kwabre Community Resource Management Area. Meanwhile, further efforts should be made to ascertain whether any roloway monkeys still survive in the Ankasa, because this site has significant conservation

potential and roloways have been reported there in the relatively recent past.

In neighbouring Côte d'Ivoire, the Roloway guenon's status is perhaps even direr. Less than ten years ago roloways were known or strongly suspected to exist in three forests: the Yaya Forest Reserve, the Tanoé forest adjacent to the Ehy Lagoon, and Parc National des Îles Ehotilé (McGraw 1998, 2005; Koné and Akpatou 2005). Surveys of eighteen areas between 2004 and 2008 (Gonedelé Bi *et al*. 2008, 2012) confirmed the presence of roloways only in the Tanoé forest suggesting that the roloway monkey may have been eliminated from at least two forest areas (Parc National des Îles Ehotilé, Yaya Forest Reserve) within the last decade. Subsequent surveys carried out in southern Côte d'Ivoire suggest a handful of roloways may still survive in two forest reserves along the country's coast. On 21 June, 2011, Gonedelé bi Sery observed one roloway individual in the Dassioko Sud Forest Reserve (Gonedelé Bi *et al.* in review; Bitty *et al*. 2013). In 2012, Gonedelé Bi and E. A. Bitty observed roloways in Port Gauthier Forest Reserve, and in October 2013, Gonedelé Bi obtained photographs of monkeys poached inside this reserve, including an image purported to be a roloway. The beard on this individual appears short for a roloway, raising the possibility that surviving individuals in this portion of the interfluvial region may in fact be hybrids. In any case, no sightings of roloways have been made in the Dassioko Sud or Port Gauthier Forest Reserves since 2012, despite regular patrols there. These reserves are described as coastal evergreen forests and both are heavily degraded due to a large influx of farmers and hunters from the northern portion of the country (Bitty *et al*. 2013). Gonedelé Bi and colleagues, in cooperation with SODEFOR (Société de Développement des Forêts) and local communities, have organized regular foot surveys aimed at removing illegal farmers and hunters from both reserves; however, the most recent surveys (August 2015) revealed that a logging company (SIDB) has begun clearing a portion of the Port Gauthier reserve. Efforts are underway to work with SODEFOR at stopping logging and other illegal activities in these reserves (Gonedelé Bi 2015).

Thus, the only forest in Côte d'Ivoire where roloways are confirmed to exist is the Tanoé forest adjacent to the Ehy Lagoon. This wet forest also harbors one of the few remaining populations of white-naped mangabeys in Côte d'Ivoire. Efforts led by I. Koné and involving several organizations (CEPA, WAPCA) helped stop a large palm oil company from causing further habitat degradation, and a community-based conservation effort has helped slow poaching within this forest (Koné 2015). Unfortunately, hunting still occurs in Tanoé and the primate populations within it are undoubtedly decreasing (Gonedelé Bi *et al*. 2013).

As the potential last refuge for roloways and white-naped mangabeys, the protection of the Tanoé forest in Côte d'Ivoire and the adjacent Kwabre Forest in Ghana should be the highest conservation priority. By any measure, the roloway monkey must be considered one of the most critically endangered monkeys in Africa and appears to be on the verge of extinction (Oates 2011).

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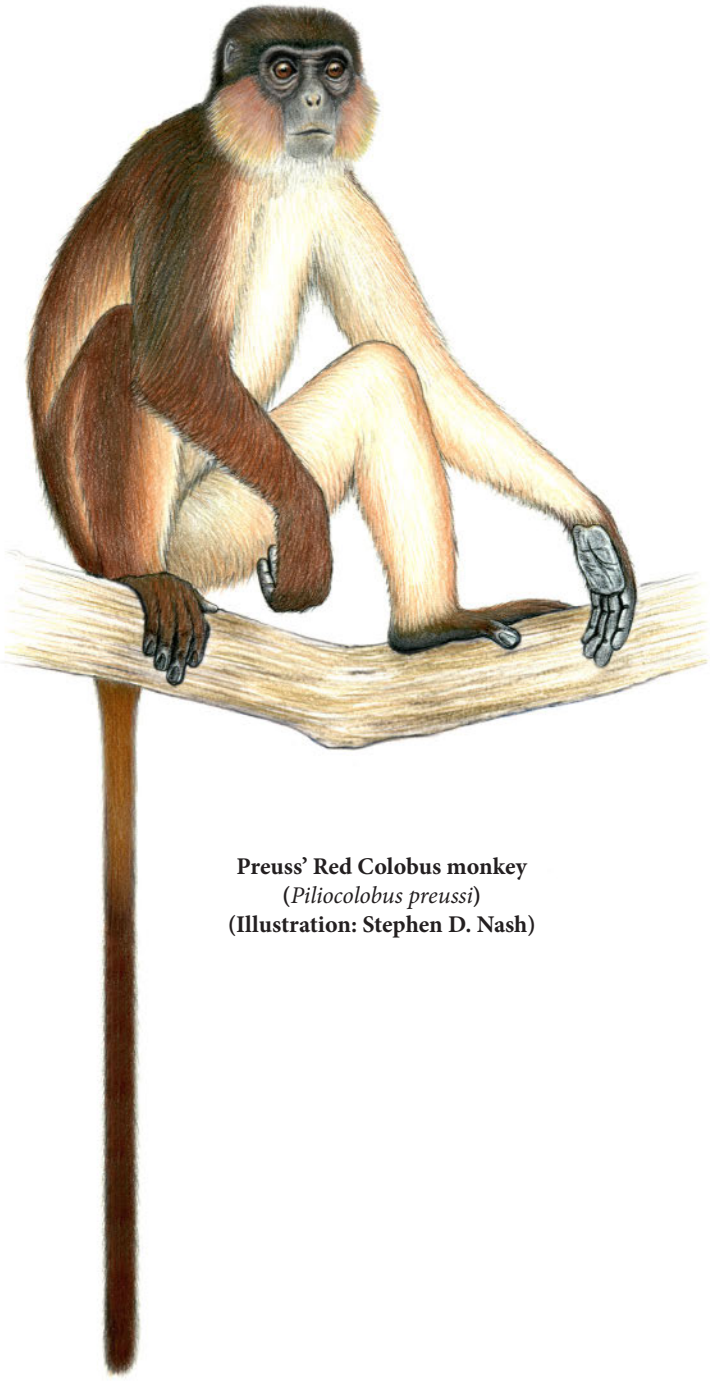
Preuss's Red Colobus

Piliocolobus preussi Matschie, 1900

Cameroon, Nigeria

(2014)

Joshua M. Linder, Bethan J. Morgan, John F. Oates, & Andrew Dunn



Preuss' Red Colobus monkey
(*Piliocolobus preussi*)
(Illustration: Stephen D. Nash)

Preuss's red colobus (*Piliocolobus preussi*) is endemic to western Cameroon and southeastern Nigeria where it is found in dense, moist, high canopy forests (Oates 2011). The taxonomic arrangement of this monkey has changed considerably in recent years; some classifications place it as a subspecies of *badius* or *pennantii* and others recognize it as a distinct species (Oates and Ting 2015). Mittermeier *et al.* (2013) place *preussi* in the genus *Piliocolobus*, following Groves (2007). Since 2008, the IUCN Red List has listed *P. preussi* as a Critically Endangered species.

Although a comprehensive assessment of the distribution and abundance of Preuss's red colobus has never been conducted, it is evident that populations of this monkey have disappeared from much of their original range since the beginning of the 20th century (Struhsaker 1999). The largest populations are now mostly found in Cameroon in the forests in and around Korup National Park (Linder and Oates 2011; Forboseh *et al.* 2007; Kupsch *et al.* 2014) and within the Ebo-Makombe-Ndokbou forest block (Morgan *et al.* 2013). In Nigeria, Preuss's red colobus is restricted to a small area of the Oban Division of Cross River National Park close to the boundary with Cameroon, and contiguous with Korup National Park; ranger patrols facilitated by the Wildlife Conservation Society confirmed its presence in this area in early 2015.

Although, as for other red colobus species, predation by chimpanzees may be a threat to the viability of some populations (Watts and Amsler 2013; Morgan *et al.* 2013), it is clear that for *P. preussi* the threats from bushmeat hunting and deforestation through logging, agriculture and infrastructure development are the major factors leading to its decline. Fa and colleagues (2006) conducted point-of-sale bushmeat surveys between August 2002 and January 2003 in the Cross-Sanaga region of Nigeria and Cameroon and estimated that 8,589 individual Preuss's red colobus monkeys were sold annually, over three-quarters of which originated in Cameroon. In Korup, bushmeat hunting appears to

be driving the decline of this species. Transect surveys conducted between 2001 and 2015 in southern Korup National Park indicate increasing hunting intensity and declining sighting frequency (groups sighted/km walked) of *Piliocolobus preussi* from 0.06 to 0.01 (Dunn and Okon 2003; Linder and Oates 2011; Linder unpublished data). *Piliocolobus preussi* is also becoming increasingly rare in northeast Korup, where sighting frequency of this species along transects has declined from 0.07 in 1990 (Edwards 1992) to 0.05 in 2004–2005 (Linder 2008) to 0.03 in 2014 (Robinson unpublished data). Hunter harvest surveys in Korup also suggest that the proportional representation of *P. preussi* has declined significantly between 1990 and 2005 (Linder and Oates 2011). Although temporal data on *P. preussi* abundance and distribution are lacking for the Ebo forest, recent *P. preuss*i encounter rates suggest that numbers are now very low.

Preuss's red colobus is one of the most endangered of all of the red colobus species, which are probably more threatened than any other taxonomic group of primates in Africa (Oates 1996; Struhsaker 2005, 2010). Elsewhere in the Gulf of Guinea region, the Bioko red colobus is now restricted to a very small area in the southwest of the island, where it is still hunted (Cronin *et al*. 2014), and the Niger Delta red colobus is in a precarious situation (Ikemeh 2015). On the other hand, a population of Bouvier's red colobus of Congo Republic, long feared to be extinct, was located in March 2015 in the Ntokou-Pikounda National Park (Devreese 2015).

To secure the long term conservation of *P. preussi*, we recommend the following actions: (i) bushmeat hunting in the forests of Cameroon's Korup National Park and the contiguous Oban Division of Cross River National Park in Nigeria must be drastically reduced through improved law-enforcement and community-based initiatives; (ii) the status of the Ebo forest in Cameroon should be upgraded to a national park and a results-driven law-enforcement regime implemented; (iii) field surveys are urgently needed to determine the current distribution and abundance of *P. preussi* outside of protected areas, and plans should be developed to work with local communities in these areas to help secure the populations. For example, the species was encountered in what is now known as the Ndokbou forest, north of the Ebo forest, in 2001, but since then no surveys have been conducted; it is not clear whether the species remains in this area, which is increasingly isolated from the Ebo forest due to industrial logging activities and acute pressure from the development of large-scale oil-palm plantations in the intervening area; and (iv) actions to raise the awareness of the existence of the species (although it may be known to hunters and bushmeat dealers the presence of the species remains largely unacknowledged by park authorities, at least within Nigeria).

Ultimately, conservationists must find ways to convince the Nigerian and Cameroon governments and local communities that Preuss's red colobus is worth saving.

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Tana River Red Colobus

Piliocolobus rufomitratus (Peters, 1879)
Kenya
(2002, 2004, 2006, 2008, 2012, 2014)

Thomas M. Butynski & Olivier Hamerlynck

Tana River red colobus (*Piliocolobus rufomitratus*)
(Illustration: Stephen D. Nash)

On the current IUCN Red List, the Tana River red colobus is presented as one of four assessed subspecies of *Procolobus rufomitratus* (i.e., as *P. r. rufomitratus*). The other three are Oustalet's red colobus *Procolobus r. oustaleti* (Trouessart, 1906), ashy red colobus *Procolobus r. tephrosceles* (Elliot, 1907), and Tshuapa red colobus *Procolobus r. tholloni* (Milne-Edwards, 1886). Here, however, we follow Groves (2001, 2005, 2007) and Groves and Ting (2013) in placing all red colobus in the genus *Piliocolobus*, and *rufomitratus* and the other three subspecies mentioned above as full species.

Gallery forests along the Lower Tana River, Kenya, are part of the East African Coastal Forests Biodiversity Hotspot. The Lower Tana River forests and some forest patches in the Tana Delta are the only habitat for two endemic primates; the Tana River red colobus and the Tana River mangabey, *Cercocebus galeritus* Peters, 1879. *Piliocolobus rufomitratus* is classified as 'Endangered' on the current IUCN Red List (Butynski *et al*. 2008a). *Cercocebus galeritus* is also classified as 'Endangered' (Butynski *et al*. 2008b).

Both the Tana River red colobus and the Tana River mangabey inhabit forest fragments (size range about 1–500 ha) along a 60-km stretch of the Lower Tana River (Butynski and Mwangi 1995; Mbora and Meikle 2004). In 2009, small populations of both species were discovered in the Tana Delta (Hamerlynck *et al*. 2012). The area of occurrence of the red colobus is <13 km², and that of the mangabey <26 km² (Butynski and Mwangi 1995). The population of the Tana River red colobus is about 1,000 individuals and declining (Butynski and Mwangi 1995; Karere *et al*. 2004). The population of the Tana River mangabey is roughly 2,000 individuals and also declining (Karere *et al*. 2004; Wieczkowski and Butynski 2013). Genetic analyses show that the effective population sizes of the two species are less than 100 individuals (Mbora and McPeek 2015).

There are six (perhaps seven) other species of nonhuman primate in the forests of the Lower Tana River, including the 'Vulnerable' Pousargues's monkey *Cercopithecus mitis albotorquatus* Pousargues, 1896, and 3–4 species of strepsirrhines (De Jong and Butynski 2012). Given the small area of remaining forest (<37 km^2), the serious threats (see below), and the occurrence of two endemic 'Endangered' primates, the forests of the Lower Tana River are the top priority for actions to conserve East Africa's primate diversity (De Jong and Butynski 2012; Butynski and De Jong in press).

Several factors render the long-term survival of the Tana River red colobus and Tana River mangabey bleak. First, forest continues to be degraded, cleared, and fragmented as a result of expanding agriculture and the taking of building materials and other products (Butynski and Mwangi 1995; Mbora and Meikle 2004; Moinde-Fockler *et al*. 2007; Duvail *et al*. 2012; Hamerlynck *et al*. 2012; Butynski and De Jong in press).

Second, there has been considerable alteration of river flow volume and the flood cycle by five hydroelectric power dams up-river (Butynski 1995; Maingi and Marsh 2002), and a sixth dam, the High Grand Falls Dam, is under construction (Hamerlynck *et al*. 2012). This will be the second largest dam in Africa and will be accompanied by large-scale irrigation schemes and water transfer to the 'Lamu Port and Lamu-Southern Sudan-Ethiopia Transport Corridor' (LAPSET). LAPSET is, potentially, the largest infrastructure project in Africa. Thus, the High Grand Falls Dam will have additional negative impacts on the floods and groundwater recharge required for the establishment and maintenance of the forests of the Lower Tana River and Tana Delta, on those people with flood-dependent livelihoods (crop farmers, livestock-raisers, and fisherman), and on the biodiversity of this biologically-sensitive region (Duvail *et al*. 2012). Construction of the US$ 2 billion High Grand Falls Dam is now a top priority of the Kenyan Government. Finance is from firms in the People's Republic of China and the Export-Import Bank of China.

Third, in January 2007, the High Court of Kenya ordered the annulment of the Tana River Primate National Reserve (TRPNR; 171 km^2) because the court found that the Reserve had not been established in accordance with the law. About half of the remaining forests gained some protection from the TRPNR. As a result of this court decision, no habitat of the Tana River red colobus, or of the Tana River mangabey, is legally protected at the present time.

Fourth, habitat loss has increased as a result of the failure of the Tana Delta Irrigation Project (TDIP) to protect forest. TDIP, financed by the Japan International Cooperation Agency (JICA), is a large rice-growing scheme managed by the Tana and Athi Rivers Development Authority (TARDA). The TDIP site had some of the most important forests for the red colobus and mangabey (Butynski and Mwangi 1994; Moinde-Fockler *et al*. 2007).

In 2012, the Tana Delta became a Ramsar Site. This internationally recognized protected area status was expected to help maintain, if not enhance, the conservation values of the Tana Delta, including those small forests in which Tana River red colobus and Tana River mangabey occur. The latest news, however, is that the most important of these forests, Onkolde (60 ha), has, since 2012, been decimated by mass loss of its dominant tree canopy species, the 'Vulnerable' *Oxystigma msoo*. This is probably the result of the lowering of groundwater caused by a TDIP embankment and, in 2015, by massive logging by people claiming to have "legal" documents (typically a euphemism for documents signed by people so high up that Kenya Forest Service staff on the ground dare not challenge them) (O. Hamerlynck pers. obs). As is often the case for Ramsar Sites, the various land-use plans for the Tana Delta have not been effectively implemented, and established user-rights and public interests have been disregarded (Butynski and De Jong in press, O. Hamerlynck and S. Duvail pers. obs).

Despite the troubles highlighted above, there is reason for hope for the Tana River forests and their primates. Partly galvanized by the participatory nature of research in the forests of the Lower Tana River, an organization called 'Ndera Community Conservancy' (NCC) has been established (Mbora and Allen 2011). The mission of this formally registered community-based organisation is to protect and conserve about half of the forest patches within the former TRPNR, and improve the viability of particular forest patches outside the former reserve. The NCC is working with government and NGO conservation initiatives and is making progress. For the NCC to make significant progress, however, the support of international conservation agencies is needed. With the NCC, government, and the international conservation community working together, the prospects for the long-term viability of the Tana River primates can be greatly improved.

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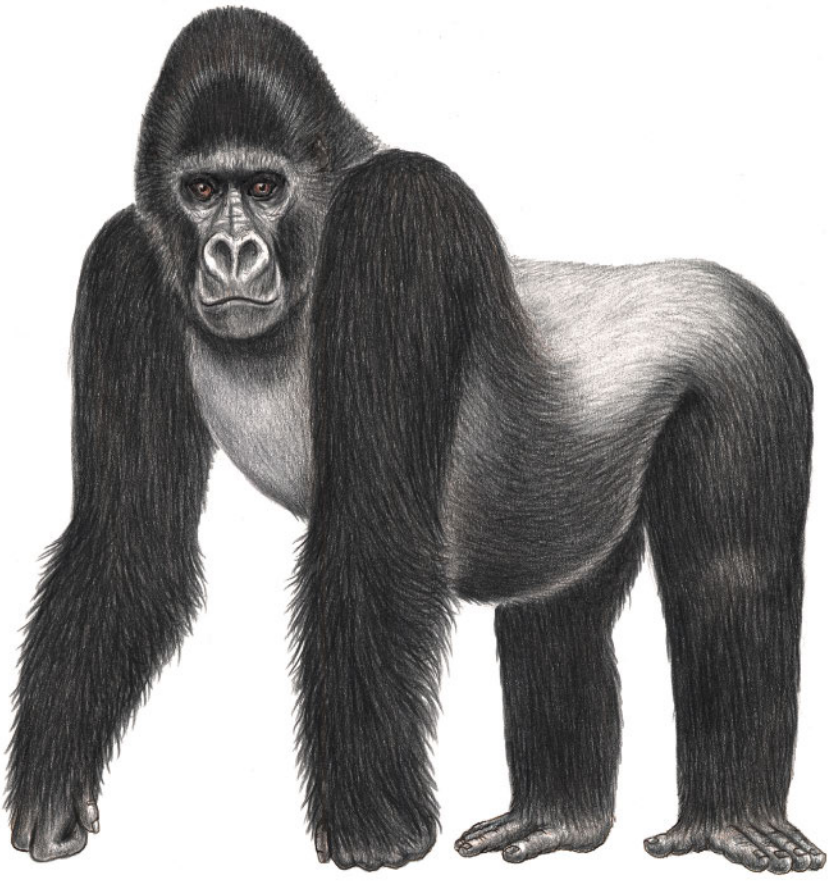
Grauer's Gorilla

Gorilla beringei graueri Matschie, 1914
Democratic Republic of Congo
(2010, 2012, 2014)

Stuart Nixon & Elizabeth A. Williamson



Grauer's gorilla (*Gorilla beringei graueri*) (Illustration: Stephen D. Nash)

Grauer's gorilla (*Gorilla beringei graueri*), one of two subspecies of eastern gorilla, is listed on CITES Appendix I and expected to be upgraded to Critically Endangered when the IUCN Red List is updated in 2016. It is endemic to the mixed lowland and montane forests of the Albertine Rift escarpment in eastern Democratic Republic of Congo (DRC), and although formerly known as the eastern lowland gorilla, the name is misleading as it ranges between 600 m and 2,900 m above sea level. The diet of Grauer's gorillas is rich in herbs, leaves, bark, lianas and vines, seasonally-available fruit, bamboo (at higher altitudes) and insects, and they show a preference for regenerating forests associated with abandoned fields and villages (Schaller 1963; Yamagiwa *et al*. 2005; Nixon *et al*. 2006).

The first surveys of Grauer's gorillas were conducted in 1959 (Emlen and Schaller 1960). This landmark study observed that eastern gorillas were rare west of the Great Lakes, had a highly discontinuous distribution and were severely threatened by hunting and habitat destruction. During the 1960s, through to the late 1980s, habitat conversion in the eastern part of their range intensified, destroying almost all montane forest outside of protected areas and exterminating a number of important high-altitude populations. Widespread killing of gorillas for bushmeat and in retaliation for crop raiding is likely to have impacted populations across their entire range during this period.

In the 1990s, efforts were made to determine the status of Grauer's gorilla in Maiko National Park (MNP; Hart and Sikubwabo 1994), Kahuzi-Biega National Park (KBNP) and adjacent forests (Hall *et al*. 1998a) and the Itombwe Massif (Omari *et al*. 1999). From these surveys, Hall *et al*. (1998b) concluded that Grauer's gorillas remained highly threatened across their range, and estimated the total number surviving to be approximately 16,900 individuals, with KBNP and MNP supporting the largest populations.

Threats to the gorillas intensified enormously throughout the 1990s and 2000s, due to persistent civil conflict in the eastern border regions of DRC. Refugees, internally displaced people and armed groups settled throughout the east of the country, putting enormous pressure on natural resources, national parks included. KBNP and MNP have been at the epicentre of this intense and illegal resource extraction for the past 20 years. The isolated KBNP highland population was decimated in the early 2000s (Amsini *et al*. 2008) following occupation by rebel forces. The status of gorillas in northern MNP is unknown as rebels control the illegal gold mines and consequently much of the park remains inaccessible. DRC's first community-managed protected areas—the Tayna Nature Reserve and the Kisimba Ikoba Nature Reserve (created in the mid 2000s specifically to protect their gorilla populations)—remain off limits due to land tenure disputes and occupation by several rebel groups.

The Congolese Institute for Nature Conservation (ICCN) faces continuing conflicts with armed groups, and highly dedicated ICCN personnel have been killed in the line of duty while attempting to protect Grauer's gorilla populations and their habitat. Destruction of forest for timber, charcoal production and agriculture continue to threaten the isolated gorilla populations that persist in the North Kivu highlands and Itombwe Massif, while poaching presents a serious and immediate threat to these gorillas across their entire range. Large numbers of military personnel stationed in rural areas and numerous rebel groups still active throughout the region have been implicated in illegal mining activities and have facilitated access to the firearms that fuel both the ongoing insecurity and an illegal bushmeat trade on a commercial scale. Since 2003, ICCN and partners have confiscated 15 Grauer's gorilla infants—casualties of poaching.

Conservation challenges are likely to increase as the DRC government continues its efforts to stabilize the east. Security will favour industrial extraction, large-scale agriculture and infrastructure. While development will increase the country's ability to support its population and participate in the global economy, it will also result in increased human settlement in forest areas critical for gorillas. Targeted conservation action in priority sites will be vital to slow further demise of this subspecies.

To address the critical situation faced by Grauer's gorillas, international and local NGOs are working with the government authorities to support protected area rehabilitation and reinforce conservation programmes. A conservation strategy with clear priorities for Grauer's gorillas has been published by IUCN (Maldonado *et al.* 2012). This action plan recognises four, broadly-defined population centres: Maiko-Tayna-Usala (including MNP and adjacent forests, Tayna Nature Reserve, Kisimba-Ikoba Nature Reserve and the Usala forest), Kahuzi-Kasese (including the lowland sector of KBNP and adjacent forests), KBNP highlands and the Itombwe Massif. In collaboration with ICCN, a consortium of NGOs led by the Wildlife Conservation Society (WCS) and Fauna & Flora International is completing a two-year project to assess the status of Grauer's gorilla across its range. Until the results of ongoing surveys are available, our best guesstimate from data collated during the past 14 years is that Grauer's gorilla numbers have been reduced to 2,000–10,000 individuals (Nixon *et al.* 2012). Further evidence for such a decline comes from an analysis of ape habitat across Africa, which estimates that suitable environmental conditions for Grauer's gorillas been have halved since the 1990s (Junker *et al.* 2012).

ICCN and partners made significant progress during 2014 and 2015, largely regaining control of KBNP and the southern sector of MNP and re-establishing a conservation presence. Significant gains have also been made in the Itombwe Nature Reserve, which will help protect the core of the Itombwe Massif and its highly fragmented gorilla population. Outside protected areas, regular community-based gorilla monitoring has been set up in Lubutu, Kasese and Biruwe/Nkuba, and may be expanded to other remote regions, such as the Usala forest. Confiscated gorilla orphans are now cared for at the Gorilla Rehabilitation and Conservation Education (GRACE) Centre near Lubero in North Kivu, and the possibility of reintroducing these gorillas at Mt. Tshiaberimu in Virunga National Park offers hope for small yet isolated subpopulations in well-protected sites. The steady recovery of the KBNP highland population

(WCS unpublished data) is encouraging evidence that highly-targeted conservation efforts can be successful even in the face of acute and sustained human pressures.

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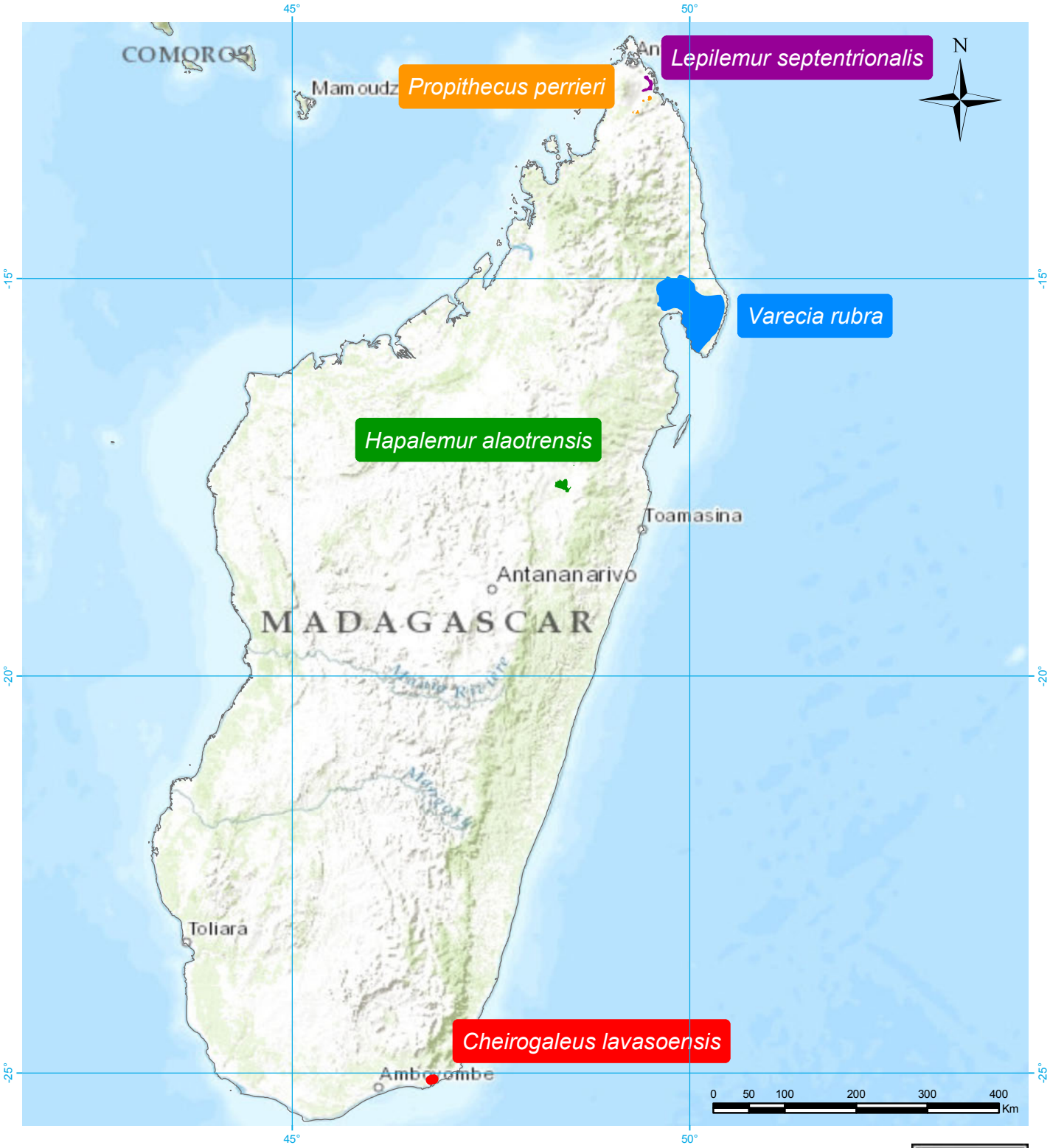
Madagascar



Madagascar Primates

- *Cheirogaleus lavasoensis*
- *Propithecus perrieri*
- *Lepilemur septentrionalis*
- *Hapalemur alaotrensis*
- *Varecia rubra*

IUCN/SSC PRIMATE Specialist Group

Lavasoa Mountains Dwarf Lemur

Cheirogaleus lavasoensis Thiele, Razafimahatratra and Hapke, 2013
Madagascar
(2014)

Andreas Hapke, Nary R. J. Andrianjaka & Dana Thiele

Lavasoa Mountains Dwarf Lemur (*Cheirogaleus lavasoensis*)
(Illustration: Stephen D. Nash)

The Lavasoa Mountains Dwarf Lemur, *Cheirogaleus lavasoensis*, was discovered in the Lavasoa-Ambatotsirongorongo Mountains in 2001 and first assigned to *Cheirogaleus crossleyi* (Hapke *et al*. 2005). Genetic data for comparison to other *Cheirogaleus* populations were largely lacking at that time and became available later (Groeneveld *et al*. 2009, 2010). Thiele *et al*. (2013) finally described *Cheirogaleus lavasoensis* as a new species. They assessed its distinctiveness based on analyses of mitochondrial and nuclear genetic data.

Cheirogaleus lavasoensis has a head-body-length of 22.2–28.5 cm and a body weight of 248–297 g. It has black eye rings, dark furry ears, and a darkly pigmented, pointed nose. The coloration on the crown, forehead, and neck is intensely reddish-brown and changes gradually to grey-brown towards the tail. The light creamy ventral coloration extends into a sharply delimited lateral stripe on the neck (Thiele *et al*. 2013). *Cheirogaleus lavasoensis* is nocturnal. Data about its life-history, ecology, and behavior are not available.

The Lavasoa-Ambatotsirongorongo Mountains are situated south of the north-southward directed Anosy- and Vohimena Mountain chains in extreme southern Madagascar. These mountains act as a climatic barrier between rainforest on their eastern flanks and dry spiny bush in their western rain shadow. This sharp ecological rupture is contrasted by a wide climatic gradient from dry spiny bush over transitional forest into humid littoral forest south of the mountain chains. The Lavasoa-Ambatotsirongorongo Mountains are situated in the center of this gradient. They are surrounded by water and lowlands and isolated from the Anosy Mountains by a plain at 20–30 m above sea level. The three main summits are aligned in west eastern direction: Grand Lavasoa (823 m), Petit Lavasoa (617 m), and Ambatotsirongorongo (438). A topographic map based on aerial photographs from 1957 (Foiben-Taosarintanin'i Madagasikara, 1979) displays one large, continuous forest on their southern and eastern flanks. This forest was isolated from the huge, continuous humid forest of the Anosy Mountain Chain by the forestless northern flank of the Lavasoa-

Ambatotsirongorongo Mountains and the forestless plain. Most of this forest has disappeared and only small fragments remain on the southern flanks of the three main summits. Their floristic composition has been characterized as a predominantly humid, transitional mixture (Andrianarimisa *et al*. 2009; Ramanamanjato *et al*. 2002).

Until recently, three of these forest fragments were the only known habitat of *Cheirogaleus lavasoensis* (Thiele *et al*. 2013). The three forest fragments had sizes of 50 ha, 30 ha, and 25 ha in 2009 (Andrianarimisa *et al*. 2009). Extensive fieldwork in nearby transitional, humid, and littoral forest yielded no evidence for the occurrence of the species (Hapke *et al*. 2005; Hapke, Gligor, and Andrianjaka pers. obs). Lei *et al*. (2014) reported the occurrence of *Cheirogaleus lavasoensis* at Kalambatritra, 170 km north of the Lavasoa-Ambatotsirongorongo Mountains at an altitude of approximately 1200 m. The Kalambatritra forest is Madagascar's westernmost rainforest and situated on the islands west-eastern drainage divide in an elevation range of 1200–1680 m (Irwin *et al*. 2001). It is separated from the eastern humid forest by 16 km of open grassland and extends over an area with a north-south diameter of approximately 60 km.

Thiele *et al*. (2013) reported data from 17 individuals of *Cheirogaleus lavasoensis* that had been captured and released between 2001 and 2006. The habitat of the species in the Lavasoa-Ambatotsirongorongo Mountains is biogeographically isolated, small, and fragmented. A survey in 2015 confirmed the persistence of *Cheirogaleus lavasoensis* in this area but also a further decline of forest cover (Andrianjaka and Hapke unpublished). The forest of Kalambatritra could harbour a considerably greater population of *Cheirogaleus lavasoensis*. At present, it is not possible to estimate its size because Lei *et al*. (2014) report data from a single individual that they could assign to the species. With respect to the differences in latitude and elevation, it is probable that there are strong ecological differences between Kalambatritra and the Lavasoa-Ambatotsirongorongo Mountains. Further research is necessary to gather information about the population size at Kalambatritra and about possible local adaptations of *Cheirogaleus lavasoensis* in the two areas. The population at Ambatotsirongorongo thus deserves high conservation priority, at least until more data become available about the population at Kalambatritra.

The entire remaining forest in the Lavasoa-Ambatotsirongorongo Mountains is situated within the Nouvelle Aire Protégée Ambatotsirongorongo. The greatest threats for the survival of *Cheirogaleus lavasoensis* in this area are further habitat loss and degradation. The main pressures are wood extraction, slash-and-burn cultivation, and accidental fires, which are frequent due to the practice of regular grassland burning. The forest fragments inhabited by *Cheirogaleus lavasoensis* are apparently remnants of the most humid patches within a formerly larger, continuous forest. Until recently, the local population relied on dryer forest remnants as main sources of construction timber and firewood. These fragments have now mostly disappeared, which generates the risk of increasing pressure onto the remaining habitat of *Cheirogaleus lavasoensis*. At least two forest fragments inhabited by *Cheirogaleus lavasoensis* harbor sources that are used by the local population for paddy irrigation and drinking water. A successful conservation program should thus integrate water management, reforestation and habitat restoration.

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Lac Alaotra Bamboo Lemur
Hapalemur alaotrensis Rumpler, 1975
Madagascar
(2000, 2014)

Fay Clark and Lance Woolaver

Hapalemur alaotrensis exclusively inhabits the dense papyrus and reed beds surrounding Lac Alaotra, Madagascar's largest lake, located on the western edge of the eastern rain forest region (Mutschler and Feistner 1995). The species occurs as two subpopulations, a small fragmented one in the northern part of the lake around the Belempona Peninsula and a larger one in contiguous marshland along the lake's southwestern shore. Its entire range appears to be less than 5,800 ha and it occurs only up to elevations of 750 m. The lake is surrounded by a vast wetland area consisting of 14,000 ha of marshes and 175,000 ha of rice fields, and constitutes an important biodiversity area (Pidgeon 1996). Guillera-Arroita *et al*. (2010) reported that difficulties in monitoring this species arise because the wetland is difficult to survey. It can only be accessed via canals cut by fishermen, resulting in limited transects for monitoring, and visibility is typically restricted to a few metres.

The Alaotra watershed is of international importance under the Ramsar Convention of 2003. The government of Madagascar also recognized the conservation value of this area by classifying it as a Temporary New Protected Area within national law N°381-2007/ MINENVEF/MAEP in January 2007, which was given permanent protection status in July 2015. However, the human population in the Alaotra watershed has rapidly increased in the last few decades, from 109,000 in 1960 to approximately 550,000 in 2003 (Pidgeon 1996; PRD 2003). People rely heavily on rice cultivation and fishing for their livelihoods (Andrianandrasana *et al*. 2005; Copsey *et al*. 2009a; Wallace *et al*. 2015), leading to severe loss, degradation and fragmentation of the Alaotran marshes (Mutschler *et al.* 1995, 2005). Conversion of marsh habitat to rice fields has been the most significant historical and continuing threat to the survival of *H. alaotrensis*. From 2001–2007 there was a decrease in the coverage of marsh vegetation of 29.7%, from 19,000 to 14,000 ha (Durrell Wildlife Conservation Trust unpublished data). Marsh burning inhibits the regeneration of *H. alaotrensis* habitat, which also declined during that same period from 9,400 to 5,800 ha.

Lac Alaotra bamboo lemur (*Hapalemur alaotrensis*) (Illustration: Stephen D. Nash)

Unfortunately, marsh burning increased considerably in 2013 (3,000 ha) and 2014 (2,600 ha) due to a lack of law enforcement and an increase in coordinated pressure by people in power organising conversion of the marsh to ricefields for their own financial benefit (Ratsimbazafy *et al.* 2013). Some of this burned marsh will regenerate if not converted to rice but the trend in annual burning has accelerated. Hunting for food and capture for pets has significantly reduced lemur numbers in the past but has been mitigated somewhat in more recent years due to intervention by conservation organizations (Razafimananahaka *et al*. in prep). Various methods

of hunting and trapping are employed by local people. Direct pursuit by dogs is the most common, but they may also be captured by using a harpoon, a snare, a stick to knock them out or into the water, by burning their reed bed habitat or just by chasing them down (Copsey *et al*. 2009a,b). Commercial drainage projects represent a potential threat. Regular burning to increase cattle pasture and facilitate local fishing reduces suitable lemur habitat and also promotes the invasion of exotic plant species that may choke the remaining marshes.

According to the most recent IUCN Red List assessment, *H. alaotrensis* is now Critically Endangered (Andriaholinirina *et al*. 2014). Their numbers have decreased from >10,000 individuals in 1994 (Mutschler and Feistner 1995) to <3,000 in 2002 (Ralainasolo 2004). The most recent population estimates for *H. alaotrensis* range from 2,500–5,000 individuals, representing a decline of approximately 30% in just over a decade (Ralainasolo 2004; J. Ratsimbazafy, unpublished data). Existing habitat is so fragmented that gene flow between existing populations is greatly reduced, affecting the long-term viability of populations (Ralainasolo *et al*. 2006). Thanks to efforts of the Durrell Wildlife Conservation Trust and local community associations, the Lac Alaotra New Protected Area was recently granted permanent protection status, providing an official legal framework for the co-management of the Protected Area by local communities living around the wetland. This includes both a strict conservation area of 8,000 ha, and a surrounding 5,200 ha zone where controlled activities (e.g. fishing) are permitted. In addition, public awareness campaigns continue to focus on the benefits of habitat conservation to the half million or more people who live by the lake-erosion control, the biological filtering of agricultural pollutants, and flood prevention. *Hapalemur alaotrensis* is currently being used as a flagship species by Madagascar Wildlife Conservation (MWC), where the economic benefits from its 'Camp Bandro' (Bandro is the vernacular word for *H. alaotrensis*) are invested back into community development (Ratsimbazafy *et al*. 2013; Rendigs *et al.* 2015).

More work is needed to secure the future of *H. alaotrensis*, a unique wetland lemur experiencing severe conflict with humans (Waeber *et al*. in press). Ratsimbazafy *et al.* 2013 suggest more stringent policy and management mechanisms to halt marsh conversion around the lake, and the general need to improve the socio-economic status of local people in order for conservation measures to be effective. The effective management of the New Protected Area (NAP) will provide the legal framework by which marsh protection and wise management of the lake and marshes can be linked to sustainable rural development that has a positive impact on the environment. Immediate actions to ensure that the New Protected Area is operational include the physical delimitation of the NAP and capacity-building and support for the management structure of the NAP which will be co-managed by local communities and resource users around Lac Alaotra. A more effective system for censusing *H. alaotrensis,* potentially using drone technology, is required to improve monitoring and refine annual population estimates (Guillera-Arroita *et al*. 2010).

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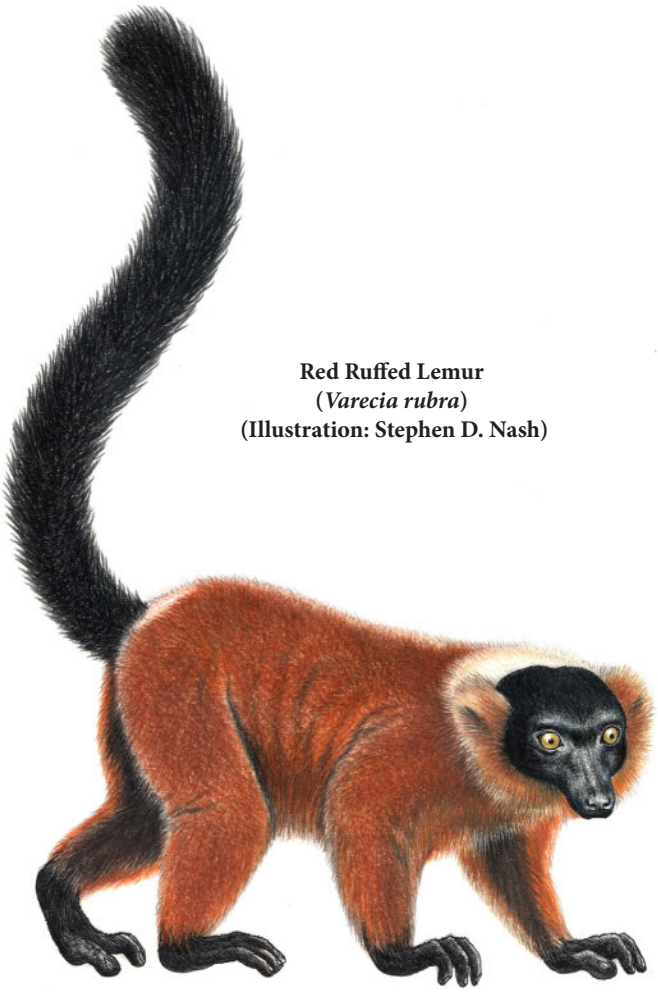
Red Ruffed Lemur

Varecia rubra (É. Geoffroy Sainte-Hilaire, 1812)

Madagascar

(2012, 2014)

Christoph Schwitzer, Russell A. Mittermeier, Edward E. Louis Jr. & Matthew Richardson



Red Ruffed Lemur
(*Varecia rubra*)
(Illustration: Stephen D. Nash)

The red ruffed lemur is confined to the Masoala Peninsula and the region immediately north of the Bay of Antongil in northeastern Madagascar (Petter and Petter-Rousseaux 1979; Tattersall 1982). It may have occurred as far north as Antalaha in the past, but this is not certain (Tattersall 1977). The Antainambalana River appears to separate it from *V. variegata subcincta*, and recent surveys have shown that the westernmost distribution of *V. rubra* is near the confluence of the Antainambalana and Sahantaha rivers (Hekkala *et al.* 2007). Variations in colour pattern are well known in this species, but have not been attributed to clear geographic regions. It may intergrade with *V. variegata subcincta*; the confluence of the Vohimara and Antainambalana rivers has been investigated as a possible contact or hybrid zone between the two, but without conclusive results (Tattersall 1982; Lindsay and Simons 1986; Vasey and Tattersall 2002; Hekkala *et al.* 2007).

With a head-body-length of 50–55 cm and a body mass of 3.0–3.6 kg (Vasey 2003), *Varecia rubra* is a large member of the Lemuridae. It inhabits primary and some secondary moist lowland forest (up to 1200 m above sea level) and prefers tall forest, where it is often observed in the crowns of large feeding trees. The species feeds mainly on fruit, supplemented with flowers, nectar, and leaves. In one study conducted between May and November (Rigamonti 1993), red ruffed lemurs fed on ripe fruits for 73.9% of their feeding time, flowers for 5.3%, and leaves for 20.9% (18.3% of these mature). Only a few plant species were used as food resources: 72.5% of the observed feeding bouts occurred in only seven tree species. The animals fed on 42 plant species altogether, compared to 106 species that would have been available to them in their home range area. The composition of the diet varied from month to month, but fruits were consistently the main item, even when they were hard to find. The core areas used within their territories always correlated with large, fruit-bearing trees. In the cold-wet season, when few fruits are available, groups split up into subgroups to use different core areas. Females are reported to eat more low-fibre, high-protein items (young leaves and flowers) prior to giving birth and during lactation, presumably to meet the higher energy demands of reproduction (Vasey 2000a, 2002). At Andranobe, 132 different plant species from 36 families were eaten over the course of a year (Vasey 2000b).

This species has been studied in the forests of Ambatonakolahy (Rigamonti 1993) and Andranobe (Vasey 1997a) on the Masoala Peninsula. Social organization is described as fission/fusion, and communities are usually multimale-multifemale and number 5–31 individuals. Home ranges cover 23–58 ha and appear to be defended (Rigamonti 1993; Vasey

2006). In one study at Andranobe, *V. rubra* spent 28% of its time feeding, 53% resting, and 19% traveling. Females fed more and rested less than males (Vasey 2005). The species is most active during the hot rainy season. Mating occurs in early July, and infants are born in October and fully weaned by February (Vasey 2007).

The red ruffed lemur is classified as Critically Endangered (Andriaholinirina *et al.* 2014) based on a suspected population reduction of ≥80% over a 3-generation time period of 24 years. The principal threats to the species are habitat loss and hunting (Simons and Lindsay 1987; Rigamonti 1996; Vasey 1996, 1997b; Borgerson 2015). Because of their large size and evident need for tall primary forest, these animals are particularly susceptible to human encroachment (Borgerson 2015), and hunting and trapping for food still takes place. Furthermore, remaining populations are concentrated on the Masoala Peninsula, where they are threatened by the frequent cyclones that hit this part of Madagascar. The only protected area where *Varecia rubra* is known to occur is Masoala National Park (Kremen 1998). Masoala was the national park most affected by the very rapid upsurge of illegal logging after the political events of early 2009, and this logging continued well into 2010. Population density has been variously estimated at 6 individuals/km² (Rakotondratsima and Kremen 2001; Borgerson 2015 [at an unnamed village site]), 17 individuals/km² (Borgerson 2015 [at an unnamed forest site away from villages]), 21–23 individuals/km² in Ambatonakolahy (Rigamonti 1993), and 31–54 individuals/km² in Andranobe (Vasey 1997b).

The IUCN lemur conservation strategy 2013–2016 (Schwitzer *et al.* 2013) proposes a suite of conservation measures for Masoala National Park to ensure the conservation of the red ruffed lemur: further patrols and surveillance; campaigns of environmental education and awareness; and support for small-scale husbandry of domestic animals as a source of protein. As of 2015, there were 600 red ruffed lemurs reported in captivity worldwide (ISIS 2015). Such populations in American and European zoos represent a safeguard against extinction, but they are unfortunately very limited in their genetic diversity (Schwitzer 2003).

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Northern Sportive Lemur

Lepilemur septentrionalis Rumpler and Albignac, 1975

Madagascar

(2008, 2010, 2012, 2014)

Edward E. Louis Jr. & John R. Zaonarivelo

Originally described based on cytogenetic and morphometric characteristics (Rumpler and Albignac 1975), the taxonomic status of the northern sportive lemur (*Lepilemur septentrionalis*) has since been supported by more detailed cytogenetic, morphogenetic and especially molecular data (Ravoarimanana *et al*. 2004; Andriaholinirina *et al*. 2006; Louis *et al*. 2006), and subsequently accepted in recent taxonomic revisions of primates (Groves 2001, 2005) and lemurs (Mittermeier *et al*. 2008; Mittermeier *et al*. 2010). With 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 Antsiranana. The number of animals observed during surveys has decreased dramatically over the past ten years. The first significant survey of the northern sportive lemur was performed in 2001 by I. Ravoarimanana, following by another one in 2007 by A. Zaramody in the Andrahona, Ankarakataova, and Sahafary regions. The population was then estimated at about 120 individuals, mainly concentrated in the Sahafary area.

Northern sportive lemur (*Lepilemur septentrionalis*)
(Illustration: Stephen D. Nash)

Expeditions by Omaha's Henry Doorly Zoo and Aquarium (OHDZA) and the Madagascar Biodiversity Partnership (MBP) in 2010 and 2011 verified the continued existence of the northern sportive lemur, albeit noting a tremendous decline in the Sahafary classified forest. Furthermore, the surveys did not detect a single animal in the Analalava forest where it had previously been seen in 2005. A follow-up survey in July 2012 to Analalava did however document one individual (Ranaivoarisoa *et al*. 2013). Fortunately, Ranaivoarisoa *et al*. (2013) also confirmed the presence of the northern sportive lemur in Montagne des Français (MDF) in 2010, but could only identify 19 individuals across its range based on capture and direct visual observations. Further surveys of the Montagne des Français region in 2012–2013 by OHDZA and MBP, including the previously known habitats of Sahafary and Analalava classified forests through to its northern extent in MDF, documented only 52 *L. septentrionalis* individuals, with 95% of these lemurs located in MDF. The most recent population estimates based on only capture surveys in 2013 in the Montagne des Français area provided the following population estimates: 1) Abatoire - 7 individuals; 2) Andranonakomba - 2 individuals; 3) Ampamakiampafana - 11 individuals; 4) Ambatobe - 2 individuals; and Berambo - 5 individuals. This is a total of 27 individuals. The species was also documented at Anketrakala and Ampitsinjozatsambo in 2012, which were not recently surveyed. Surveys in the region continue to estimate the population around 50 individuals in 2015.

In 2008, the Service d'Appui à la Gestion de l'Environnement 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. However, sustained human encroachment from the city of Antsiranana continues to finance the production of charcoal and collection of sand, activities that are threatening this last remaining northern sportive lemur population (Ranaivoarisoa *et al*. 2013). Thus, habitat loss from uncontrolled long-term charcoal practices is the primary challenge to overcome. The northern sportive lemur is nocturnal, spending the day sleeping in tree holes, and very little is known about its ecology and behaviour. However, recent work has shown that its folivorous diet and predilection for new-growth leaves complicates any attempts or plans to maintain it in captivity. Currently, there is no record of any 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 programs and community-based interactions are, therefore, the only viable solutions. The combination of a very small range composed of rapidly deteriorating suitable habitat with high pressure from charcoal production continues to place the Critically Endangered northern sportive lemur (Andrainarivo *et al*. 2011) on the cusp of extinction.

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Perrier's Sifaka
Propithecus perrieri Lavauden, 1931
Madagascar
(2000, 2002, 2004, 2014)

Matthew A. Banks, Erik R. Patel, Lounès Chikhi & Jordi Salmona

Perrier's sifaka (*Propithecus perrieri*). (Illustration: Stephen D. Nash)

Perrier's sifaka (*Propithecus perrieri*) is intermediate in size relative to other members of the genus *Propithecus* (Ranaivoraisoa *et al.* 2006; Lehman *et al.* 2005) and is characterized by an all-black pelage, naked black face and striking orange-red eyes (Mittermeier *et al.* 2010). At present, Perrier's sifaka occurs only in dry deciduous forests on limestone karst and semi-evergreen transitional forests on sandstone soils and has a diet of predominantly leaves and flowers (Lehman and Mayor 2004). Little is known about many details of its biology including its behaviour and past distribution. Its current distribution is the smallest of all *Propithecus* species and it has been recognized as Critically Endangered since 1996 (IUCN 2015; Mittermeier *et al.* 2010; Salmona *et al.* 2013). Its geographic range is restricted to the extreme northeast of Madagascar, some 50 km to the south of Antsiranana (Diego Suarez). It extends from the eastern edges of the Analamerana limestone massif, along the Indian Ocean coast to the sandstone forests of the Andrafiamena mountains as far west as the peaks of Ambohibe northeast of the rural commune of Marivorahona. It is bound in the north by the Irodo River and in the south by the Andrafiamena mountain range (Banks 2012; Zaonarivelo *et al.* 2007). Despite evidence of the species' presence in the Ankarana National Park in the 80s and 90s (Hawkins *et al.* 1990; Meyers 1996), three recent surveys in 2003, 2004 and 2012 (Banks *et al.* 2007; Rasoloharijaona *et al.* 2005; Salmona *et al.* 2013) failed to find Perrier's sifakas there. Furthermore, suggestions that its distribution might also extend south of the Andrafiamena mountains and into the Andavakoera forest (Schwitzer *et al.* 2006) could not be confirmed during two surveys to the area in 2006 and 2012 (Zaonarivelo *et al.* 2007; Salmona *et al.* 2013).

Earlier estimates of its total population size based on total remaining habitat within the species' range suggested that less than 1000 individuals persist in the wild. Estimates of the effective population size from field data (about 230 individuals; Banks *et al.* 2007) and from genetic data (Ne ~50–100; Salmona *et al.* submitted), further support that the population is small. However, doctoral thesis work by Banks between 2007 and 2012 provided revised estimates of population density that addressed differences in habitat quality in dry and semi-evergreen forest types. After modelling detection in this elusive species as a function of behaviour, observer, and habitat differences, the study reached a population size estimate of 2133 (95% CI: 1761–2584) individuals. Although a small number of forest patches within the range are not included in these estimates, the area sampled (252 km^2) comprises 85% of the total remaining habitat available estimated (296.6

km^2) by Banks *et al.* (2007), for this species. Population densities were found to be up to an order of magnitude higher in sandstone forests. Despite representing only a small portion of the habitat (about 12%), sandstone forests likely host nearly 40% of the remaining Perrier's sifaka population (Banks 2013). Ominously, a Landsat 7 imagery analysis of the region between 1994 and 2003 reveals that sandstone forests were the most susceptible to forest loss, with a total loss of more than 60% over this period.

Although its habitat is fragmented, Perrier's sifaka can cross open areas for distances of up to 600 m (Mayor and Lehman 1999). Other sifaka species are known to disperse over much larger distances of open habitat (Meyers and Wright 1993; Richard *et al.* 1993). This and evidence from studies of Perrier's sifaka population levels in eleven forest fragments (range: 1.1 and 124 km^2) and occupancy patterns in 45 fragments less than 1 km^2 in area indicate that the population is not strongly influenced by the fragmentation of forests and the matrix of open grassland habitats. However, with fewer than 2,200 known individuals left in the wild (Banks 2013), not all of which are reproductive, and a long generation time of about 18 years, the viability of the population is at a high risk of being compromised (generation time between 6 and 20 years, based on data from Verreaux's sifaka, *P. verreauxi*; Lawler *et al.* 2009; Morris *et al.* 2011). Genetic data are extremely limited. Mayor *et al.* (2002) identified relatively high levels of genetic diversity compared to other sifaka species and useful genetic markers for the genus *Propithecus*. Some of these markers together with markers from other sifaka species were used for the first population study conducted by Salmona *et al.* (submitted). The results of this study suggest that the three main forest fragments do not "behave" as differentiated populations. They also suggest that, in the best interests of safeguarding the species, measures to maintain connectivity between forest fragments should be implemented (Salmona *et al.* submitted). Even though Perrier's sifaka may have the ability to cross open grassland, most sifakas encountered on the ground were elusive and fled from humans (Salmona *et al.* submitted). Furthermore, studies of occupancy patterns show that Perrier's sifakas avoid forest patches in proximity to human settlements with ≥ 10 households where they are susceptible to attacks from dogs, particularly when attempting to cross matrix habitat (Banks 2013). The combined effects of deforestation, fragmentation and human activity could prevent them from routinely crossing open land, thereby decreasing gene flow and further fragmenting the remaining population (Salmona *et al.* submitted).

Decades ago, several authors reported Perrier's sifaka presence outside of its current distribution (Hawkins *et al.* 1990; Meyers 1996). Moreover, subfossils of *P.* cf. *diadema* (most probably *P. perrieri*) were reported in Ankarana cave (Jungers *et al.* 1995; Godfrey *et al.* 1996) and far north from its current distribution in Montagne des Français, Andavakoera cave (Dewar *et al.* 2013 and reference therein). Perrier's sifaka paleodistribution and population size may have been larger than today. In addition recently found genetic signatures of population decline suggest that the population underwent a major decline in the past 5,000 years (Salmona *et al.* in prep.) similar to the decline detected in the neighbouring golden crowned sifaka (Quéméré *et al.* 2012). Although it is not clear which events brought Perrier's sifaka to its current restricted distribution and small population size, it is likely due to the conjugated effects of climatic and human driven forest size fluctuations.

There is also emerging evidence that pressures on lemur populations in northern Madagascar are on the rise (Rakotodravony 2006, Reuter *et al.* 2014) and local guides indicate that the absence of regular surveys across the region since the Banks' studies ended in 2012 has substantially exacerbated the impact from threats such as hunting, fire, land conversion and habitat loss (J. R. Zaonarivelo pers. comm). Given the small total population size, persistence of pressures from local threats and the paucity of wildlife patrols in remote forests critical for Perrier's sifaka survival, a return to monitoring activities and an appraisal of its population levels are urgently needed. Moreover, a unified regional management plan is required to safeguard this species from extinction. The species' natural range and potential areas of migration/seasonal presence overlap with three protected areas all with different protected statuses, independently managed by Madagascar National Parks (Analamerana and Ankarana) and Fanamby (Andrafiamena). Stakeholders involved must operate at different levels of the decision making process (e.g. park services, ministries, universities, tour operators, local businesses, farmers, etc.) making the integration of all perspectives a real challenge. Reaching targets for conservation with this species given the diverse group of stakeholders involved requires a clearly defined institution, committed to leading a species conservation plan with incentives for action that are inclusive and take advantage of the strengths of the different participants.

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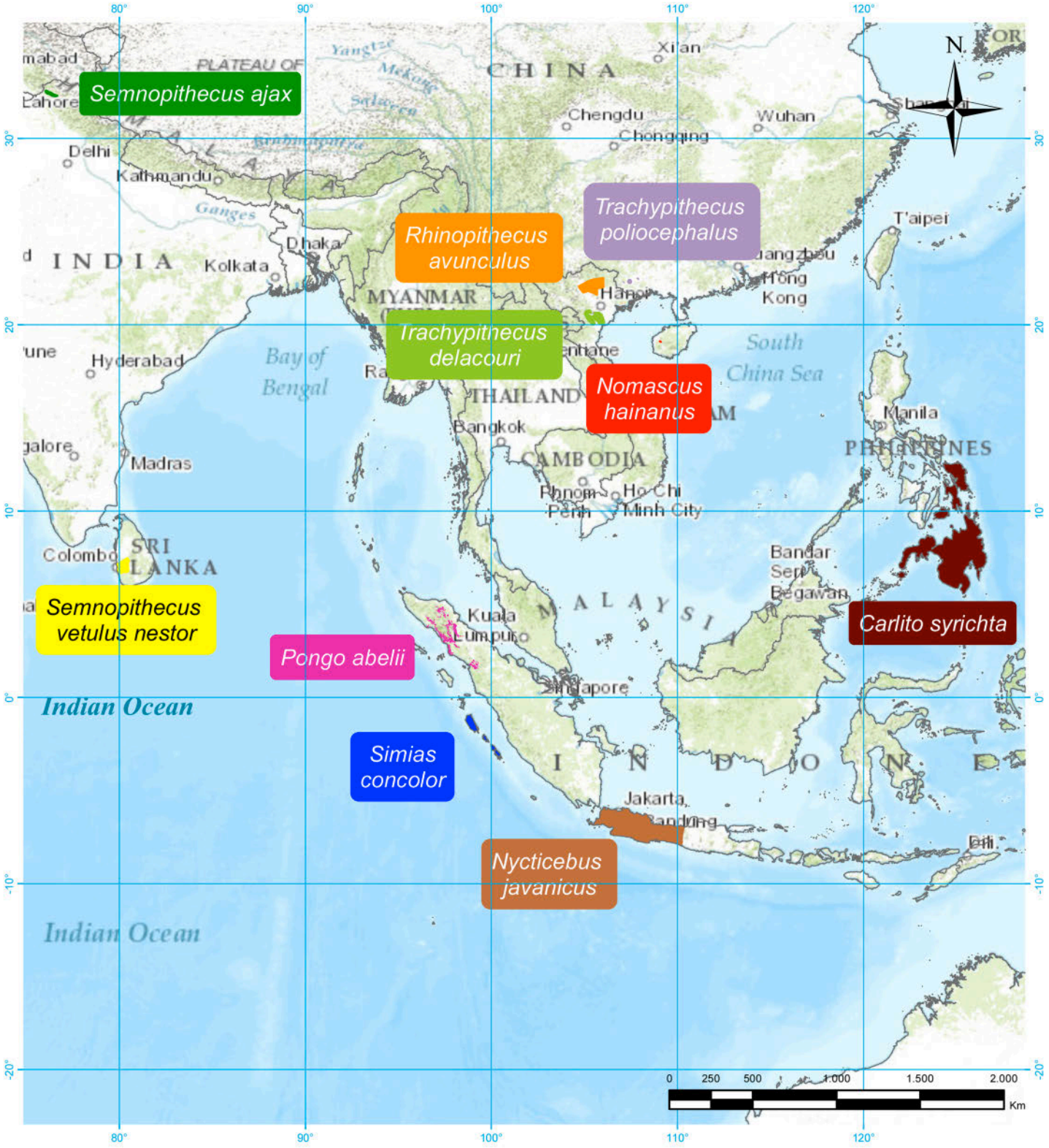
Asia



Asian Primates

- *Carlito syrichta*
- *Nomascus hainanus*
- *Nycticebus javanicus*
- *Pongo abelii*
- *Rhinopithecus avunculus*
- *Semnopithecus ajax*
- *Semnopithecus vetulus nestor*
- *Simias concolor*
- *Trachypithecus delacouri*
- *Trachypithecus poliocephalus*

IUCN/SSC PRIMATE Specialist Group

Philippine Tarsier
*Carlito syrichta** (Linnaeus, 1758)
Philippines
(2014)

Myron Shekelle, Sharon Gursky, Stefan Merker & Perry Ong

Philippine Tarsiers (*Carlito syrichta* subspecies) (Illustrations: Stephen D. Nash)

Modern genetic tools reveal that Philippine tarsiers, *Carlito syrichta*, are a group of three distinct genetic lineages: the Bohol-Samar-Leyte lineage, the Dinagat-Caraga lineage and the Mindanao-Zamboanga lineage (Brown *et al*. 2014). However, these distinct genetic lineages do not perfectly match the three possible subspecies, *T. syrichta syrichta* from Leyte and Samar, *T. s. carbonarius* from Mindanao, and *T. s. fraterculus* from Bohol, creating a very uncertain situation in which some populations might be threatened with extinction. Furthermore, we don't know which one, or ones, are threatened, and with deforestation having removed nearly all of the Philippine tarsiers' original habitat in many places, there is urgent need to act now and identify these unique primate taxa as a conservation priority.

The history of the Philippine tarsier's IUCN threat status has changed through the years, indicating a profound lack of knowledge. Originally listed as Endangered (EN), it was downgraded in 1996 to Lower Risk/Conservation Dependent (LR/CD). Subsequently it was listed as Data Deficient (DD), and is currently listed as Near Threatened (NT) (Shekelle & Arboleda 2008). What this reflects is that the metapopulation of all tarsiers found within the Philippines, scattered over 4 large islands (Samar, Leyte, Bohol, and Mindanao) and an unknown number of smaller islands (e.g., Dinagat, Basilan) might not be in imminent danger of going extinct across its entire range. However, this merely masks the true state of some populations, representing genetically distinct lineages, which could more likely be in trouble.

The three genetically distinct lineages do not correspond with the distribution of the three subspecies described, albeit recognized as such by only a few taxonomists. Brandon-Jones *et al*. (2004) inferred that the lack of acceptance is more due to the limited morphological evidence needed for taxonomic separation. As with many taxonomically cryptic nocturnal species, morphological variation is minimal. This is further complicated by museum specimens of Philippine tarsiers being greatly skewed toward just two sites on the Gulf of Davao on Mindanao Island. Thus, taxonomic clarity of the three lineages hinges on acquiring more morphological data as well as genetic data across its known range, quickly, before these distinct lineages disappear and the opportunity to reveal the real species status within the metapopulation is lost forever.

**Carlito syrichta* is used as it is the most recent published name, however some authors prefer using *Tarsius syrichta* when referring to Philippine tarsiers.

The Philippines have been described as the region where "megadiversity meets mega deforestation" (Panela 2014). On the list of The World's 10 Most Threatened Forest Hotspots, the Philippines rank 4th, ahead of Madagascar, a country that is infamous for its conservation crisis (CI 2011). Indeed, perhaps as little as 7% of the original forests remain in the Philippines. This is particularly distressing since whether tarsiers are obligate forest species or not remains unclear. It is yet another issue that must be resolved quickly.

Further compounding the threats to Philippine tarsier survival are the seeming increase in frequency and intensity of typhoons due to climate change. In 2013, supertyphoon Haiyan swept directly over critical tarsier habitats in Leyte and Samar. The direct effects of this supertyphoon and other expected typhoons, as well as other natural disasters of similar magnitude, on tarsier survival are still being studied, but are most likely devastating (Gursky unpublished data).

Finally, while tarsiers are used as an ecotourism mascot on the island of Bohol, the regulation of this practice is weak. Many tarsiers are on display as roadside attractions in conditions that can be heartbreaking, especially as these are nocturnal animals on display during the daytime. Given the difficulties of keeping tarsiers alive in captivity, it is assumed that mortality among these animals is high, and that replacements are collected from the wild when they die. Furthermore, there is a burgeoning illegal trade in tarsiers as pets, which unfortunately, is probably caused to some degree by the tarsiers' status as tourism mascots. While the use of tarsiers as tourism mascots is laudable and proves their ability to promote tourism and thereby conservation, the industry in the Philippines needs to improve how tarsiers are managed for tourism and conservation.

Therefore, the inclusion of the Philippine tarsier among the world's 25 most endangered primates, even though its current IUCN threat status is just Near Threatened, is due to the following: First, the NT status was accorded to the metapopulation of all Philippine tarsiers on all islands. With the identification of three distinct genetic lineages of tarsiers, the extent of occurrence of each genetic lineage is now considerably reduced and the population status of each lineage is unknown, thus the threat of extinction for some distinct genetic lineage is now a real possibility. Second, the Philippines are a land of mega deforestation and it remains unclear whether non-forest habitats, though reported to be potential alternative habitats, could sustain the three distinct genetic lineages of tarsiers in perpetuity. Third, climate change might lead to the increased frequency and intensity of typhoons, particularly in a highly vulnerable country like the Philippines, thus combined with the first two factors described above, this might further exacerbate the extinction risk of tarsiers. Finally, tarsiers possess great power to promote conservation through their role as tourism mascots, but unfortunately, owing to weak management, current practices could exacerbate the risk of extinction, and this needs correction. The reasons enumerated above were based on the best available information and should be further validated on the ground. Experts with direct experience of these tarsiers need to be mobilized immediately to determine if the three genetic lineages are indeed separate species, as well as the population level of each lineage; assess the suitability of non-forest habitats to sustain tarsier populations; study how the extinction risk of tarsiers is impacted by catastrophic disturbances; and examine ways by which tarsiers as a flagship species for conservation through ecotourism can be enhanced without causing them harm.

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Javan Slow Loris

Nycticebus javanicus É. Geoffroy Saint-Hilaire, 1812
Indonesia
(2008, 2012, 2014)

K. Anna I. Nekaris & Vincent Nijman

All Asian lorises are imperiled by the devastating loss of their habitat. An even greater immediate threat to Asian slow lorises, however, is their high demand in the rampant Asian pet and traditional medicine trades and their use as tourist photo props (Schulze and Groves 2004; Das *et al*. 2009; Nekaris *et al*. 2010; Osterberg and Nekaris 2015). Their wide availability as pets has led to new threats, including their body parts made available as fashionable key rings (Nijman *et al*. 2014) and a rising Internet trade fuelled by multiple viral Internet videos of pet lorises (Nekaris *et al*. 2013). Easy to catch due to their slow locomotion, numbers of slow lorises in animal markets far outstretch the ability of these slow-reproducing primates to recover their population numbers in the wild (Nekaris and Starr 2015). Indeed, this threat raised international concern, resulting in the transfer of all members of the genus *Nycticebus* to CITES Appendix I in 2007 (Nekaris and Nijman 2007). Eight species of slow loris are now recognized: *N. coucang* (greater), *N. pygmaeus* (pygmy), *N. bengalensis* (Bengal), *N. menagensis* (Philippine), *N. borneanus* (Bornean), *N. kayan* (Kayan), *N. bancanus* (Sody's), and *N. javanicus* (Javan) (Chen *et al.* 2007; Munds *et al*. 2013; Pozzi *et al.* 2014). All slow lorises suffer from trade throughout their range, but when combined with tremendous habitat loss, no other species has been harder hit than the Javan slow loris.

Javan slow loris (*Nycticebus javanicus*)
(Illustration: Stephen D. Nash)

Recognized by the IUCN as a species in 2006, and currently listed as Critically Endangered (IUCN 2014), the Javan slow loris is distinguished easily from its congeners in several respects. Both morphologically and genetically, it is most similar to, yet still distinct from, the largest slow loris, *N. bengalensis* of mainland Asia (Roos 2003; Groves and Maryanto 2008). Weighing about 1 kg, the most distinctive feature of the Javan slow loris is its facial mask, comprised of bold fork marks leading from the eyes and ears to the crown of the head, revealing a white diamond pattern on the forehead (Nekaris and Jaffe 2007). Despite being legally protected since 1973, with its creamy neck, bold dorsal stripe, and panda-like face, it is no wonder that Indonesian pet traders in the 1990s targeted Javan slow lorises above other endemic loris species. Since 2002, however, the numbers of Javan slow lorises in trade have decreased, with a stark rise in numbers of (Sumatran) greater slow lorises (*N. coucang*), a species whose threat status must

also be carefully monitored. Indeed, over three years of markets surveys on Java from 2012 to 2015, quadruple the number of greater than Javan slow lorises were counted, with traders claiming that Javan slow lorises could no longer be found (Nijman *et al.* in press). In November 2013 alone, nearly 300 greater slow lorises were confiscated in two raids. The smaller raid, yielding 76 individuals, was followed by the almost immediate death toll of 31 individuals. All of these animals were confiscated before ever making it to markets, showing the dramatic extent of this trade.

The Javan slow loris is found only on the Indonesian island of Java. Java has a long history of cultivation and deforestation that already started *c.*1000 AD, but took off in 1830 when the Dutch colonial government imposed the so-called 'cultuurstelsel'. To support this agro-economic system, farmers were forced to grow export crops on communal grounds, which were often forest (Whitten *et al*. 1996). By the end of the 19th century the natural forest was severely fragmented, and at the beginning of the last century the remaining forest, especially in West and Central Java, showed a fragmentation pattern very similar to that seen today. Over the last few decades, the decrease in forest area has been slow. At present, less than 10% of the original forest remains, most of it covering the higher slopes of the central mountains.

GIS models made available by Thorn *et al*. in 2009 suggested that historic forest loss and continued degradation mean that less than 20% of habitat suitable for Javan slow loris remains and that only 17% of the potential distribution of Javan slow loris is currently within the protected area network of Java. Voskamp *et al*. (2014) and Nekaris *et al.* (2014) investigated nine of these areas along with an additional six unprotected areas. Their results concurred with those obtained by three separate research groups, with animals occurring at 0.02 to 0.20 animals per km, when they could be found at all, meaning 5-50 km must be walked to see a single slow loris (Nekaris and Nijman 2008; Winarti 2008). Roads and human disturbance have been shown to correlate negatively with Javan slow loris abundance (Winarti 2008). During Voskamp *et al*.'s study, numbers of slow lorises were higher in agroforest that in some cases was extremely disturbed by humans. Nekaris *et al*. (2014) found that walking speed significantly influenced the number of slow lorises spotted.

Also urgently required are programmes to mitigate trade in all species of slow loris. A number of studies have found that slow lorises are not always a targeted group, but that they do have economic value throughout their range. Rather than seeking a slow loris, villagers moving through the forest simply pick up a loris when they happen to see it (Starr *et al*. 2008). Similarly, when forests are clear cut (for agriculture or cash crops), villagers pick through the felled trees and collect the slow lorises; with a defense mechanism to cling to branches rather than to flee, and with their nocturnal senses stunned by bright daylight, lorises are an easy target (Ratjacsek 1998). Nijman and Nekaris (2014) showed that traditional beliefs about slow lorises may hinder people from hunting them, particularly beliefs regarding their being venomous or poisonous.

Slow lorises are often targeted, with, in Java, specialized collectors searching the countryside. In addition, locals who come across a slow loris may collect it, and pass it on to middlemen. Most slow lorises collected in this manner end up in the so-called bird markets that are found in most major towns in Java. Once they arrive at a market, slow lorises face other threats. To avoid being bitten by slow lorises, which are one of few venomous mammals, traders habitually cut or pull out an animal's lower front teeth (Nekaris *et al*. 2013). Most of these slow lorises die due to general infection, dental abscess or pneumonia (Nekaris and Starr 2015). Those that do survive are no longer able to eat their preferred food (gum) (Wiens *et al.* 2006), or to engage in the important behavior of social grooming with the toothcomb, meaning that any confiscated animals are unlikely to survive if released to the wild.

Reintroduction itself is a threat to the Javan slow loris; the major trade hubs on Java and the neighboring island of Sumatra receive different species of slow loris from throughout the region. The similar appearance of slow lorises to the untrained eye results in release of other slow loris species into Java, with potential for disastrous effects from hybridization or displacement of native species by introduced ones (Nekaris and Starr 2015). The ability for slow lorises to persist in human habitat if left undisturbed means that well-meaning people may translocate animals to habitat that is unknown to the animals, causing potential death, disease transmission, and invasive species issues (Kumar *et al*. 2014).

Moore *et al*. (2014) assessed the success of reintroduction of Javan slow lorises, finding up to a 90% death rate. Illness, hypothermia and exhaustion were all implicated in the death of lorises. Sadly, reintroductions were started before anything was known about the behavior, ecology or wild distribution of slow lorises. No habitat assessment could be made since it was not even known in what type of habitat the species occurred. Subsequently it is reported that success is improving, but no published data are available. A related study of pygmy slow loris in Vietnam found that the season in which lorises are released as well as the age of the releases is vital for success (Kenyon *et al*. 2014).

In 2011, the first long-term study of Javan slow loris behavioural ecology was instigated by the Little Fireface Project in Garut District, West Java, Indonesia (Nekaris 2014; Rode-Margono *et al*. 2014). This multi-disciplinary project has obtained the first data about slow loris behavior in an agroforest matrix, including home range size, social organization, infant dispersal, and feeding ecology. Some notable discoveries have been that both sexes disperse from their natal range at about 18 months old, that dispersal distances are some 1–2 km from the natal range, that home range sizes are large relative to the size of the animal (5–10 ha), and that the diet of lorises comprises mainly gum, supplemented with nectar and insects. Several initiatives have been put into place to conserve slow lorises in the area and in other parts of Java. National workshops have been held for law enforcement officers and rescue center employees to feed essential data into a national slow loris action plan. At the local level, slow lorises are totally dependent on local people for their protection, feeding on human-planted tree species and residing on farmland. Thus a major conservation program, combining empowerment activities, conservation education and village events, has been launched, and it is hoped that it can be used as a model for other key slow loris sites in Indonesia (Nekaris and Starr 2015).

For a long time, slow lorises were thought to be common throughout Indonesia, and the presence of animals in trade was believed to be an indicator of their abundance. We are only beginning to unravel the complexity of their taxonomy and distribution, leading to an overall bleak picture. While Java has an impressive and comprehensive protected area network, encompassing over 120 terrestrial conservation areas covering some 5,000 km², enforcement of environmental laws and active protection of forest is lacking in most of these parks. Besides curbing the illegal trade, it is paramount that these conservation areas, and indeed all other remaining forest areas on the island, be effectively protected.

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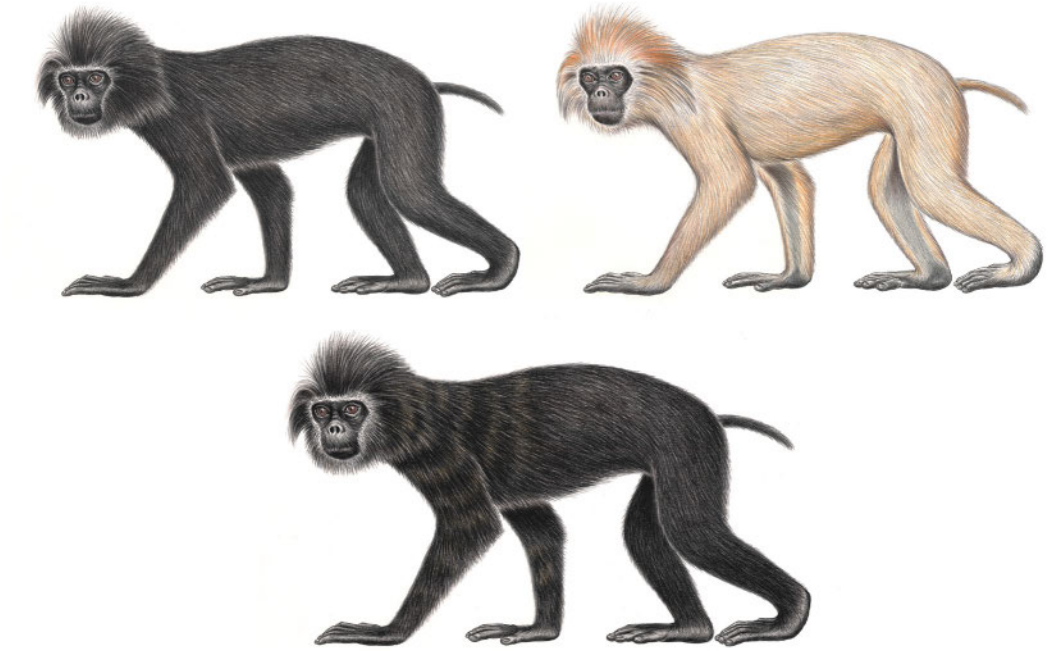
Pig-tailed Snub-nosed Langur
Simias concolor Miller, 1903
Indonesia
(2002, 2004, 2006, 2008, 2010, 2012, 2014)

Lisa M. Paciulli, Jatna Supriatna & Christian Roos



The two color forms of the Pig-tailed snub-nosed langur (*Simias concolor concolor*) above, and the Siberut subspecies (*Simias concolor siberu*), below (Illustrations: Stephen D. Nash)

The Pig-tailed snub-nosed langur (*Simias concolor*) is again serving as the flagship species for the six Mentawai Island primates. The other five species inhabiting the 7,000-km² archipelago west of Sumatra are Kloss's gibbon (*Hylobates klossii*), the Pagai surili (*Presybtis potenziani*), the Siberut surili (*P. siberu*), the Pagai macaque (*Macaca pagensis*), and the Siberut macaque (*M. siberu*). *Simias* is a monotypic genus with two subspecies: *S. c. concolor* / masepsep (Miller, 1903) that inhabits Sipora, North Pagai Island, and South Pagai Island; and *S. c. siberu* / simakobu (Chasen and Kloss, 1927), which is restricted to Siberut Island (Zinner *et al.* 2013).

Simias concolor is classified as Critically Endangered on the IUCN Red List (Whittaker and Mittermeier 2008), and is threatened mainly with hunting, commercial logging, and human encroachment (Whittaker 2006). The Pagai Island populations have to contend with forest conversion to oil palm plantations, forest clearings, product extractions by local people (Whittaker 2006), and opportunistic hunting (Paciulli 2004). Where hunting occurs on the Mentawai Islands, it has devastating effects on *Simias*, as it is the preferred game species (Mitchell and Tilson 1986; Fuentes 2002; Paciulli and Sabbi 2016). Entire groups can be eliminated in a single hunting excursion (Hadi *et al.* 2009a). On the Pagais, few men report actively hunting (Paciulli 2004), but on Siberut, 24% of the men still hunt, with 77% targeting pig-tailed snub-nosed langurs (Quinten *et al.* 2014). On Siberut, hunting reduces group size, and appears to significantly affect adult sex ratios and the number of immatures in groups (Erb *et al.* 2012a).

Simias numbers also decline significantly after timber removal. On the Pagai Islands, densities averaged 5.17 individuals/sq km (13.4/sq mi) in unlogged forests and 2.54/sq km (6.6/sq mi) in forests that had been logged approximately 20 years earlier (Paciulli 2004). It is estimated that on the Pagais, there are approximately 3,347 pig-tailed snub-nosed langurs, 1,049 Kloss's gibbons, 1,545 Pagai surilis, and 7,984 Pagai macaques (Paciulli and Viola 2009). All of the primate species seem to reach their highest known densities in the Peleonan Forest, site of the Siberut Conservation Project in northern Siberut (Waltert *et al.* 2008). In Peleonan

peat-swamp forests, *S. c. siberu* has densities as high as 65.5 individuals/sq km (Quinten *et al*. 2010).

The uncertainty of Indonesian government land-use means that land function and thus, protection level on the Mentawai Islands can change at any time with little notice, putting the species at further risk. There is only one large protected area for *Simias*: the 190,500-ha Siberut National Park (SNP), a UNESCO Biosphere Reserve that covers 47% of Siberut. Although SNP serves as the main reserve for ~51,000 primates (Quinten *et al*. 2015), hunting is significantly more prevalent there than elsewhere, with ~4,800 primates being removed per year (min. 6.4 % of the population) (Quinten *et al.* 2014). Drastic measures need to be taken to ensure that the Peleonan Forest on Siberut and areas on the Pagai Islands are truly protected.

Whittaker (2006) suggested the following conservation actions for *S. concolor*: 1) increased protection for Siberut National Park, which currently lacks enforcement; 2) formal protection of the Peleonan forest in North Siberut, which is home to unusually high primate populations and is easily accessible; 3) protection of areas in the Pagai Islands by cooperating with a logging corporation that has practiced sustainable logging there since 1971; 4) conservation education, especially regarding hunting; and 5) the development of alternative economic models for the local people to reduce the likelihood of selling their land to logging companies.

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Delacour's Langur

Trachypithecus delacouri (Osgood, 1932)
Vietnam
(2000, 2002, 2004, 2006, 2008, 2010, 2012, 2014)

Tilo Nadler

Delacour's langur (*Trachypithecus delacouri*)
(Illustration: Stephen D. Nash)

Delacour's langur is endemic to Vietnam, occurring in a very restricted area in the north of the country that comprises about 5,000 km² between 20°–20.40°N and 105°–106°E. The distribution is closely related to the limestone mountain ranges in the Provinces Ninh Binh, Thanh Hoa, Hoa Binh and Ha Nam.

During the decades following the discovery of Delacour's langur in 1930 there was only scant information on its existence and distribution. In 1987, the first sightings of live animals were reported from Cuc Phuong National Park (Ratajszczak *et al*. 1990). Intensive surveys by the Frankfurt Zoological Society in the decade before 2000 confirmed 18 completely isolated populations with a total of 280 to 320 individuals. Five localities were found where local people reported that this species had been extirpated.

Trachypithecus delacouri is listed as Critically Endangered (Nadler *et al*. 2008). The most important factor in the decline in numbers is poaching, which is not primarily for meat, but for bones, organs and tissues that are used in the preparation of traditional medicines. The recorded numbers of animals hunted over 10 years (1990–1999) totalled 320, an annual loss of more than 30 individuals, but the real number is undoubtedly higher (Nadler 2004; Nadler *et al*. 2003).

Monitoring of selected populations carried out after 2000 showed a continuously severe decline of some subpopulations. Two important subpopulations, in Cuc Phuong National Park and Pu Luong Nature Reserve, decreased in numbers by 20% during five years (2000–2004) (Luong Van Hao and Le Trong Dat 2008). The population in Ngoc Son Nature Reserve was extirpated (Le Trong Dat *et al*. 2008). Nine subpopulations were extirpated in a single decade (2000–2010).

Between 2012 and 2015, 20 surveys were carried out for a status assessment in all nine areas with remaining subpopulations. The surveys were organized with major support from Ocean Park Conservation Foundation, Hong Kong. The results show again a further decline compared to the last decade. The total number is estimated at 234-275 individuals (Nadler 2015). The majority of the individuals occurs in Van Long Nature Reserve, and combined with the population in the adjacent planned extension area the number totals

164–191 individuals. The remaining 70–84 individuals occur in the additional seven areas; one with about 30 individuals and the remaining six each with 4–10 individuals. None of these populations are likely viable in the long term, even if hunting could be eliminated. Only four of these areas are protected, but in most protected areas poaching is also common and it is to be expected that populations in unprotected areas will disappear in a short time. During the last 20 years the number of isolated subpopulations shrunk from 18 to 9 and the area of occupancy from about 400 km² to about 250 km² (Nadler 2015).

Together with the planned extension area, Van Long Nature Reserve harbours the largest remaining population of Delacour's langurs. The animals are well protected due to close cooperation between the provincial forest protection authorities, and a local guard unit paid and trained since the establishment of the Nature Reserve in 2001 and the following 10 years by a project of Frankfurt Zoological Society. The protection project continues through management of the Endangered Primate Rescue Center (EPRC) with support from different donors (EAZA, Thin Green Line). Since the establishment of the nature reserve, the population of Delacour's langurs has grown by about 100% (Ebenau 2011; Nadler 2010).

Efforts to save this species are one focus of the Vietnam Primate Conservation Programme of the EPRC at Cuc Phuong National Park. The EPRC was established in 1993 primarily to safeguard the future of this and other highly endangered Vietnamese primate species. The EPRC is the only facility which keeps this species. The centre started a breeding programme with five confiscated animals, and 21 individuals have been born since 1996. The first reintroduction of three captive bred Delacour's langurs was carried out in 2011 and continued in 2012 with the release of two individuals. This pilot project was the first reintroduction of leaf-eating langurs, following the IUCN guidelines for nonhuman primate reintroduction (Baker 2002). The animals were equipped with GPS-radio collars and tracked for nearly one year. The reintroduced animals should strengthen the smaller subpopulation in the larger part of Van Long Nature Reserve to support the exchange of individuals of the fragmented area of the nature reserve (Nadler 2012). The protection work in Van Long Nature Reserve has a high acceptance in the surrounding communes based on a very close contact between the communes, the Management Board of the reserve and the EPRC (Elser and Nguyen Hong Chung 2013).

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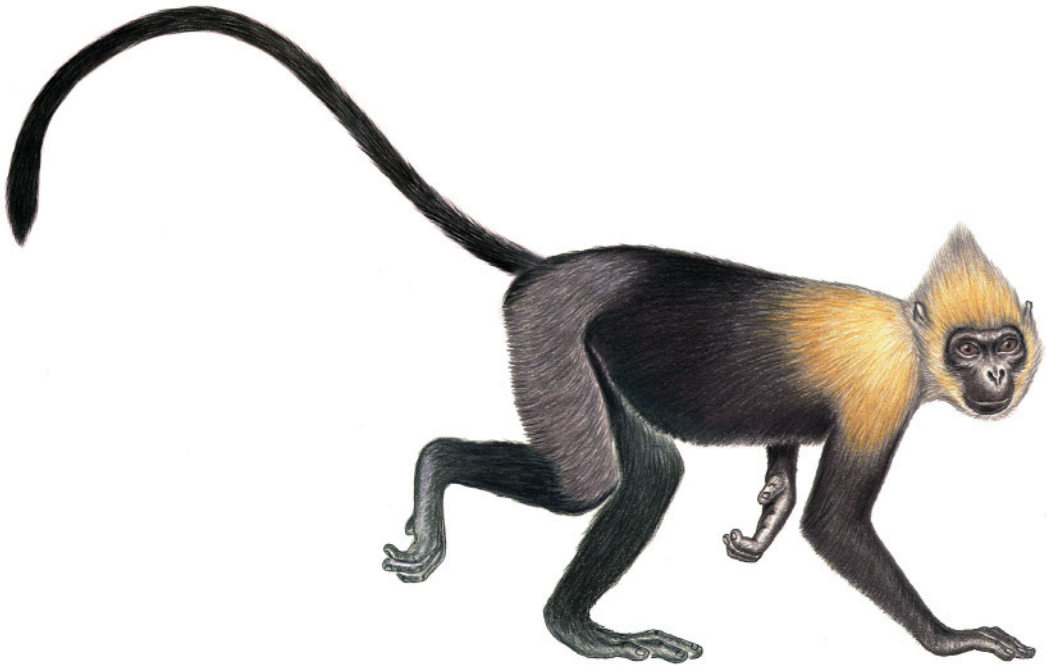
Golden-headed or Cat Ba Langur
Trachypithecus poliocephalus (Trouessart, 1911)
Vietnam
(2000, 2002, 2004, 2006, 2008, 2010, 2012, 2014)

Neahga Leonard, Richard J. Passaro, Daniela Schrudde, Roswitha Stenke, Phan Duy Thuc & Martina Raffel



Golden-headed or Cat Ba langur (*Trachypithecus poliocephalus*) (Illustration: Stephen D. Nash)

The Cat Ba langur (previously known as the golden-headed langur), *Trachypithecus poliocephalus*, is probably the most endangered of the Asian colobines, and is assessed as Critically Endangered (Bleisch *et al.* 2008). This species occurs only on Cat Ba Island, an island in the Gulf of Tonkin off the northeastern Vietnamese shore (Stenke and Chu Xuan Canh 2004). The Cat Ba Archipelago is adjacent to the world-famous Ha Long Bay, a spectacular karst formation that was invaded by the sea following the last major glaciation. The favoured habitat of the Cat Ba langur is tropical moist forest on limestone karst hills, a habitat preference it shares with the other five taxa of the *T. francoisi* group.

While there are no systematic and reliable data available on the historic density of the langur population on Cat Ba Island, reports by indigenous people suggest the entire island of Cat Ba (140 km^2) and some smaller offshore islands were previously densely populated by langurs. Hunting has been identified as the sole cause for the dramatic and rapid population decline from an estimated 2,400–2,700 in the 1960s to approximately 50 individuals by 2000 (Nadler and Long 2000). The langurs were poached mainly for trade in traditional medicines and for sport. Since the implementation of strict protection measures in 2000, the langur population on Cat Ba Island has stabilized (Nadler *et al.* 2003) and since 2003 has been on the increase, with current numbers estimated as being between 64–70 individuals (N. Leonard, Cat Ba Langur Conservation Program Project Manager, pers. obs.).

Although the growth of the population is encouraging, the overall status of the species remains critical and the total population worryingly small. Habitat fragmentation and hunting has divided the remaining population into several isolated sub-populations, some of which consist of all-female, non-reproducing social units. The total reproductive output of this species over the years has been low due to the small population and the long inter-birth cycle, but records indicate that the birth rate is increasing and 2014 and the first half of 2015 saw a substantial jump in birth rates with 16 infants being born in 18 months (N. Leonard pers. comm.).

In 2012, after many years of planning and preparation,

one group consisting of two females was successfully translocated from a small off-shore islet where they had become stranded to the relative safety of the strictly protected core zone of Cat Ba National Park. Here they quickly assimilated into existing groups containing males, thus allowing them the opportunity to reproduce for the first time ever. It is hoped that continued protection efforts and additional population management interventions such as these will enhance the rebound of this species.

The Cat Ba Archipelago and adjacent Ha Long Bay are nationally and internationally recognized for their importance to biodiversity conservation. Cat Ba National Park was established in 1986. It presently covers more than half of the main island. Ha Long Bay was established as a World Heritage site in 1994 and the combined archipelago includes some 1,500–2,000 large and small islands, cliffs and rocks. In 2004 the Cat Ba Archipelago was designated a UNESCO Man and the Biosphere Reserve. Despite the conservation designations and laws to protect the region, nature and wildlife protection on Cat Ba Island is deficient. Environmental awareness and commitment among the local communities is slowly increasing and hunting/trapping of all animals is illegal on Cat Ba, but efforts to effectively conserve the langurs and their habitat continue to face major obstacles due to ever increasing tourism development paired with a steadily increasing human population and severe deficiencies in law enforcement (Stenke 2005; N. Leonard pers. obs.). As is common elsewhere in the region, poaching by the local people is driven by livelihood issues, brought about by low incomes and a lack of employment opportunities. Immense local and regional demand for wildlife and animal parts for food and dubious traditional medicines provide a market for poached animals and plants. It appears that langur hunting completely stopped years ago, but hunters continue to poach other animals and plants within langur areas, placing the langur habitat in jeopardy. Strict enforcement of the established protections is necessary for the survival of all species on Cat Ba that are targeted by the Asian wildlife trade.

A conservation program for the Cat Ba langur was initiated on Cat Ba Island in November 2000 by Allwetterzoo Münster and the Zoological Society for the Conservation of Species and Populations (ZGAP), Germany. The aim of the Cat Ba Langur Conservation Program is to provide for the protection of the langurs and their habitat, conduct research that will help inform future population management decisions, and to help contribute to the conservation of the overall biodiversity of the Cat Ba Archipelago in collaboration with Vietnamese authorities.

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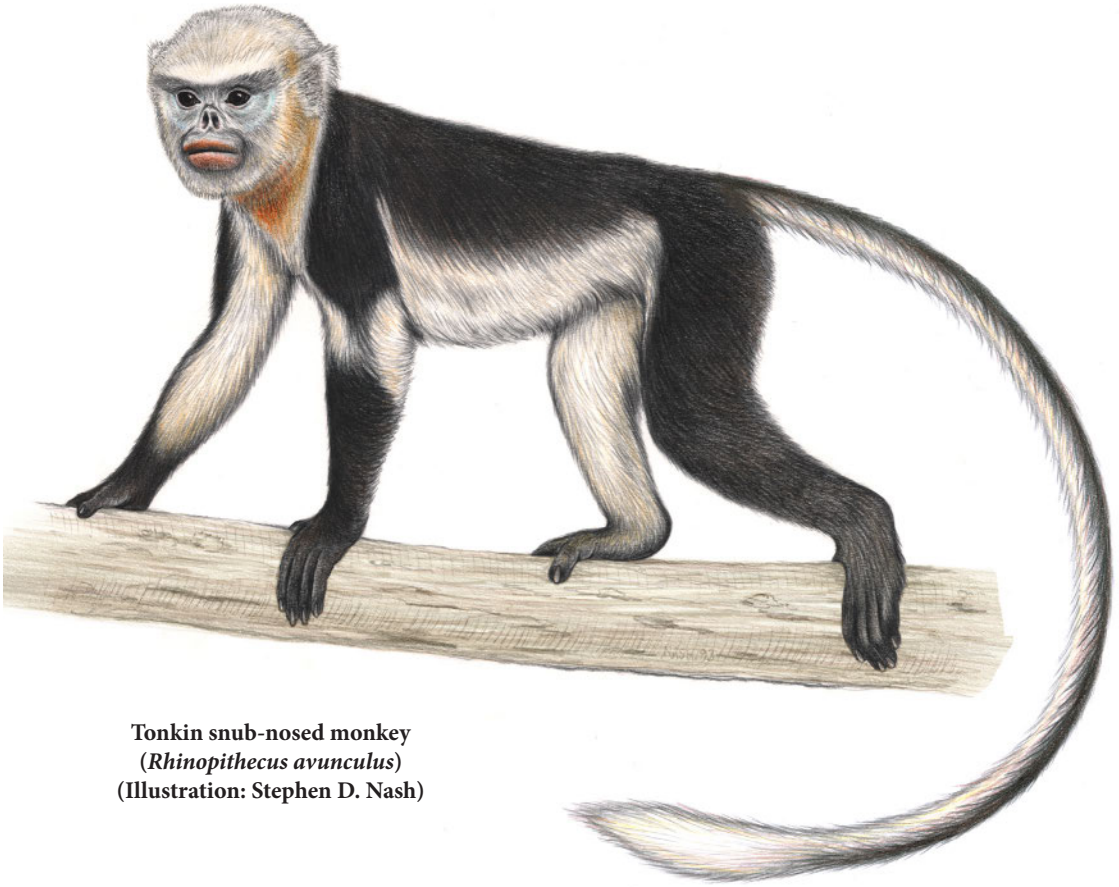
Tonkin Snub-nosed Monkey
Rhinopithecus avunculus Dollman, 1912
Vietnam
(2000, 2002, 2004, 2006, 2008, 2010, 2012, 2014)

Le Khac Quyet, Dong Thanh Hai & Tilo Nadler



Tonkin snub-nosed monkey (*Rhinopithecus avunculus*) (Illustration: Stephen D. Nash)

The Tonkin snub-nosed monkey, *Rhinopithecus avunculus*, is one of five unusual, large Asian colobine monkeys of the genus *Rhinopithecus*, all of which have a characteristic turned-up nose. Three species are endemic to China and the newly discovered Burmese snub-nosed monkey, *R. strykeri*, is found in Myanmar and China. *Rhinopithecus avunculus* is found only in northeastern Vietnam. It was discovered in 1911, and collected on perhaps no more than two more occasions over the course of the next 50 to 60 years. Consequently, this species was presumed to be extinct by a number of primatologists until it was rediscovered in 1989. Historically the species occurs only east of the Red River between about 21°09'-23°N. Due to widespread deforestation and intensive hunting in recent decades, its distribution has become severely restricted (Nadler *et al.* 2003). The total population of the Tonkin snub-nosed monkey is currently believed to be less than 250 individuals.

Rhinopithecus avunculus is Critically Endangered (Le Xuan Canh *et al.* 2008). Recent evidence suggests there are just five known locations where Tonkin snub-nosed monkeys occur, and they are completely isolated. In 1992, a population was found in Na Hang-Chiem Hoa region of Tuyen Quang Province. As a result of the discovery, Na Hang Nature Reserve was established in 1994. The nature reserve comprises two separate areas; the Ban Bung and Tat Ke sectors. A study in 1993 estimated a population of between 95 and 130 individuals in each sector respectively (Boonratana and Le Xuan Canh 1994), which was probably an overestimation (Thach Mai Hoang 2011). The most recent field surveys in 2010 found and estimated only 5–10 individuals in the Tat Ke sector, and 13–16 individuals in Ban Bung sector (Thach Mai Hoang 2011). Hunting is still the main threat to the monkeys in the Na Hang Nature Reserve. During surveys in 2010, local hunters, hunter shelters

and gunshots were recorded commonly in both Tat Ke and Ban Bung Sectors. Conservation activities carried out by several organizations have been unsuccessful, and it has resulted in a reduction of this population (Thach Mai Hoang 2011).

A population of about 70 individuals was estimated for Cham Chu Nature Reserve in 2001, also in Tuyen Quang Province. Based on interviews of local people during a survey that was reported in 1992, the population was believed to have dropped to only 20–40 individuals (Long and Le Khac Quyet 2001). A survey in 2006 provided no sightings and no reliable evidence of the survival of the population. Local reports indicate, however, a small group of 8–12 individuals still in the area (Dong Thanh Hai *et al*. 2006). The current threats to the populations of the monkeys are hunting and habitat destruction. Conservation efforts should target reducing human activities inside the reserve.

A population of about 60 Tonkin snub-nosed monkeys was discovered in early 2002 and a census in April 2015 confirmed 125–130 individuals in the Tonkin Snub-nosed Monkey Species/Habitat Conservation Area at Khau Ca, Ha Giang Province (Le Khac Quyet personal observation). This is the only population that is not immediately threatened. Here, population and habitat monitoring, conservation education, public awareness and community participatory activities are being linked to increased protection efforts under the supervision of the University of Colorado Boulder, Fauna and Flora International (FFI) and Denver Zoo.

In 2007, a new population of about 20 Tonkin snub-nosed monkeys was discovered in a small forest patch in Tung Vai Commune of Quan Ba District close to the border with China (Le Khac Quyet and Covert 2010), and was confirmed by a census in 2014 (Nguyen Van Truong 2014). This is the second population of Tonkin snub-nosed monkey discovered in Ha Giang Province. The newly discovered population at Tung Vai appears to be threatened by hunting and habitat loss due to timber exploitation, shifting cultivation and the collection of non-timber forest products for commercial purposes. The immediate conservation measures are likely to be training and establishing patrol groups, raising awareness, assessing the range of the monkeys, and assessing the impact of cardamom production on their habitat.

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Chamba Sacred Langur

Semnopithecus ajax (Pocock, 1928)

India and Pakistan

(2014)

Martina Anandam, Vishal Ahuja & Sanjay Molur

Chamba Sacred langur (*Semnopithecus ajax*)
(Illustration: Stephen D. Nash)

The least known of all langurs, *Semnopithecus ajax*, inhabits the steep, rugged mountains of western Himalaya. The species was described in 1928 by the famed British naturalist Reginald Innes Pocock. In his first account of the langur from Chamba published in the Journal of the Bombay Natural History Society, he described it as 'a handsome langur' with shaggy outer mane and stark dark forearms and a 'dusky brownish-grey upper side' (Pocock 1928). Adult males have a greyish-brown mane, unique to the species. All individuals have a bushy cowl and a generally big built stature like Ajax, the Greek warrior of Homer's Iliad (Brandon-Jones 2004).

Brandon-Jones (2004) extends their distribution to Kishtwar near Maru or Petgam, in Wardwan Valley (33°40' N and 75° 44' E) and in Chinab near Siri or Sereri Village (33° 19'N and 76° 03' E). Although Lydekker (1877) considered the Kishtwar and Chinab species to be *Semnopitheus schistaceus*, Brandon-Jones (2004) argued that they are "presumably" *S. ajax*. Brandon-Jones classified Nepal langurs as *Semnopithecus entellus ajax* owing to their similarity in coat colour and body size to Pocock's *ajax* (Pocock 1928, 1931; Brandon-Jones 2004). Photographs from Vaishno Devi in Jammu and Kashmir (Bhairo Baba's Friends Now 2011) indicate a clear upper 'shaggy' mane, dark forearms and the bushy cowl, but these need to be corroborated and substantiated with better, conclusive evidence. One group of langurs observed in Kangra sport orangish-brown fur with the females appearing more slender, similar to langurs of lower elevations, while males do have the characteristic mane of *S.ajax* (Martina Anandam and Vishal Ahuja, pers. obs). Minhas *et al*. (2012, 2013) and Mir *et al*. (2015) report *S. ajax* from Machiara National Park, in Azad Jammu and Kashmir, Pakistan (34°31'N and 73°37'E) and Dachigam National Park (34°05'–34°12'N and 74°54'–75°09'E), Jammu and Kashmir, India. Although Groves (2001) indicates a wide range for *S. ajax*, from Dehra Dun in Uttarakhand to Pakistani Kashmir in the northern end of the western Himalaya, Groves and Molur (2008) restricted the range of *S. ajax* to Chamba due to variations in other populations and possible hybridization or the probable distribution of *S. schistaceus*. *Semnopithecus ajax* occurs at elevations of 1200 m to 4000 m (Anandam *et al*. 2013b).

According to M. Anandam and V. Ahuja (pers. obs., unpublished) diet varies with season but the most commonly preferred species are *Rhododendron arboreum*, *Aesculus indicus*, *Quercus leucotrichophora*, *Pyrus pashia*, *Berberis asiatica*, *Rubus* spp., *Urtica* spp., *Abies pindrow* and crops such as *Zea mays* (maize), *Pyrus malus* (apple) and *Raphanus* sp. (radish). The langurs were observed to feed on mud and lick rocks, a behaviour not uncommon in primates. Langurs are persistent crop raiders, and cultigens make up a substantial portion of the diet. Seventy-six villages/sites of 244 surveyed in 2012–2013 reported crop raiding in Chamba.

A multi-male, multi-female social organization is typical of Himalayan langurs. *S. ajax* typically lives in groups of 25-40. The difficult terrain in Chamba and the shy nature of these langurs make it difficult to obtain group counts and carry out population censuses. Little is known, therefore, of their home range and habitat preference. Bachelor bands of adult and sub-adult males are rare

but do occur. Aggression is infrequent except during mating when adult males pursue each other in trees, but the behaviour is not as acute as in lowland langurs. Grooming is the most common social activity in adults as play is in juveniles and sub-adults. Allo-mothering or 'aunting', where infants are handled by females other than the mother, is also common. The langurs in Chamba have only two types of vocalization; the deep bark mostly by adult males and occasionally by females, and the squeals of infants and juveniles. The unanimous barks are used by males during group movement and occasionally in agonistic situations. Minhas *et al.* (2010a), however, report that in Machiara resident grey langur males give "morning whoops", which, despite the name, continue throughout the day in addition to barks. Such discrepancies warrant further taxonomic research on the Himalayan langurs.

Unregulated and uninformed tourism is a growing threat in Chamba. In Khajjiar-Kalatop Wildlife Sanctuary (3069 ha), a protected area within their range, tourist resorts, hotels and restaurants are burgeoning to feed the growing numbers of tourists from around the country. There is no proper waste disposal, leaving enormous amounts of non-biodegradable waste around the area. Roads are constructed to connect the formerly inaccessible areas to support trekkers and uninformed wildlife 'enthusiasts'. No environmental impact assessment has been carried out to study the effect of these activities on the wildlife. Logging and lopping are other concerns around Chamba. In the 2012–2013 survey (Anandam *et al*. 2013b), 76 sites reported conflict with langurs due to crop raiding. About 25% of respondents from the 244 sites surveyed expressed negative attitudes towards crop-raiding mammals such as black bears, porcupines, macaques and langurs. Crop raiding therefore is a threat and an issue in need of immediate attention.

The Chamba Sacred Langur is classified as Endangered based on its restricted distribution in Chamba as the populations reported from elsewhere need to be genetically resolved. The species is currently restricted to less than 5,000 km^2 in range and with a restricted area of occupancy of less than 500 km^2, occurring in several severely fragmented locations. The species is threatened by continuing decline in habitat quality from anthropogenic effects such as developmental activities, logging and fragmentation. It is also affected by the increasing human-langur conflicts in and around the only known protected area where it occurs and in other places within the valley. Current estimates also indicate the number of mature individuals to be less than 250, although this needs to be established with proper counts.

The Himalayan Langur Project set up conservation studies in Chamba in 2012, and has since been working to establish a comprehensive, research-based conservation programme for the Chamba Sacred Langur. The project is working with communities in Chamba to devise sustainable crop-protection measures and establishing a community conservation platform for future collaboration and conservation action. The aim is to unite farming communities and other concerned stakeholders into building a cooperative system to address issues concerned with human-wildlife interactions and others of environmental and conservation concern. The project also runs education and sensitization programs and has launched a conservation magazine, '*Achamba!*'. More information on the project and recent updates can be found at www.zooreach.org.

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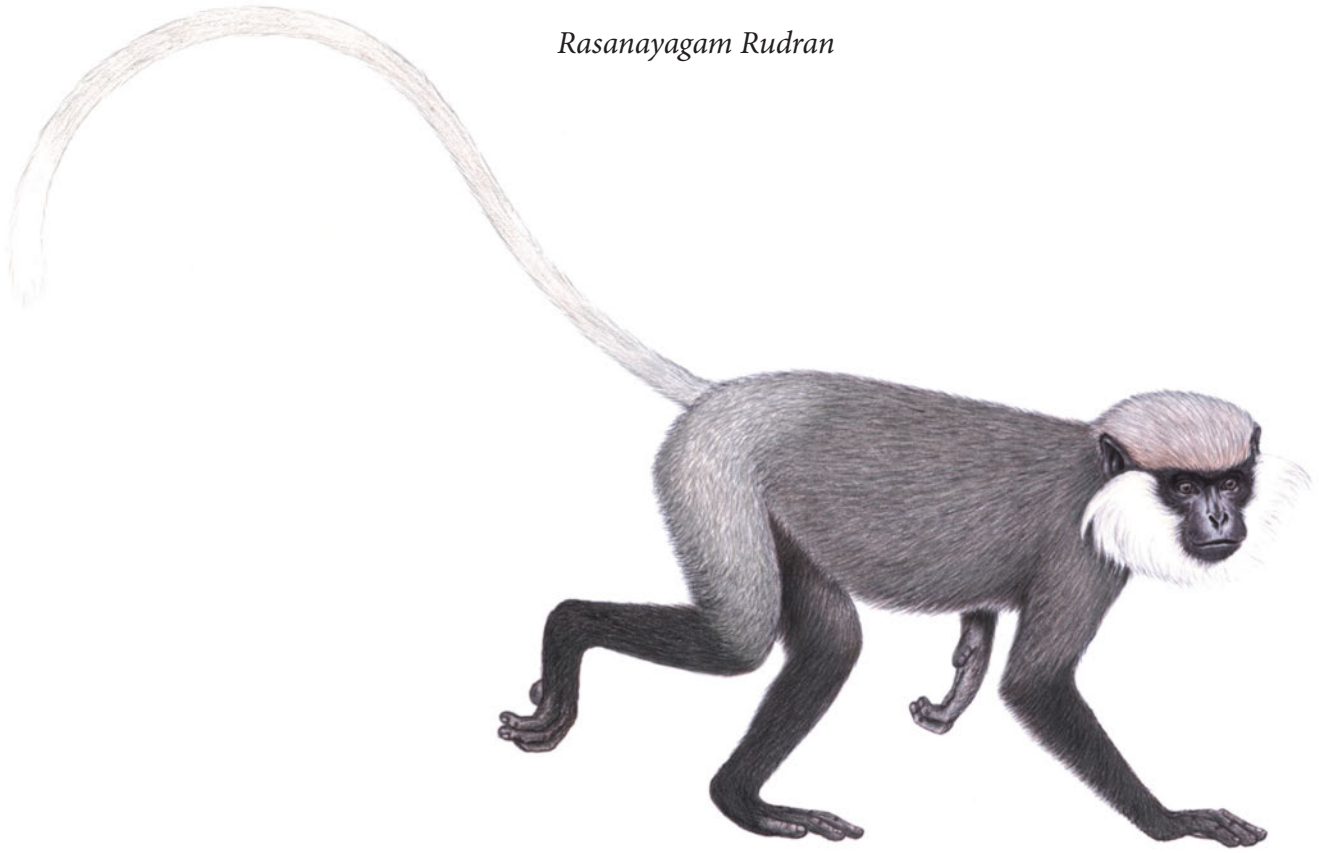
Western Purple-faced Langur

Semnopithecus vetulus nestor Bennett, 1833
Sri Lanka
(2004, 2006, 2008, 2010, 2012, 2014)

Rasanayagam Rudran



Western purple-faced langur (*Semnopithecus vetulus nestor*)
(Illustration: Stephen D. Nash)

The range of the western purple-faced langur (*Semnopithecus vetulus nestor*) is around Colombo, capital of Sri Lanka, in the most densely populated region of the country. Urbanization, therefore, poses a serious threat to the survival of this endemic and endangered monkey (Molur *et al*. 2003; Dittus *et al*. 2008; Rudran *et al*. 2009; Mittermeier *et al*. 2012). Urbanization has been so extensive that it has not been possible to obtain a reliable count of *S. v. nestor*'s population. The real predicament of this monkey, however, is in the size and extent of what remains of its natural habitat. In 2007, a 1,500-km survey conducted through one-third of its historical range (Hill 1934; Phillips 1935; Hill and Burn 1941) showed that nearly 81% consisted of deforested and human-dominated landscapes (Rudran 2007).

While reducing the preferred habitat of *S. v. nestor*, deforestation has also severely depleted the folivorous diet of this highly arboreal monkey. *Semnopithecus v. nestor* now subsists mainly on fruits from domestic gardens within the "urban jungle" (Dela 2007; Rudran 2007). The nutritional consequences of feeding on a low diversity diet of cultivated fruits are unclear. However, they are likely to be detrimental over the long term; *S. v. nestor* is adapted to obtaining its nutrients and energy from leaves with the help of a highly specialized stomach containing symbiotic bacteria (Bauchop and Martucci 1968).

Depletion of *S. v. nestor*'s preferred habitat and diet are not the only problems deforestation has created for this monkey's survival. Deforestation has resulted in extensive habitat fragmentation, which forces it to travel on the ground, for which it is ill adapted. This makes young individuals vulnerable to capture as pets. While on the ground *S. v. nestor* also runs the risk of being killed by village dogs or speeding vehicles. An additional source of mortality is death by electrocution when the monkeys are forced to travel along power lines due to the lack of arboreal pathways (Parker *et al*. 2008).

Another detrimental effect of habitat fragmentation was also evident during the 2007 survey (Rudran 2007). *Semnopithecus v. nestor* was seen or recorded as present only in 43% of the sites surveyed in the eastern half of its historical range (N = 23 sites), and 78% of the survey sites in the western half (N = 27 sites). Areas surrounding these sites were often devoid of monkeys, which suggested the occurrence of local extinctions. In some parts of its range *S. v. nestor* is occasionally shot and killed while feeding in gardens (Dela 2004). Deforestation results in a host of human-induced fatalities, which reduce group sizes and disrupt their social organization.

Although *S. v. nestor* faces a perilous future, there is hope that it can be conserved. One reason for hope is that most people living within this monkey's range follow the Buddhist doctrine of compassion toward all living things. Promoting this doctrine and Buddha's own reverence of the forest therefore presents opportunities to deter deforestation in a country steeped in cultural traditions. Another reason for optimism stems from a government decision, taken about five years ago, to increase Sri Lanka's forest cover from 27% to 36% using native plants as part of the country's economic development goals (Yatawara 2011).

Even before the government took the above decision, my team and I had launched a research project to help reforest degraded habitats and establish safe havens for *S. v. nestor*. The research commenced in June 2009 in the largest forest patch that it now inhabits (about 21 km^2). This forest became our study site due to its size, and also because it surrounded two reservoirs (Kalatuwawa and Labugama) that supply water to 1.2 million inhabitants of Sri Lanka's capital. Because of its importance to people, this forest was a secure safe haven for maintaining a viable population of *S .v. nestor* over the long term. Our field investigations focused on discovering the plants that were important for *S. v. nestor* so that they could be used in a reforestation initiative to increase the size of this monkey's habitat. This research ended in December 2010 and the resulting publication (Rudran *et al*. 2013) was submitted to the Forest Department of Sri Lanka with a request for permission to initiate a reforestation programme.

The Forest Department's permission is pending, but delays are inevitable when working with Sri Lanka's government institutions. Therefore, we sought and found a private land belonging to a Buddhist temple near the study site to begin our reforestation effort. Arrangements were made for the temple's young monks to collect seeds and saplings of selected species from the adjoining forest and maintain them in a nursery established by the project. Storage bins were also installed to collect and recycle excess food donated to the monks by devout Buddhists, so that compost could be made to fertilize the seedlings. Despite this effort it was clear from the outset that reforestation and research were not enough to ensure the survival of *S. v. nestor.* What was required was a broad-based programme that also included public education to promote nature conservation, and community development activities to improve the quality of life of impoverished rural people who were having a significant negative impact on natural habitats.

Since the project's inception, its public education programme has conducted conservation oriented classroom lectures and nature walks for primary and secondary school children living close to the research site. During the past two years these activities were conducted well beyond the research site as well. Similarly, community development activities that were initially conducted near the research site have been carried out elsewhere in the historical range of *S. v. nestor*'s distribution. For instance, a Health and Eye Care Clinic was held in February 2014 for elders living in Kosgama, a village several kilometers from the research site. This clinic distributed medicines for common old-age ailments such as arthritis, diabetes and hypertension to 48 seniors, and reading glasses to 79 of them. It also identified cataracts in nine elders who were assisted by the project to undergo corrective surgery at a nearby hospital. All assistance given to the elderly was free of charge, and this made people realize that the project was interested in their welfare as well as that of the endangered monkey. This had a positive impact especially on the adult children of the elders who had little interest in the project's conservation message. Living in extended families they were preoccupied with anxieties of caring for themselves, their children and their parents, but when the project helped to alleviate these anxieties they became more receptive to the project's conservation initiatives.

Besides helping adults indirectly through their children and parents, the project also helped them directly through other initiatives. It empowered young women by conducting workshops and training them to make patchwork cloth bags. These bags were sold

in the community and in nearby towns for reasonably attractive prices and provided extra income. A home gardening project was launched to help mothers housebound with child-rearing encumbrances to grow vegetables, to improve family nutrition and sell excess produce at nearby markets.

While the project strengthened its ties with local communities another threat in the form of human-monkey conflicts began to cause serious concern. This problem has continued to intensify (Nahallage *et al*. 2008), and currently affects all eight subspecies of Sri Lankan monkeys. Nearly 27% of the complaints (N = 371) received between 2007 and 2013 by the Department of Wildlife Conservation were about *S. v. nestor*. Most of the complaints about these monkeys came from residents living around the capital city who tend to be well educated and professionally qualified. Hence several newspaper articles were published to highlight the plight of *S. v. nestor*, and to remind the mainly Buddhist public about its cultural heritage of showing compassion to all living things.

Appeals via newspaper articles and all other on-going activities to help conserve *S. v. nestor* will continue. These activities will be replicated when another site is identified as suitable for the long-term survival of *S. v. nestor.* While working to conserve *S. v. nestor*, it was hard to ignore the fact that the other three subspecies related to it, three macaque subspecies and two loris subspecies, all of which are endangered and endemic to Sri Lanka, were also in dire straits either because of deforestation or because of conflict with humans, or both. The current project was expanded to address the conservation of these animals as well. The present focus of this expanded initiative is a site in the central highlands of Sri Lanka, which is home to the montane subspecies of the purple-faced langur, the toque macaque, and the red slender loris. Reforestation, public education and community development activities are currently underway at this site. Meanwhile vehicle surveys are being conducted in various parts of Sri Lanka to evaluate the intensity of human-monkey conflict, human attitudes towards this problem, and to identify areas outside the country's protected area system that local communities could manage under the supervision of wildlife authorities to derive benefits through the sustainable use of forest products, and through nature tourism and educational activities.

Active participation of local communities in managing and deriving sustainable benefits from natural habitats is a new concept in Sri Lanka. It was presented and discussed during two workshops where it received favorable responses from government authorities and non-governmental organizations. Accordingly, this idea has been incorporated into a Conservation Action Plan for Sri Lanka's non-human primates that is currently being developed following IUCN guidelines. When this Action Plan is completed it will be submitted to the Sri Lankan government and the IUCN for approval.

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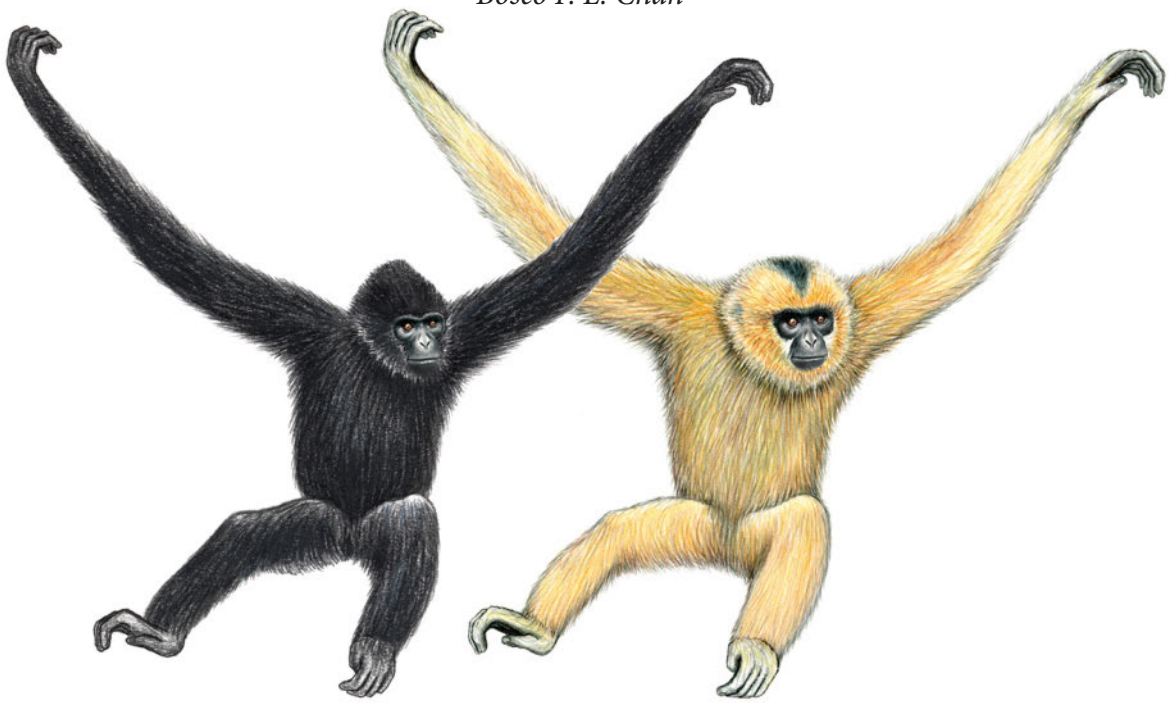
Hainan Gibbon

Nomascus hainanus (Thomas, 1892)
China (Island of Hainan)
(2000, 2002, 2004, 2006, 2014)

Bosco P. L. Chan



Hainan gibbon (*Nomascus hainanus*). Male (left) and female (right) (Illustrations: Stephen D. Nash)

The taxonomy of the northern group of crested black gibbons, genus *Nomascus*, has been resolved by molecular, pelage and vocalization studies (Geissmann *et al.* 2000; La Q. Trung and Trinh D. Hoang 2004; Mootnick 2006; Roos and Nadler 2005; Roos *et al.* 2007). The Hainan gibbon, *Nomascus hainanus*, is restricted to the Chinese island of Hainan. All of the nine species and subspecies of crested gibbons are endangered, and the Hainan gibbon is the most endangered of them all (Geissmann 2003; Geissmann and Chan 2004; Wu *et al.* 2004). Adult males are entirely black. Adult female Hainan gibbons vary from a buffish to a beige brown and have a black cap (Geissmann *et al.* 2000; Mootnick 2006). Mootnick and Fan (2011) provided detailed descriptions of the species comparing it to the other crested gibbons. The closest relatives of the Hainan gibbon are the eastern black crested gibbon (*Nomascus nasutus*) and western black crested gibbon (*N. concolor*) (Fan *et al.* 2006, 2010).

In the 1950s there were estimates of >2,000 Hainan gibbons on the island of Hainan in 866,000 ha of forests across 12 counties (Liu *et al.* 1984). By 1989, the Hainan gibbon population was reduced to a single relict population of 21 gibbons in four groups, restricted to a patch of primary montane rainforest on Mt. Futouling covering 16 km^2 in Bawangling National Nature Reserve (Liu *et al.* 1989). In 1998, the population was said to be 17 (Kadoorie Farm & Botanic Garden 2001). A gibbon survey in October 2003 found two groups, and two lone males, comprising a total of 13 individuals (Fellowes and Chan 2004; Geissmann and Chan 2004; Chan *et al.* 2005; Zhou *et al.* 2005); another survey in 2001–2002 estimated 12–19 individuals in four groups (Wu *et al.* 2004). Determined conservation efforts including regular monitoring of the population were launched in 2003 (Fellowes *et al.* 2008; Mootnick *et al.* 2012), and the population has been slowly recovering, with the latest population census confirming at least 25 gibbons living in four breeding groups. The exact number of individuals cannot be ascertained due to a number of dispersed subadults leaving their natal groups in recent years.

Since 2003, when the first Hainan Gibbon Action Plan was launched (Chan *et al.* 2005), several teams have

continued to work roughly in line with the action plan. One team consists of the Hong Kong-based Kadoorie Farm and Botanic Garden, the Bawangling National Nature Reserve, and the Hainan Wildlife Conservation Bureau of the Hainan Provincial Forestry Department. Their work include surveying for remnant Hainan gibbons in other potential sites throughout the island, regular monitoring of the gibbons, restoring the degraded lowland forest, and community conservation work in villages adjoining the gibbon home range. The conservation action plan was revised in an international conservation-planning workshop co-organised by the Zoological Society of London in 2014, in an attempt to inject new directions to enhance existing conservation work.

With less than 30 Hainan gibbons confirmed, surviving in just one small forest block, the Hainan gibbon is considered by some to be the most critically endangered primate in the world. Following over a decade of determined conservation effort, support from the government and the surrounding ethnic group community, conservation of the gibbons and their habitat has improved significantly. With the steady increase in the gibbon population, there is an urgent need to secure and expand suitable lowland forest for the survival of the remaining gibbons and their habitats, which will require continued effort and cooperation among all parties.

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Sumatran Orangutan

Pongo abelii Lesson, 1827

Indonesia (Sumatra)

(2000, 2002, 2004, 2006, 2008, 2014)

Ian Singleton, Serge A. Wich, Matthew G. Nowak and Graham Usher

The Sumatran (*Pongo abelii*) and Bornean (*P. pygmaeus* Linnaeus, 1760) orangutan species comprise the genus *Pongo* (Groves 2001) and the Sumatran orangutan is considered a single taxonomic unit. The Sumatran orangutan is facing the more immediate threat of extinction and is listed as Critically Endangered on the IUCN Red List of Threatened Species (IUCN, 2015).

Sumatran orangutan (*Pongo abelii*)
(Illustration: Stephen D. Nash)

The species is endemic to Sumatra, Indonesia, where wild populations persist mainly in remaining lowland forests in the provinces of Aceh and North Sumatra (78% and 22% of the species' range, respectively; Wich *et al*. 2011). Two reintroduced populations are being established, one further south in Jambi Province, and one in the far north of Aceh.

The most recent published estimates suggest that only around 6,600 wild orangutans remain in just nine fragmented habitat units from the central regions of Aceh, south to the Batang Toru River in North Sumatra (Wich *et al.* 2008; noting the loss of the East Singkil population since 2008). More recent surveys, as yet unpublished, indicate the total population to be larger than 6,600, largely since orangutans were found to be at higher densities at higher elevations over their range than previously suspected, but the overall trend in orangutan numbers and habitat area remains decidedly downwards (Wich *et al*. in press).

Genetic studies looking at mitochondrial DNA (Nater *et al*. 2011) indicate the southernmost populations of around 550 individuals near the Batang Toru river, are genetically quite distinct from more northern populations, indicating very limited dispersal of females. These studies also suggest the Batang Toru orangutans could be the last remnants of an ancestral, central and southern Sumatra population, from which all orangutans further north, and those in Borneo, could have descended. These findings have led to debate that the genetic differences may warrant distinct species status, in which case the Batang Toru orangutans would immediately become the world's most endangered great ape species.

Orangutans are extremely vulnerable to extinction. Their exceptionally slow reproductive rate, low densities and very large home ranges mean viable populations

require vast areas of contiguous rainforest. They are also mainly restricted to lowland forests. Sumatran females give birth to one infant every eight or nine years (Wich *et al*. 2009) and the loss of as little as 1% of females each year can place a population on an irreversible trajectory to extinction (Marshall *et al*. 2009).

The largest Sumatran orangutan populations are found in the Leuser Ecosystem, the only place in the world where viable wild populations of the Sumatran orangutan, Sumatran tiger, Sumatran rhinoceros and Sumatran elephant co-exist. Approximately 78% of the Sumatran orangutan distribution lies within the Leuser Ecosystem (Wich *et al*. 2008). Recognising its unique biodiversity the Leuser Ecosystem is listed as one of the "World's Most Irreplaceable Protected Areas" (Le Saout *et al*. 2013). It is a 26,000 km^2 protected area straddling the border of Aceh with North Sumatra, established by Presidential Decree in 1998 and since 2008 is also now a National Strategic Area for its Environmental Function, the protection of which is required under National Laws.

Within the Leuser Ecosystem lie the smaller 1,025 km^2 Singkil Swamps Wildlife Reserve and the 7,972 km^2 Gunung Leuser National Park, part of the UNESCO Sumatran Rainforest World Heritage Site. The smaller National Park is mostly high mountains, and, as Sumatran orangutans are rarely found above 1,500 m above sea level most orangutans are outside of the National Park, but within the Leuser Ecosystem.

The primary threat to Sumatran orangutans is habitat destruction and fragmentation. Even within the Leuser Ecosystem forests are still being cleared at a large scale, primarily for conversion to oil palm plantations but also for mining, settlement and agricultural encroachment.

Precise rates of forest loss are difficult to determine, but primary lowland forests in Sumatra have been devastated over the last 30 years. Wich *et al.* (2011) report that 49.3% of all Sumatra's forests were lost between 1985 and 2007. In Aceh and North Sumatra the figures were 22.7% and 43.4%, respectively. If only the most important orangutan habitat is examined – i.e. forests below 1,000 m – for the 1985–2007 period at least 28% of Aceh's forests were lost and 49% of North Sumatra's. When only the most species-rich forests (below 500 m) are considered, forest loss between 1985 and 2007 was 36% for Aceh and 61% for North Sumatra. For the carbon rich peat swamp forests that harbour the very highest densities of Sumatran orangutans, the loss was 33% for Aceh and 78% for North Sumatra.

Numerous roads have also been cleared in orangutan habitat in recent years, often in remote areas and over unsuitable terrain. These roads are increasingly fragmenting remaining orangutan populations and opening new access for encroachment, settlement and illegal wildlife poaching. Gaveau *et al*. (2009) concluded deforestation rates could easily increase from 294 to 385 km^2 per year if all new roads scheduled for construction at that time would be built. Forest cover present in 2006 would shrink by >25% (9226 km^2) and orangutan habitat would be reduced by 16% (1137 km^2), resulting in the conservative loss of an estimated 1384 Sumatran orangutans, or 25% of the global population at that time, directly linked to road construction alone. These losses would largely arise from extensive losses (56%) of forest cover in lowland forests (<500 m.a.s.l.) where the highest densities occur.

Sumatran orangutan infants continue to be illegally kept and traded as pets. Illegal pets tend to be a byproduct of forest conversion, which results in the death of almost all wild orangutans in an area. They are often deliberately killed (e.g., as pests or to obtain their infants) or die more gradually due to starvation and malnutrition. Numbers of infants confiscated as illegal pets fluctuate between 15 and 35 per year (SOCP data). Orangutans are also still regularly killed as pests for raiding fruit crops at the forest edge. Recent years have seen two prosecutions in North Sumatra for orangutan trading. Penalties remain light but this reflects a slight improvement in the efforts of law enforcement and conservation agencies.

Conservationists are extremely concerned that each of the major threats to orangutans is increasing and highly likely to get considerably worse over the coming years as a result of an illegal new spatial land use plan developed by the government of Aceh Province. The new plan completely ignores the existence of the Leuser Ecosystem despite numerous laws and regulations requiring its protection. Due to its illegality, the plan has not been approved by Indonesia's National Government, but the provincial government insists on implementing its development plan and opening up large tracts of the Leuser Ecosystem regardless of National Law. This is evidenced by the fact that whilst the Leuser Ecosystem is not recognized in the spatial plan, the Governor of Aceh has issued a regulation explaining how to obtain concessions for plantations and other activities within its boundaries. Furthermore, numerous currently illegal roads already cross many parts of the Leuser Ecosystem and if not quickly cancelled the new plan will effectively legalise many of them, potentially sounding the death

knell for Sumatra's iconic megafauna species, including Sumatran orangutans. The issue has been brought to global attention, and efforts are still underway to replace the existing spatial plan with a new, legal plan, that complies with all existing National Laws and regulations. Efforts include offers of technical expertise and funding from several quarters, but the current stand-off between the Aceh Provincial and National Government remains a barrier to resolving this crisis.

Key conservation interventions rely heavily on rapidly replacing the currently illegal Aceh spatial plan and upholding national laws and regulations, without which the future of Sumatra's orangutans, tigers, rhinos, and elephants looks particularly bleak. A dramatic and rapid improvement in enforcement of wildlife and forest laws and far greater consideration for environmental issues in spatial planning decisions are urgently needed. Implementing patrols, increasing prosecutions, halting illegal and legal forest conversion, mining and road construction, promoting forest restoration, and addressing human-orangutan conflicts are also seen as prerequisites for the species' survival.

There is some room for optimism, however. Legal challenges have resulted in the cancellation of one palm-oil concession permit within the Leuser Ecosystem and a IDR 366 billion (about USD 27 million) fine for the company (the largest ever for burning forests), in addition to prison terms for its Director and Operations Manager. Other companies are also facing similar charges. Nevertheless, as long as the illegal Aceh Provincial spatial plan continues to be implemented even these positive developments will not halt the downward slide towards the extinction of the species in the coming decades.

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Neotropics

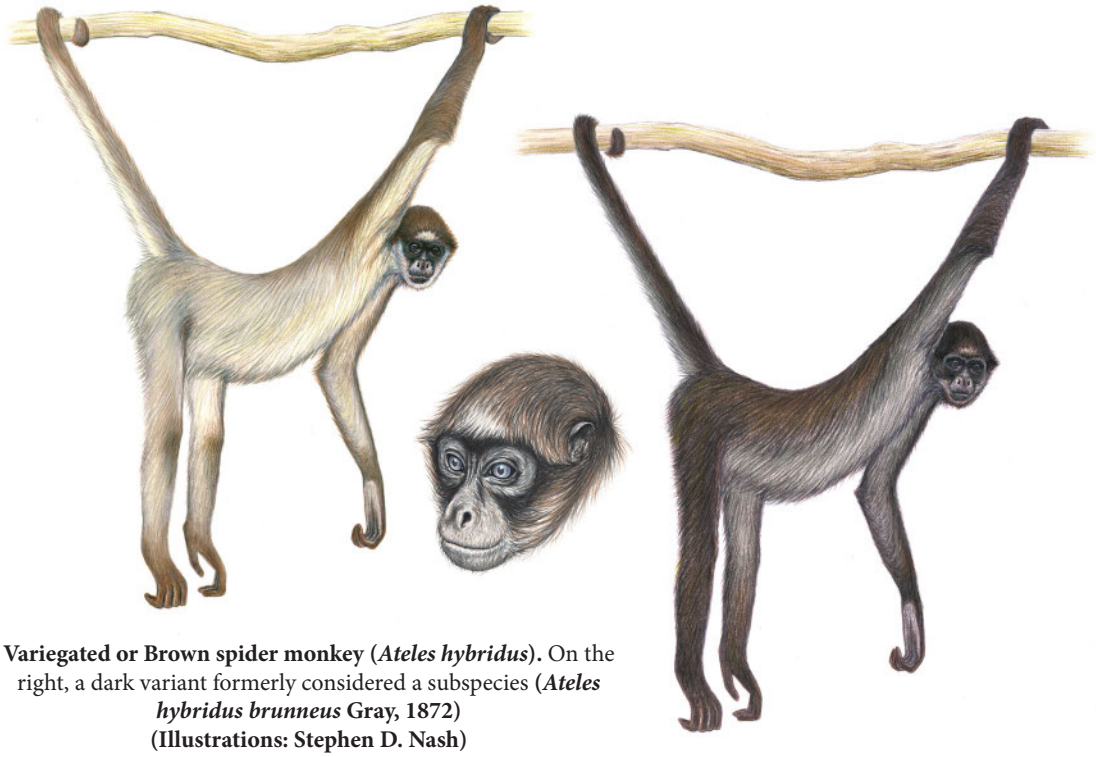
Neotropical Primates

- *Alouatta guariba guariba*
- *Ateles fusciceps fusciceps*
- *Ateles hybridus*
- *Callicebus oenanthe*
- *Cebus kaapori*

IUCN/SSC PRIMATE Specialist Group

Brown Spider Monkey
Ateles hybridus I. Geoffroy Saint Hilaire, 1829
Colombia, Venezuela
(2004, 2006, 2008, 2010, 2012, 2014)

Andrés Link, Pablo Stevenson & Ana Gabriela de Luna



Variegated or Brown spider monkey (*Ateles hybridus*). On the right, a dark variant formerly considered a subspecies **(*Ateles hybridus brunneus* Gray, 1872)**
(Illustrations: Stephen D. Nash)

Brown spider monkeys (*Ateles hybridus*) are restricted to Colombia and Venezuela, occurring in the middle Magdalena River valley, extending into northeastern Colombia and western Venezuela (Defler 2003) with an isolated population in northeastern Venezuela. Brown spider monkeys are considered to be Critically Endangered due to habitat loss and fragmentation, hunting and the pet trade (Morales-Jiménez *et al.* 2008a,b; Link *et al.* 2013). Recent phylogenetic studies provide evidence that populations of brown spider monkeys along the eastern and western banks of the Magdalena river have had gene flow between them and do not form monophyletic clades, thus questioning the validity of the subspecific classification (Link *et al.* 2015). Nonetheless, further studies should explore if populations nearer to the headwaters might be hybrids of more distantly related populations in the northeastern and northwestern areas of their distribution.

Spider monkeys are large primates with extremely slow reproductive cycles. Females have their first offspring at the age of 7 to 8 years, and give birth to a single offspring every 3-4 years. In addition, they use large areas of undisturbed forests (when available) where they feed preferentially on a large variety of ripe fleshy fruits (Di Fiore *et al.* 2008). Thus, spider monkeys are especially vulnerable to hunting and are one of the first vertebrates to go locally extinct under anthropogenic pressures to their habitat or populations. Historically, *A. hybridus* has suffered from habitat destruction and fragmentation and is absent from heavily intervened areas (Link *et al.* 2008). In Colombia, less than 18% of its historical distribution remains in forested habitats and <1% of their current habitats are protected. Most of its range has been transformed into extensive cattle-ranching and more recently into large oil palm plantations (Cordero-Rodríguez and Biord 2001; Link *et al.* 2013). The forests of the Magdalena River valley in Colombia, the Catatumbo area in Colombia, and the lowland forests in the state of Zulia and the piedmont of the Perijá Mountains in Venezuela are heavily destroyed because of expansionist cattle-ranching activities. For example, in the Perijá Mountains only 30% of the forest

is relatively well preserved and protected (Portillo-Quintero and Velásquez 2006). Both in Colombia and Venezuela, brown spider monkeys are heavily hunted (Lizarralde 2002; Link *et al*. unpublished data).

The remaining populations of brown spider monkeys are surrounded by increasing human populations, and are already facing high levels of threat. Legal and illegal mining as well as habitat destruction for large scale monoculture (e.g., oil palm) pose an imminent threat for the remaining populations. Given that most of its wild populations are outside protected areas, there is an urgent need to protect some of the priority areas for brown spider monkeys such as Serranía San Lucas and Serranía del Perijá. These areas may hold some of the largest populations of wild brown spider monkeys (Link *et al*. 2013).

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Ecuadorian Brown-headed Spider Monkey

Ateles fusciceps fusciceps Gray, 1866

Ecuador, Colombia

(2012, 2014)

Diego G. Tirira, Alba Lucia Morales-Jiménez & Paola Moscoso-R.

Ecuadorian brown-headed spider monkey(*Ateles fusciceps fusciceps*)
(Illustrations: Stephen D. Nash)

Ateles fusciceps lives in Central and South America, from southeast Panama to Ecuador, west of the Andes along the Chocó Eco-region. It is a diurnal species that inhabits mostly evergreen humid tropical and subtropical forests. The subspecies *Ateles fusciceps fusciceps* inhabits the Pacific Coast of Ecuador and possibly southern Colombia, in an elevational range of 100 to 1,800 m above sea level. The subspecies is distributed in Ecuador from the northwestern Andes Mountain Range, in Esmeraldas Province to the northwest of Pichincha and Santo Domingo provinces, extending to the western borders of Imbabura and Carchi Provinces (Tirira *et al.* 2011).

Ateles fusciceps fusciceps mainly inhabits large continuous forest patches in primary or secondary forest and prefers the highest levels of the canopy. Its presence in certain localities may be due to suitable habitat conditions such as continuous canopy cover and high abundance of large and tall trees. The species lives in groups of up to 35 individuals. The size of subgroups varies from 1 to 10 individuals (Gavilánez-Endara 2006; Estévez-Noboa 2009; Cueva and Pozo 2010; Moscoso 2010). Its diet comprises mainly ripe fruits; this is supplemented with leaves, flowers, seeds, aerial roots, invertebrates, fungi, decaying wood, mud and termitaria.

Ateles f. fusciceps is classified as Critically Endangered on the IUCN Red list (Cuarón *et al.* 2008) and in the *Red Book of Mammals of Ecuador* (Tirira *et al.* 2011), because of its restricted distribution and the small size of its natural populations. Extensive and ongoing deforestation and hunting are the main threats for the species in Ecuador; destruction of the humid tropical and subtropical rainforest in western Ecuador has surpassed 80% of its original area (MAE, 2012). Tirira (2004) presented information on the historical and current distribution of the subspecies, reporting several localities where it is locally extinct, including the type locality (Hacienda Chinipamba, west of Ibarra, Intag Valley, Imbabura Province), the whole central coast of Ecuador, and the surroundings of the Cayapas, San Miguel, Ónzole and Santiago rivers, in the Esmeraldas Province. Nevertheless, in some localities such as Playa de Oro (in Esmeraldas Province) where conditions have improved (e.g., hunting has ceased), populations of this subspecies are recovering (Moscoso 2010). Currently, *A. f. fusciceps* is concentrated in the interior part of Esmeraldas

Province, and adjacent regions of Imbabura and Carchi Provinces, as well as a small portion of northwest Pichincha Province. Some recent observations have been made in Los Bancos in Pichincha Province (Moscoso *et al*. 2011; S. Shanee unpublished data); and in Flavio Alfaro in the northwest of Manabí Province (Cervera and Griffith submitted). Nevertheless, it is uncertain if these populations are connected with other subpopulation of the subspecies.

Priority areas for the conservation of *A. f. fusciceps* are the Cotacachi-Cayapas Ecological Reserve and its area of influence (mainly along the western border of unprotected forests), Corredor Awacachi, the Awa Ethnic Forest Reserve, north of the Mira River and close to the Colombian border, and the buffer and surrounding areas of these reserves (Moscoso *et al*. 2011). The buffer area of the Cotacachi Cayapas Ecological Reserve, especially de Tesoro Escondido Coop within the Canandé area, is possibly the area that is harbouring the greatest subpopulations of *A. f. fusciceps* in Ecuador (Moscoso 2010; Peck *et al*. 2011). Population density estimates in the buffer areas of the Cotacachi-Cayapas Ecological Reserve and the Awa Ethnic Reserve are 0.2–8.5 individuals/km² (Gavilánez-Endara 2006; Cueva 2008; Estévez-Noboa 2009; Moscoso 2010). The presence of *Ateles fusciceps fusciceps* in Colombia is uncertain, but there is an unconfirmed record of *A. fusciceps* in Barbacoas, Nariño. Thus far, no subpopulation bigger than 50 individuals has been found. Preliminary genetic analyses from samples from the south of Colombia and the north of Ecuador show two different monophyletic clades (Morales-Jimenez, unpublished data).

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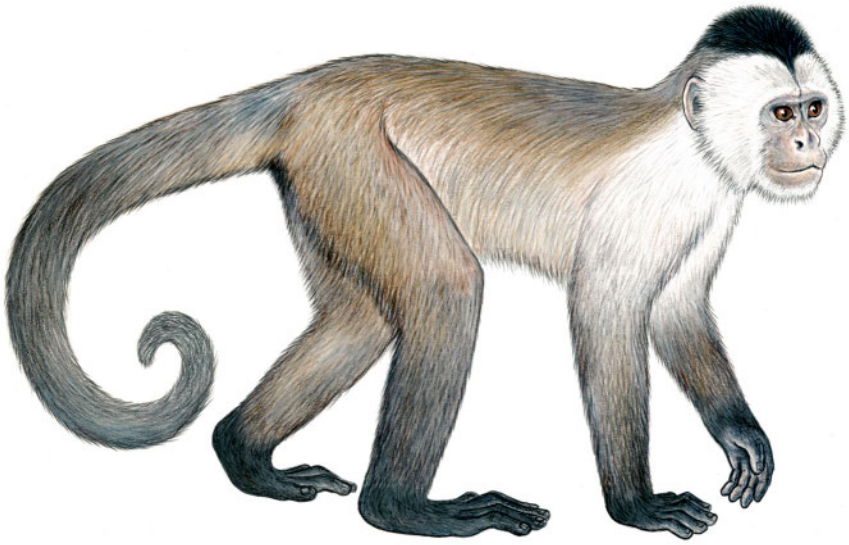
Ka'apor Capuchin
Cebus kaapori Queiroz, 1992
Brazil
(2012, 2014)

Marcos de Souza Fialho, Leandro Jerusalinsky, Maurício Talebi & Fabiano R. Melo



Ka'apor capuchin (*Cebus kaapori*)
(Illustration: Stephen D. Nash)

The Ka'apor capuchin (*Cebus kaapori*), first described just over 20 years ago, is found in the eastern edge of the Brazilian Amazon, in the north-eastern part of the state of Pará and north-western part of the state of Maranhão (Queiroz 1992). Its range extends from the east of the lower Rio Tocantins to the Rio Grajaú where it enters the Zona dos Cocais (Queiroz 1992; Ferrari and Queiroz 1994; Ferrari and Souza 1994; Silva and Cerqueira 1998; Carvalho *et al.* 1999; Cunha *et al.* 2007). It has been observed only in tall lowland terra firma forest, generally below 300 m above sea level, and has not been recorded in seasonally inundated forest or secondary forest (Rylands and Mittermeier 2013). The birth season is from June to July. Besides surveys and abundance studies, recent research provided additional ecological information about the species (Oliveira *et al.* 2014). This capuchin is generally seen in small groups of up to ten individuals, sometimes accompanying the also Critically Endangered bearded sakis (*Chiropotes satanas*) (Ferrari and Lopes 1996; Carvalho *et al.* 1999).

The known range of *C. kaapori* is suspected to include an area of around 15,000 km² in the most densely populated region (Carvalho *et al.* 1999), with the highest level of deforestation and habitat degradation, in the entire Brazilian Amazon. More than 70% of the forest has been destroyed in the process of converting land to farmland and pasture (Carvalho *et al.* 1999; Almeida and Vieira, 2010). Deforestation continues, and most of the remaining forests now comprise isolated, usually hunted and degraded, patches of farmland. *Cebus kaapori* occurs in only two protected areas: the Gurupi Biological Reserve and the Lago de Tucuruí Environmental Protection Area. A large part of the forest of the Gurupí Biological Reserve has been logged and destroyed since its creation in 1988, and other pressures such as wildlife traffic and drugs plantations affect this reserve. Ferrari and Lopes (1996) estimated a density of 0.98 individuals/km² there. Another survey revealed a relative abundance of 0.99 groups/10 km in the Fazenda Cauaxi in Paragominas (Carvalho *et al.* 1999). Lopes (1993) saw three groups in 480 km in the Gurupí Biological Reserve which means 0.06 groups/10 km. Recently, Buss *et al.* (2014) found 0.25 groups/10 km at the same Gurupí Biological Reserve.

Campos (2009), using Population Viability Analysis Vortex software, found that only three populations could be considered viable in the long term (100 years). These populations are found in Caru, Awá, Alto Turiaçu, Araraibóia Indigenous Areas in Maranhão State, and Alto Rio Guama Indigenous Area in Pará State, beside Gurupí Biological Reserve.

Due to the threats of habitat loss and hunting, and a drastic population reduction (more than 80% over the past three generations (48 years)), *C. kaapori* is classified as Critically Endangered on the IUCN Red List (Kierulff and Oliveira 2008), the same category it received in the national assessment of Brazil (Fialho *et al*. 2015), where it is endemic. Lopes and Ferrari (1993) and Ferrari and Queiroz (1994) concluded that *C. kaapori* is one of the most threatened of all the Amazonian primates. It would seem that the Ka'apor Capuchin is naturally rare; it is hunted and is susceptible to any, even light, disturbance or degradation of its habitat. For example, selective logging of trees providing fruit, which forms a significant part of the diet, is a considerable threat for this species (Lopes 1993). Why it is so rare may be related to competition with the sympatric Guianan brown capuchin (*Sapajus apella*) and naturally low densities may reflect the need for large home ranges. *Cebus kaapori* is maintained in only two zoological institutions; the Centro de Primatologia do Rio de Janeiro (CPRJ) and Fundação Parque Zoológico de São Paulo (Marcos Fialho, unpublished data). Guajá Indians keep them as pets (Queiroz 1992).

More recently, an inventory of primate species, including *Cebus kaapori*, inhabiting the "arc of deforestation" in the Brazilian Amazon was carried out as well as an abundance study at the Gurupi Biological Reserve. Partial results show that this species has a healthy population found in this reserve, despite anthropogenic pressures affecting the area (Buss *et al*. 2014).

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San Martín Titi Monkey

Callicebus oenanthe Thomas, 1924

Peru

(2012, 2014)

Jan Vermeer

San Martín titi monkey (*Callicebus oenanthe*) (Illustrations: Stephen D. Nash)

The San Martín titi monkey was discovered in 1924, but until 2007 was only known from six museum specimens and scarce observations, all from the Alto Mayo Valley in northeastern Peru (Thomas, 1924, 1927; Hershkovitz 1990; Mark 2003; Rowe and Martinez 2003; De Luycker 2006). Extensive surveys by the team of Proyecto Mono Tocón have shown that the distribution of the species extends from the Alto Mayo Valley in the south, restricted largely (but not completely) by mountain ranges in the west, south and north, and the Río Huallaga in the east (Boveda-Penalba *et al*. 2009). It inhabits the lowland forest on the eastern foothills of the Andes, rarely occurring at altitudes above 1,200 m above sea level.

Callicebus oenanthe is endemic to the department of San Martín, which has the highest deforestation rates in Peru. Although its original range was estimated to have been approximately 14,000 km², its habitat has been reduced to less than 6,500 km², of which only 1,900 km² is thought to be covered with good habitat (Shanee *et al*. 2013). The forest cover data used for this study were from 2007/2008; meanwhile deforestation has been relentless and the situation is even worse today.

The San Martín titi monkey is highly variable in coloration (Boveda-Penalba *et al*. 2009; Vermeer *et al*. 2011). Most animals in the north are brownish with a white mask, while in the south many lack the typical mask and have a darker or more orange color (Proyecto Mono Tocón unpublished data).

Only a few larger populations are living in protected areas, and it is doubted that these populations are viable. Most San Martín titi monkeys were found to live in isolated populations in small forest fragments, with little chances for survival. Connecting isolated forest patches is mostly impossible due to human presence. The situation is even more complicated as the San Martín titi monkey seems to prefer the edges between primary and secondary forest, where

human pressure is often very high (Proyecto Mono Tocón unpublished data). The species can be found on the borders of some protected areas. Although a number of (relatively) small conservation concessions and private conservation areas have been created in the range of the San Martín titi monkey, only two may harbor viable populations. Unfortunately, most of its habitat is still unprotected, and is rapidly destroyed for agriculture and logging.

The San Martín titi monkey is Critically Endangered (Veiga *et al*. 2011) as it is estimated that a population reduction of ≥80% has occurred over the last 25 years. The isolation of non-viable populations in small forest patches increases the risk for the species. More support from national and regional governments and (international) conservation organizations is urgently needed to save this species from extinction.

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Northern Brown Howler
Alouatta guariba guariba (Humboldt, 1812)
Brazil
(2012, 2014)

Leonardo Gomes Neves, Leandro Jerusalinsky, Anthony Rylands, Fabiano R. Melo & Maurício Talebi

Male northern brown howler(*Alouatta guariba guariba*)
(Illustration: Stephen D. Nash)

The brown howler is separated into two subspecies, the northern brown howler, *Alouatta guariba guariba*, and southern brown howler, *A. g. clamitans* (Rylands *et al.* 2000; Groves 2001, 2005). Following a study of the morphology of the cranium and hyoid apparatus of the two forms, Gregorin (2006) considered them to be full species, using the name *A. fusca* (É. Geoffroy Saint-Hilaire, 1812) rather than *A. guariba* (Humboldt, 1812) for the northern form, following the recommendation of Hershkovitz (1963). Rylands and Brandon-Jones (1998) argued that the correct name is in fact *guariba*. Kinzey (1982) concluded that *A. g. guariba* occurred north of the Rio Doce; *clamitans* to the south. Rylands *et al.* (1988) observed what they believed to be *A. g. clamitans* further north, in the middle Jequitinhonha valley, and indicated that the Rio Jequitinhonha basin, not the Rio Doce, divided the two howlers. The extreme rarity of brown howlers north of the Jequitinhonha has confounded attempts to clarify the taxonomy. Only recently have few and minuscule populations been located in southern Bahia (Neves *et al.* 2015; L. G. Neves, unpublished data). Gregorin (2006) argued that the original range of the northern brown howler in fact extended from Bahia (Rio Paraguaçú) south along the coastal forest to the state of Rio de Janeiro (crossing as such the lower and middle Rio Doce), and that *clamitans*, the southern form, occurs inland north as far as the upper and middle Jequitinhonha. This would be compatible with the findings of Rylands *et al.* (1988) in the Jequitinhonha valley and, in this case, some of the populations surveyed by Chiarello (1999) may have been of the northern subspecies *A. g. guariba*. Here, we maintain the names and subspecific classification as used by Rylands *et al.* (2000), Groves (2001, 2005), and Glander (2013).

Both sexes of *A. g. guariba* are a red-fawn colour, the females being rather duller in colour. *Alouatta g. guariba* inhabits lowland, submontane and montane Brazilian Atlantic forest. It is a folivore-frugivore, including more fruit in its diet according to seasonal availability (Neville *et al.* 1988; Mendes 1989; Chiarello 1994; Glander 2013; Rylands and Mittermeier 2013). As such, brown howler monkeys are important seed dispersers for several plant

species (Chiarello and Galetti 1994). While the parent species *Alouatta guariba* is widely distributed and is classified as Least Concern on the IUCN Red List, *A. g. guariba* has a considerably more restricted range and is classified as Critically Endangered (Mendes *et al.* 2008), the same category it received in the Brazilian red list (Neves *et al.* 2015). The primary threats are widespread forest loss and fragmentation throughout its range, due to logging and agriculture (Horwich 1998) and hunting (Melo 2005; Canale *et al.* 2012). Disease epidemics such as yellow fever that affected *A. guariba clamitans* and *A. caraya* (Holzmann *et al.* 2010) could represent an additional threat to this taxa.

A conservation project for *A. g. guariba* is now ongoing, as an immediate effect of federal conservation public policy, an action plan for 27 threatened mammals of the Brazilian Atlantic Forest that includes the species (Brazil, MMA, ICMBio-CPB 2010). Surveys carried out since 2012, by the Instituto de Estudos Sócioambientais do Sul da Bahia (IESB), the State University of Santa Cruz (UESC) (with the support of Conservation International, the Rainforest Trust, and the Mohamed Bin Zayed Fund) and the ICMBio/Centro de Primatas Brasileiros, have attempted to locate and count surviving populations, to understand better the threats to their survival, and establish the limits to their geographic distribution. To date - and after 24 months of field research - ten populations remain in small and widely separated forest patches. The surveys found 17 groups and 31 individuals in the following locations: 1) Itajú de Colônia – two groups and one individual seen; 2) Itarantim – two different groups, vocalizations only; 3) Caatiba – three groups and a total of nine individuals; 4) Itapetinga – two different groups, vocalizations only; 5) Macarani – one group, nine individuals; 6) Ribeirão Largo – one group heard, vocalizations only; 7) Pouso Alegre – one group, two individuals; 8) Itambé – two groups, vocalizations only; 9) Boa Nova – one group, two individuals (L. G. Neves, unpublished data). The surveys indicate that most of the surviving populations are those in the valleys of the Rio Pardo and Rio Jequitinhonha. Further north, in the cacao-growing region of southern Bahia, they have been largely hunted out. The most recent expedition for this region of northern Bahia took place in July 2015 and confirmed a reduction in the geographic distribution of the species now ranging up north to the Boa Nova municipality, which is about 200 km from the Rio Paraguaçú, the historical original distribution's northernmost point.

There are a number of protected areas in the northern brown howler's range in Bahia and northeastern Minas Gerais, all created since 1980. Nevertheless, the only strictly protected area where they have been confirmed is the Mata Escura Biological Reserve (51,046 ha, created in 2003), just north of the middle Rio Jequitinhonha, where they coexist with the also Critically Endangered *Brachyteles hypoxanthus* and the Endangered *Sapajus xanthosternos* (Melo 2005). This reserve is being constantly impacted by the nearby rural settlements, with fires, logging and hunting, besides other interests in the area such as quilombola communities. Adding the locations in the lower reaches of Jequitinhonha basin reported by Rylands *et al.* (1988), the known population today is unlikely to number more than 250 mature individuals, and no subpopulation is believed to exceed 50 mature individuals (Neves *et al.* 2015). Howlers have not been seen further north in the Una Biological Reserve (18,500 ha, created in 1980) for more than 60 years. It is not known if they still occur in the submontane and montane forest of the Serra das Lontras National Park (11,336 ha, created in 2010). Future surveys will target protected areas and the limits of their supposed range—the Rio Paraguaçú in the north to the Rio Doce in the south, and protected areas in southern Bahia. An ongoing initiative is underway at the Serra Bonita Private Reserve, Camacan, Bahia, owned by Vitor Becker, and managed by the NGO Instituto Uiraçú, with, in collaboration with the CPPB/ICMBio, the successful release of two confiscated pets—an incipient reintroduction of the species that has not been seen or heard there for more than 50 years. A promising initiative has begun in the extreme south of the state of Bahia in the Pau-Brasil National Park (19,027 ha, created in 1999), and surrounding private reserves. This is led by the park administrators in partnership with researchers of the Federal University of São Paulo-Diadema. The aim of this project is to elucidate initial demographic parameters and to establish the species as a conservation flagship for the region at this critical protected habitat, thus enhancing public awareness for the conservation of this threatened primate.

Overall, the main conservation threat to wild populations is hunting. It has resulted in the surviving populations being very small and isolated. A future metapopulation management plan will need to incorporate translocation of threatened populations.

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The **Species Survival Commission (SSC)** is one of six volunteer commissions of IUCN, a union of sovereign states, government agencies and non-governmental organizations. SSC's mission is to conserve biological diversity by developing and executing programs to save, restore and wisely manage species and their habitats. Survival of the world's living primate species and subspecies is the principal mission of the IUCN SSC Primate Specialist Group (PSG), over 500 volunteer professionals who represent the front line in international primate conservation. The PSG website is www.primate-sg.org.

The **International Primatological Society (IPS)** was created to encourage all areas of non-human primatological scientific research, to facilitate cooperation among scientists of all nationalities engaged in primate research, and to promote the conservation of all primate species. The Society is organized exclusively for scientific, educational and charitable purposes. For more information about IPS, visit www.internationalprimatologicalsociety.org.

Conservation International (CI). Building upon a strong foundation of science, partnership and field demonstration, CI empowers societies to responsibly and sustainably care for nature, our global biodiversity, for the well-being of humanity. With headquarters in Arlington, VA, CI works in more than 40 countries on four continents. For more information about CI, visit www.conservation.org.

Bristol Zoological Society (BZS) runs Bristol Zoo Gardens and the Wild Place Project. BZS undertakes conservation action and conservation research in both the UK and the developing world. Its mission is to save wildlife through conservation action and engaging people with the natural world. For more information about BZS, visit www.bristolzoo.org.uk.

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