

Primates in Peril

The World's 25 Most Endangered Primates
2016–2018



Edited by

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



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Illustrations by Stephen D. Nash

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Front cover photo: Gee's golden langur (*Trachypithecus geei*). Photo © Dilip Chetry

Back cover photo: White-thighed colobus (*Colobus vellerosus*). Photo © Luke Larter

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Here, we present the 2016–2018 iteration of the World's 25 Most Endangered Primates list, drawn up during an open meeting held during the XXVI Congress of the International Primatological Society (IPS), Chicago, 22 August 2016.

We have updated the species profiles from the 2014–2016 edition (Schwitzer *et al.* 2015) 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: 2016–2018

Here we report on the ninth 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 2016–2018 list of the world's 25 most endangered primates has five species from Africa, six from Madagascar, nine from Asia, and five from the Neotropics (Table 1). Madagascar tops the list with six species. Indonesia has four, Brazil, Colombia, Côte d'Ivoire, Ghana, Panama, Vietnam, and possibly Nigeria have two, and Benin, Bhutan, China, Costa Rica, Democratic Republic of Congo, Ecuador, El Salvador, Guatemala, Honduras, India, Mexico, Nicaragua, Sri Lanka, Tanzania, and Togo each have one.

The changes made in this list compared to the previous iteration (2014–2016) were not because the situation of the twelve species that were dropped (Table 2) has improved, although for some species, such as, for example, *Lepilemur septentrionalis*, better protection measures are now in place. By making these changes we intend rather to highlight other, closely related

species enduring equally bleak prospects for their future survival.

Twelve of the primates were not on the previous (2014–2016) list (Table 3). Ten of them are listed among the world's most endangered primates for the first time. The Niger Delta red colobus and Bornean orangutan had already been on previous iterations, but were subsequently removed in favour of other highly threatened species. The 2016–2018 list contains two members each of the genera *Trachypithecus* and *Ateles*, thus particularly highlighting the severe threats that large-bodied primates are facing in all of the world's primate habitat regions.

During the discussion of the 2016–2018 list at the XXVI Congress of IPS in Chicago in 2016, 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 2016–2018.

| | | |
|-------------------------------------|--------------------------------|---|
| Africa | | |
| <i>Paragalago orinus</i> | Mountain galago | Tanzania |
| <i>Cercopithecus roloway</i> | Roloway monkey | Ghana, Côte d'Ivoire |
| <i>Colobus vellerosus</i> | White-thighed colobus | Côte d'Ivoire, Ghana, Togo, Benin, possibly Nigeria |
| <i>Ptilocolobus epieni</i> | Niger Delta red colobus | Nigeria |
| <i>Gorilla beringei graueri</i> | Grauer's gorilla | Democratic Republic of Congo |
| Madagascar | | |
| <i>Microcebus gerpi</i> | Gerp's mouse lemur | Madagascar |
| <i>Hapalemur alaotrensis</i> | Lac Alaotra bamboo lemur | Madagascar |
| <i>Lemur catta</i> | Ring-tailed lemur | Madagascar |
| <i>Lepilemur jamesorum</i> | James' sportive lemur | Madagascar |
| <i>Propithecus perrieri</i> | Perrier's sifaka | Madagascar |
| <i>Daubentonia madagascariensis</i> | Aye-aye | Madagascar |
| Asia | | |
| <i>Nycticebus javanicus</i> | Javan slow loris | Indonesia |
| <i>Simias concolor</i> | Pig-tailed snub-nosed langur | Indonesia |
| <i>Trachypithecus poliocephalus</i> | Golden-headed or Cat Ba langur | Vietnam |
| <i>Rhinopithecus avunculus</i> | Tonkin snub-nosed monkey | Vietnam |
| <i>Trachypithecus geei</i> | Gee's golden langur | India and Bhutan |
| <i>Semnopithecus vetulus</i> | Western purple-faced langur | Sri Lanka |
| <i>Macaca nigra</i> | Crested macaque | Indonesia |
| <i>Nomascus hainanus</i> | Hainan gibbon | China |
| <i>Pongo pygmaeus</i> | Bornean orangutan | Indonesia (Borneo) |
| Neotropics | | |
| <i>Plecturocebus caquetensis</i> | Caquetá titi monkey | Colombia |
| <i>Cebus kaapori</i> | Kaapor capuchin | Brazil |
| <i>Alouatta guariba guariba</i> | Northern brown howler | Brazil |
| <i>Ateles geoffroyi</i> | Geoffroy's spider monkey | Mexico, Guatemala, Nicaragua, Honduras, El Salvador, Costa Rica, Panama |
| <i>Ateles fusciceps</i> | Brown-headed spider monkey | Ecuador, Colombia, Panama |

Table 2. Primate species included on the 2014–2016 list that were removed from the 2016–2018 list.

| | | |
|----------------------------------|------------------------------|---------------------|
| Africa | | |
| <i>Galagoides rondoensis</i> | Rondo dwarf galago | Tanzania |
| <i>Ptilocolobus preussi</i> | Preuss's red colobus | Cameroon, Nigeria |
| <i>Ptilocolobus rufomitratu</i> | Tana River red colobus | Kenya |
| Madagascar | | |
| <i>Cheirogaleus lavasoensis</i> | Lavaso Mountains dwarf lemur | Madagascar |
| <i>Varecia rubra</i> | Red ruffed lemur | Madagascar |
| <i>Lepilemur septentrionalis</i> | Northern sportive lemur | Madagascar |
| Asia | | |
| <i>Carlito syrichta</i> | Philippine tarsier | Philippines |
| <i>Trachypithecus delacouri</i> | Delacour's langur | Vietnam |
| <i>Semnopithecus ajax</i> | Chamba sacred langur | India |
| <i>Pongo abelii</i> | Sumatran orangutan | Indonesia (Sumatra) |
| Neotropics | | |
| <i>Ateles hybridus</i> | Brown spider monkey | Colombia, Venezuela |
| <i>Callicebus oenanthe</i> | San Martín titi monkey | Peru |

Table 3. Primate species that were added to the 2016–2018 list. The Niger Delta red colobus and Bornean orangutan were added to the list after previously being removed. The other ten species are new to the list.

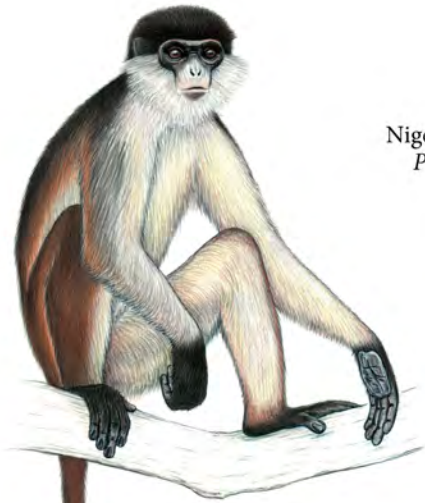
| | | |
|-------------------------------------|--------------------------|---|
| Africa | | |
| <i>Paragalago orinus</i> | Mountain galago | Tanzania |
| <i>Colobus vellerosus</i> | White-thighed colobus | Côte d'Ivoire, Ghana, Togo, Benin, possibly Nigeria |
| <i>Ptilocolobus epieni</i> | Niger Delta red colobus | Nigeria |
| Madagascar | | |
| <i>Microcebus gerpi</i> | Gerp's mouse lemur | Madagascar |
| <i>Lemur catta</i> | Ring-tailed lemur | Madagascar |
| <i>Lepilemur jamesorum</i> | James' sportive lemur | Madagascar |
| <i>Daubentonia madagascariensis</i> | Aye-aye | Madagascar |
| Asia | | |
| <i>Trachypithecus geei</i> | Gee's golden langur | Vietnam |
| <i>Macaca nigra</i> | Celebes crested macaque | India |
| <i>Pongo pygmaeus</i> | Bornean orangutan | Indonesia (Borneo) |
| Neotropics | | |
| <i>Plecturocebus caquetensis</i> | Caquetá titi monkey | Colombia |
| <i>Ateles geoffroyi</i> | Geoffroy's spider monkey | Mexico, Guatemala, Nicaragua, Honduras, El Salvador, Costa Rica, Panama |

Table 4. Primate species considered during the discussion of the 2016–2018 list at the IPS Congress in Chicago that did not make it onto the list, but are also highly threatened.

| | | |
|----------------------------|--------------------------|---------------------------------|
| Asia | | |
| <i>Nycticebus bancanus</i> | Bangka slow loris | Indonesia (Borneo, Bangka Isl.) |
| <i>Tarsius</i> sp. | Mantehage Island tarsier | Indonesia (Mantehage Island) |
| Neotropics | | |
| <i>Saguinus leucopus</i> | Silvery-brown tamarin | Colombia |



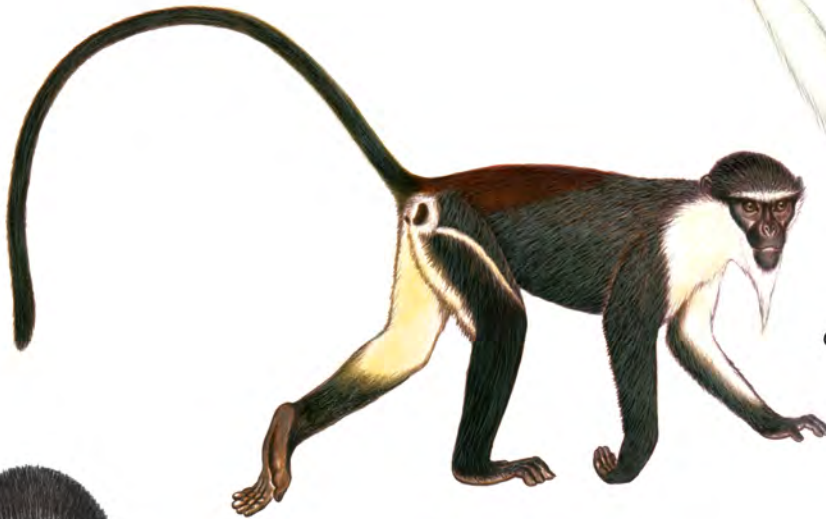
Photos of some of the Top 25 Most Endangered Primates. From top to bottom, left to right: 1. *Microcebus gerpi* (photo by Blanchard Randrianambinina); 2. *Nomascus hainanus* (juvenile)(photo by Zhao Chao); 3. *Plectrocebus caquetensis* (juvenile) (photo by Thomas R. Defler); 4. *Alouatta guariba guariba* (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. *Lepilemur jamesorum* (photo by Edward E. Louis, Jr.); 9. *Trachypithecus geei* (photo by Dilip Chetry); 10. *Trachypithecus poliocephalus* (photo by Tilo Nadler); 11. *Gorilla beringei graueri* (photo by Russell A. Mittermeier); 12. *Pongo pygmaeus* (photo by Russell A. Mittermeier); 13. *Nycticebus javanicus* (photo by K. Anna I. Nekaris); 14. *Daubentonia madagascariensis* (juvenile) (photo by Edward E. Louis, Jr.); 15. *Ateles geoffroyi* (photo by Russell A. Mittermeier).



Niger Delta red colobus
Piliocolobus epieni
Africa



White-thighed colobus
Colobus vellerosus
Africa



Roloway monkey
Cercopithecus roloway
Africa



Grauer's gorilla
Gorilla beringei graueri
Africa



Mountain galago
Paragalago orinus
Africa



Gerp's mouse lemur
Microcebus gerpi
Madagascar



Aye-aye
Daubentonia
madagascariensis
Madagascar



James' sportive lemur
Lepilemur jamesorum
Madagascar

Perrier's sifaka
Propithecus perrieri
Madagascar



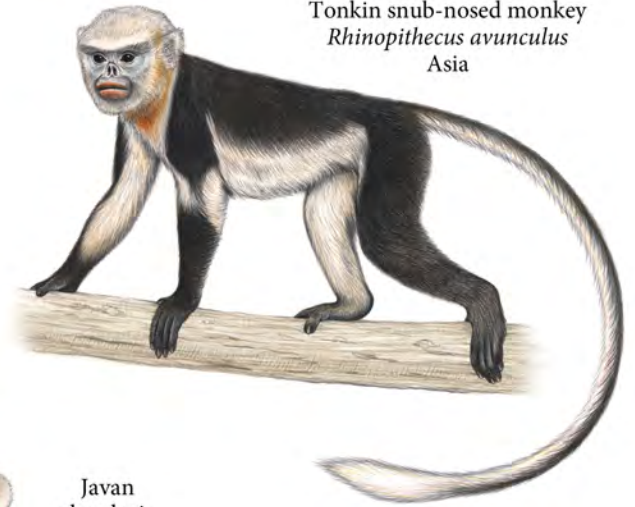
Lac Alaotra Bamboo lemur
Hapalemur alaotrensis
Madagascar

Ring-tailed lemur
Lemur catta
Madagascar





Crested macaque
Macaca nigra
Asia



Tonkin snub-nosed monkey
Rhinopithecus avunculus
Asia



Javan
slow loris
*Nycticebus
javanicus*
Asia



Bornean orangutan
Pongo pygmaeus
Asia



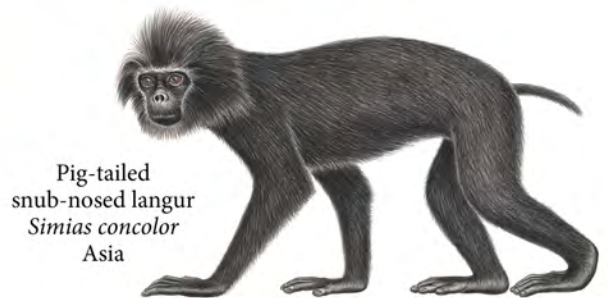
Western
purple-faced langur
Semnopithecus vetulus
Asia



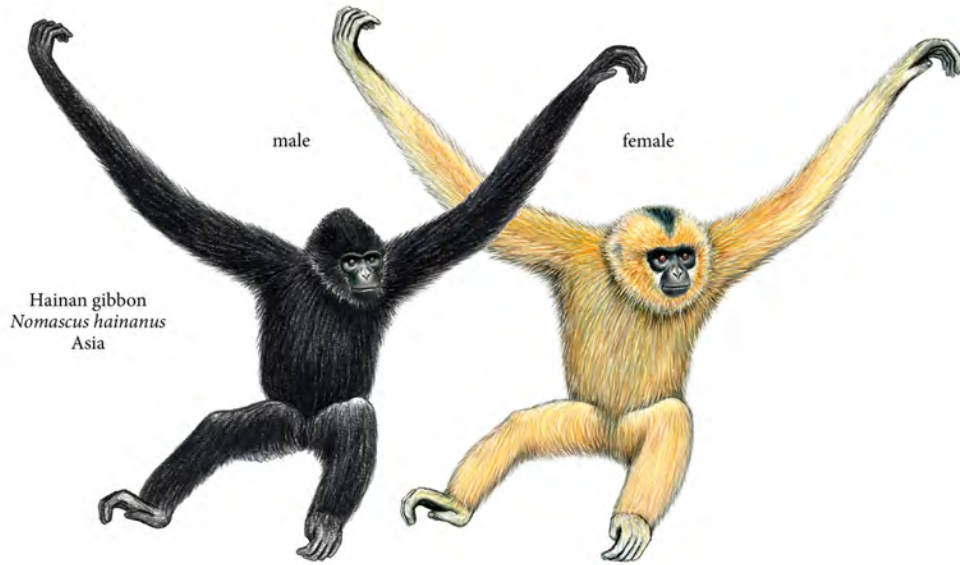
Gee's golden langur
Trachypithecus geei
Asia



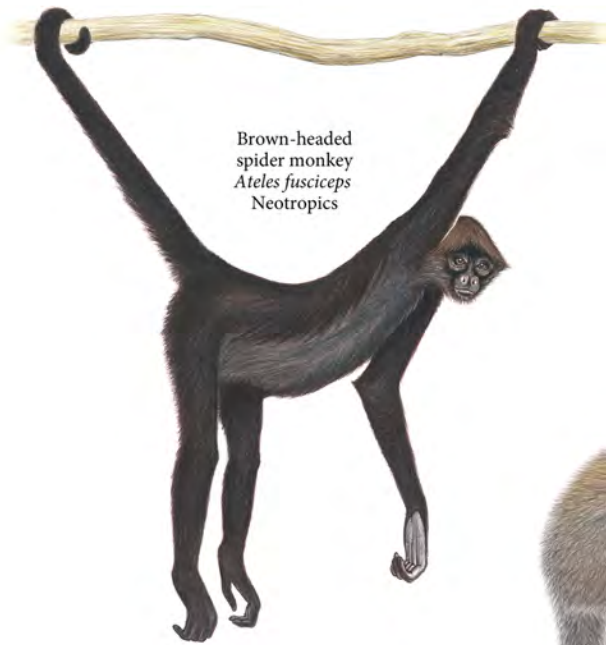
Golden-headed langur
or Cat Ba langur
*Trachypithecus
poliocephalus
poliocephalus*
Asia



Pig-tailed
snub-nosed langur
Simias concolor
Asia



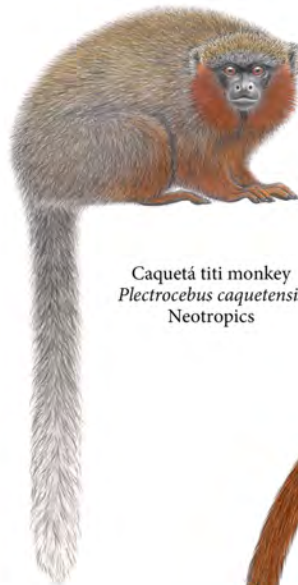
Hainan gibbon
Nomascus hainanus
Asia



Brown-headed
spider monkey
Ateles fusciceps
Neotropics



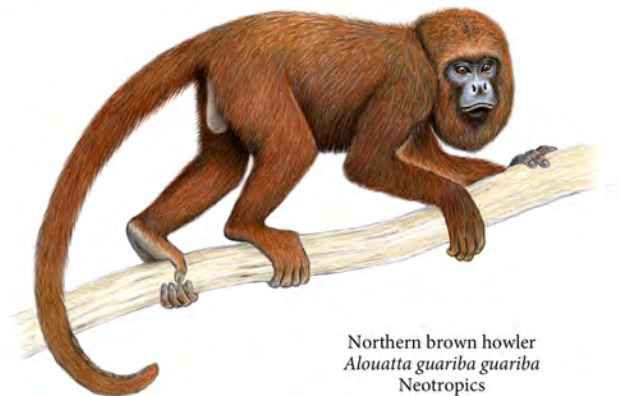
Geoffroy's
spider monkey
Ateles geoffroyi
Neotropics



Caquetá titi monkey
Plectrocebus caquetensis
Neotropics

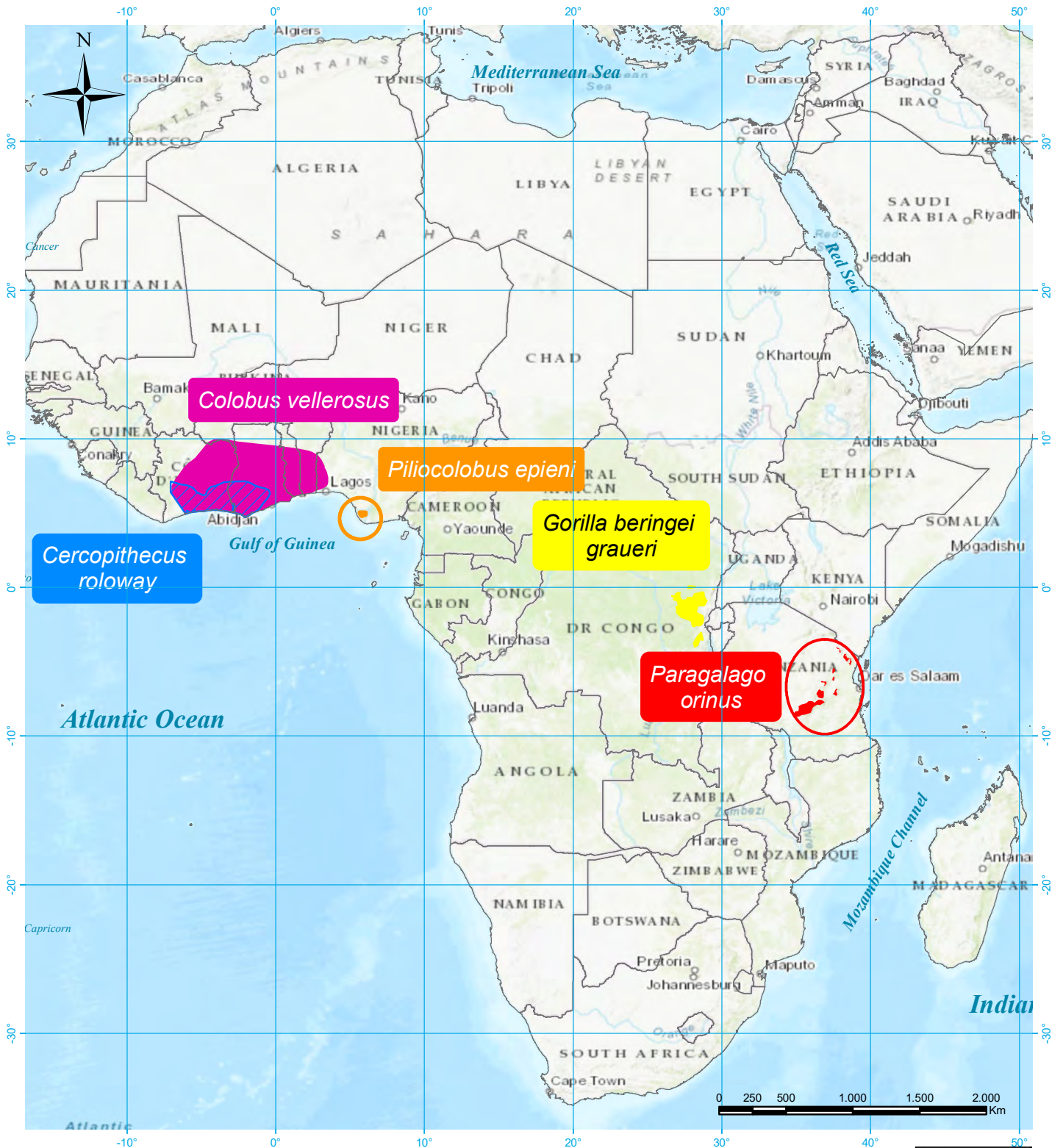


Ka'apor capuchin
Cebus kaapori
Neotropics



Northern brown howler
Alouatta guariba guariba
Neotropics

Africa



African Primates

- | | | | |
|---|------------------------------|---|---------------------------------|
|  | <i>Cercopithecus rolaway</i> |  | <i>Gorilla beringei graueri</i> |
|  | <i>Colobus vellerosus</i> |  | <i>Piliocolobus epieni</i> |
|  | <i>Paragalago orinus</i> | | |



Mountain Galago
Paragalago orinus (Lawrence & Washburn, 1936)
Tanzania
(2016)

Andrew Perkin

Paragalago orinus is one of the smallest of all galago species, identified by its distinctive “double unit repetitive call” together with a very bushy tail and long nose stripe (Perkin *et al.* 2013). Male genital morphology is also characteristic (Perkin 2007). The taxonomy of this galago taxon has changed since being originally described as *Galago demidovii orinus* Lawrence and Washburn, 1936, from a specimen taken from the Uluguru Mountains. Honess (1996) and Honess and Bearder (1996) recognised the taxon as the dwarf mountain galago *Galagoides orinus* based on differences in vocalizations and morphology, which was subsequently accepted by Kingdon (1997), Groves (2001), Grubb *et al.* (2003), Mittermeier *et al.* (2013), and Kingdon (2015). Recently, Masters *et al.* (2017) placed this taxon in a new East African dwarf galago genus *Paragalago*.

Current knowledge indicates that this species is endemic to Tanzania in the Eastern Arc Mountains. However, they only appear to occur in seven out of eleven of the Eastern Arc Mountains. Surveys have revealed populations in the Uluguru, Rubeho, Udzungwa, Nguru, Ukaguru, East Usambara Mtns., and West Usambara Mtns. (Lawrence and Washburn 1936; Allen and Loveridge 1927; Honess 1996; Butynski *et al.* 1998; Perkin 2001). The Mahenge (Bracebridge 2004), and the N. Pare (Doggart *et al.* 2008) and S. Pare (Gwegime *et al.* 2014) mountains do not appear to hold any *P. orinus*. Sister populations of unidentified dwarf galagos occur in the Taita Hills, Kenya (Perkin *et al.* 2002), and the Southern Highlands, SW Tanzania (Perkin, unpubl. data) and have to date not been identified as *P. orinus*. There are indications that significant variation exists between the identified populations of *P. orinus*. Intraspecific variation in vocal patterns between two different mountain block populations is known in *P. orinus* (Perkin *et al.* 2013).

Mountain dwarf galagos have a mixed diet of insects and fruit, feeding close to the ground as well as mid-storey and the forest canopy, and move by vertical clinging and leaping. For daytime sleeping sites they use tree holes as



The Mountain Galago (*Paragalago orinus*)
(Illustration: Stephen D. Nash)

well as large nests, but it is unknown if they are active nest builders or just occupy squirrel dreys (Bearder *et al.* 2003). *Paragalago orinus* is probably subject to predation by owls and other nocturnal predators. Among these, genet, palm civets and snakes invoke intense episodes of alarm calling (Perkin *et al.* 2013).

The conservation status of *P. orinus* has remained as Near Threatened since 2008 (IUCN 2017). No detailed surveys have been conducted to assess the population

status of *P. orinus*, but distribution surveys estimate the area of occurrence to be 2,375 km². However, this is a very fragmented population occurring in forests that are split across seven isolated mountain blocks ranging from 141–1,552 km. These mountain blocks vary in their level of protection and degree of threat. However, most remaining populations of *P. orinus* are within formally designated protected areas; forest reserves, nature reserves under the Tanzania Forestry Service, or national parks under the Tanzania National Parks Authority, and some patches of privately-owned forest. *Paragalago orinus* is threatened by habitat loss due to agricultural expansion, and alteration because of illegal logging. It can occur in habitats altered by logging, but there are no data to know if this is an adaptation or due to survey bias. Whilst the increased protection status afforded by the national parks (Udzungwa Mountains NP) and nature reserves (Udzungwa Scarp, Kilombero, Amani, Mgamba, and Uluguru NR) has reduced these threats, the overall population trend can be inferred to be downwards since habitat degradation and clearance still continues at varying levels.

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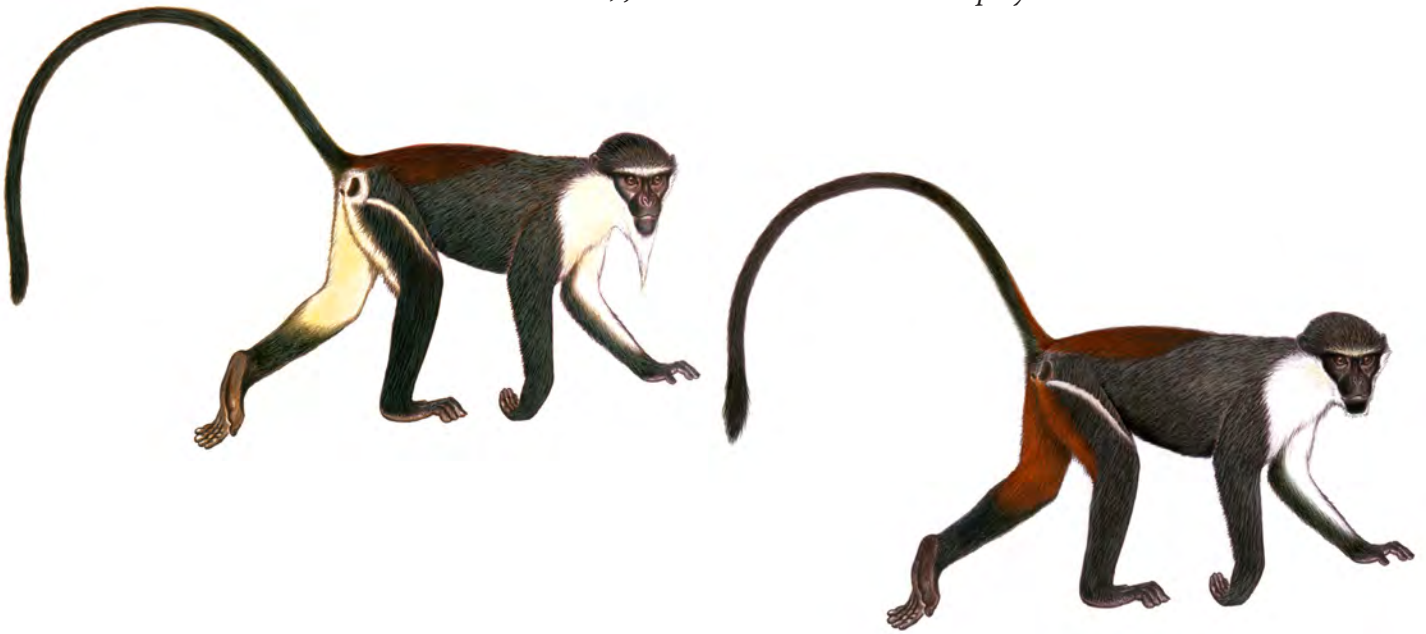
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Roloway Monkey
Cercopithecus roloway (Schreber, 1774)
Ghana, Côte d'Ivoire
(2002, 2006, 2008, 2010, 2012, 2014, 2016)

W. Scott McGraw, John F. Oates & Andrea Dempsey



The Roloway Monkey (*Cercopithecus roloway*) left, and the closely-related Diana Monkey (*Cercopithecus diana*), right (Illustrations: Stephen D. Nash)

Cercopithecus roloway and its close relative *Cercopithecus diana* are highly attractive, arboreal monkeys that inhabit the Upper Guinean forests of West Africa. The roloway monkey, which once occurred in many of the southern forests of Ghana and central and eastern Côte d'Ivoire, is distinguished from the Diana monkey by its broad, white brow line, long, white beard and yellow thighs. Because individuals with intermediate coat patterns are known from near the Sassandra River in Côte d'Ivoire, some scientists treat the roloway and Diana as subspecies of one species, *C. diana* (Oates 2011). Of the two forms, the roloway is the more seriously threatened with extinction, and it is now rated as Critically Endangered on the IUCN Red List (Koné *et al.* 2017).

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 (*Piliocolobus waldroni*) once inhabited many of the same forest areas as the roloway, but is now almost certainly extinct (Oates 2011). Unless much more

effective conservation action is taken very quickly, there is a strong possibility that the roloway monkey will also disappear in the near future.

Over the last 50 years, roloway monkeys have been steadily extirpated in Ghana. In south-western Ghana – once a stronghold of *C. roloway* – an ornithological study showed a 600% increase in both legal and illegal logging between 1995 and 2008 (Arcilla *et al.* 2015). Illegal logging, which makes up 80% of timber harvested in Ghana, is particularly devastating; because it is wholly unregulated or monitored, there are no limits on number, size or species of trees taken. One-third of illegal logging is by companies that take more than their quota, expand into protected areas, and/or continue to log after their permit has expired. The remaining two-thirds are rogue illegal chainsaw operators (Arcilla *et al.* 2015). Additional factors causing the roloway monkey's decline are clearing for agriculture, charcoal production and bushmeat hunting. Hunting has very likely been the major cause in the recent crash in roloway populations; bushmeat is a major food source for Ghanaians, with an estimated 80% of the rural population dependent on bushmeat as their main source of protein (Dempsey 2014; Trench 2000).

Several recent surveys have failed to confirm the presence of roloway monkeys in any reserves in western Ghana, including the Ankasa Conservation Area, 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). Community-owned forests along the Tano River (referred to as the “Kwabre Community Rain Forest”) in the far southwestern corner of the country are the only localities in Ghana where any roloways have been recorded 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 2014; Dempsey 2014; Osei et al. 2015). WAPCA has supported a community-based conservation project with villages around these forests, establishing a Kwabre Community Resource Management Area, which works to protect the forest through the sustainable management of natural resources. Meanwhile, further efforts should be made to ascertain whether any roloway monkeys still survive in Ankasa, because this site has significant conservation potential and roloways have been reported there in the relatively recent past, as well as the Amazuri Wetlands Area.

In neighbouring Côte d’Ivoire, the Rolloway guenon’s status is perhaps even more dire. Less than twelve 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 Iles Ehotilé (McGraw 1998, 2005; Koné and Akpatou 2005; Gonedelé Bi *et al.* 2013). 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 Iles Ehotilé, Yaya Forest Reserve) within the last dozen years. 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. In June 2011, Gonedelé bi Sery observed one roloway individual in the Dassioko Sud Forest Reserve (Gonedelé Bi *et al.*

2014 in review; Bitty *et al.* 2013). In 2012, Gonedelé Bi and A. E. 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, including during the most recent patrols (February 2017). 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) had 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, and immediately across the Tano River from the Kwabre forest in Ghana. This wet forest also harbours 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 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 is evidently on the verge of extinction (Oates 2011). In addition, the captive population is now also so small that extinction in captivity is also a strong possibility (Lefaux and Montjardet 2016).

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White-Thighed Colobus

Colobus vellerosus (I. Geoffroy Saint-Hilaire, 1834)
Côte d'Ivoire, Ghana, Togo, Benin, possibly Nigeria
(2016)

Reiko Matsuda Goodwin, Georges Nobimé & Edward D. Wiafe



The White-thighed Colobus (*Colobus vellerosus*) (Illustration: Stephen D. Nash)

high demand by the kings and chiefs of West African kingdoms for ceremonies as well as for international trade as ornaments and clothes (Oates 1977).

The white-thighed colobus currently has a fragmented distribution from the area between the Sassandra and Bandama rivers in Côte d'Ivoire to Benin, perhaps extending to southwestern Nigeria (Matsuda Goodwin *et al.* 2017). It inhabits lowland rainforests, swamp forests, seasonally inundated semi-deciduous forests, savannah-woodlands and gallery forests (Oates 2011), typically concentrating its activity in the mid to upper canopy. In the absence of hunting, this species can tolerate moderate levels of habitat degradation and can live in forest edges, forest fragments, logged forest, and regenerating forest (Oates 2011). Thus, it is unregulated hunting that is the primary factor in the demise of this species across its entire range, although habitat loss and degradation have also contributed significantly to population decline. In Ghana, for example, logging, especially illegal logging, increased by 600% between 1995 and 2008 (Arcilla *et al.* 2015), and more than 50% of the forested area in most forêts classées (= forest reserves, abbreviated as FRs) in Côte d'Ivoire, Togo, and Benin have been either logged or converted to plantations for oil palm, cacao and rubber, or pasture and farmland in the last 50 years (Bitty *et al.* 2015; Osei *et al.* 2015).

The white-thighed colobus (*Colobus vellerosus*) is one of four species of black-and-white colobus monkeys in Africa. These species are notable for the spectacular loud roars of adult males that accompany impressive leaping displays in the canopy (Oates *et al.* 2000). Since 1980, *C. vellerosus* is recognized as a species distinct from the king colobus (*Colobus polykomos*) based on vocalizations and skull morphology (Oates and Trocco 1983). *Colobus polykomos* inhabits forests to the west of the distribution range of *C. vellerosus*. Both species have a mainly black body and a long, white tail, but the white-thighed colobus has silvery thighs and a white ruff that surrounds the face. The king colobus has black thighs and long streaks of grey-white hair at the margin of the face that mingle with black fur at the neck and shoulders (Oates 2011). During and before the 19th century, the beautiful skin of the white-thighed colobus was in

Comprehensive assessments of *C. vellerosus* populations in its range countries have not been attempted, although localized surveys have been conducted. Based on information from such surveys in Côte d'Ivoire, the status of this species appears dire. It has probably been extirpated from almost all the national parks (NP) and FRs where it formerly occurred (McGraw *et al.* 1998), and now survives mostly in community forests (CFs)

(Gonedelé Bi *et al.* 2010, 2013, 2014). The human population of Côte d'Ivoire has increased by 50% in the last 15 years, and two periods of civil unrest (2002–2004 and 2010–2011) had adverse effects on primate populations (Fischer 2004; Campbell *et al.* 2008). The white-thighed colobus was common in Comoé NP before the 1990s, but it has become extremely rare in recent years (J. Lapuente pers. comm.). Between 2000 and 2006, Gonedelé Bi *et al.* (2010, 2013) conducted surveys in the eastern part of Côte d'Ivoire, covering three national parks (Azagny, Banco, Îles Ehotilé), three forest reserves (Bossématié, Yaya, Mabi), two sacred groves (Dinaoudi and Soko), and the Tanoé community forest bordering the Ehy lagoon in the far southeast (adjacent to Kwabre Peat Swamp Forest in southwestern Ghana). This species was found only at Dinaoudi and in flooded forest at Tanoé at a low encounter rate of 0.04 groups/km (Gonedelé Bi *et al.* 2013). After the second period of civil unrest, Gonedelé Bi *et al.* (2014) conducted surveys between the Sassandra River and Bandama River and found a hybrid black-and-white colobus population at the Grébouo 1 sacred grove. This population has now been extirpated (Gonedelé Bi pers. comm.). Hunting still continues even in the Tanoé-Ehy Forest, where CEPA and WAPCA have been engaging in community conservation initiatives (Gonedelé Bi *et al.* 2013).

Although historically the white-thighed colobus had a wide distribution in Ghana (Grubb 1998), it has probably now been extirpated from almost all, if not all, FRs and NPs (Oates 2006; Gatti 2010; Wiafe 2013; Osei *et al.* 2015). A few groups still inhabit Bia NP, Mole NP (Burton 2010; Osei *et al.* 2015) and the Kakum Conservation Area, where enforcement against poaching has been strengthened in recent years (Wiafe 2016). In these forests, encounter rates were reduced by 95–100% in the last 30 years, suggesting >90% population decline (Matsuda Goodwin *et al.* 2016). Its continued occurrence at Yoyo FR is uncertain (Osei *et al.* 2015). Some individuals still seem to occur in Atewa Range FR (Globally Significant Biodiversity Area, GSBA), but unregulated mining, logging, and farming decreased the closed forest by 33% from 1986 to 2013 (Kusimi 2015). The status of a few individuals seen in 2012 in Kwabre Forest, across the Tano River from the Tanoé-Ehy Forest in Côte d'Ivoire, is uncertain (Osei *et al.* 2015; R. Horwich pers. comm.). The only locality where a stable population seems to be present is in and around the Boabeng-Fiema Monkey Sanctuary (BFMS) where 365 individuals were recorded in 2007

(Kankam *et al.* 2010). Nevertheless, recent population data from BFMS are lacking. Because surveys have not been conducted in many other protected areas and forests in Ghana, determining the precise status of this species for this country is difficult. Given the observed high levels of hunting and forest destruction, however, large populations are not expected to remain in any forest. For example, Digya NP may still sustain some individuals, but 46.7% of its closed forest has been lost, with farmland increasing twofold between 1985 and 2000 (Twumasi *et al.* 2016). Checks on five markets near Digya NP found no evidence of this species still being sold (Owusu-Ansah 2010).

The white-thighed colobus has probably been extirpated in Burkina Faso (Ginn and Nekaris 2014). In Togo, recent surveys found this species in Togodo Faunal Reserve, Fazao-Malfakassa NP and Yikpa CF (G. Segniagbeto pers. comm.). The species may be nearly extinct in Benin. Only 1–2 groups have been recently located in the Lama FR, and one group each occurs in the Kikélé sacred forest and Bonou swamp forest. Interviews suggest some individuals may still live in community forests near Kikélé (R.C. Ota pers. comm.). Pénésoulou FR had one group in July 2013 (R.C. Ota pers. comm.), but it was not observed during surveys in July 2016 (Matsuda Goodwin *et al.* 2016). It has been previously reported to occur in the FRs of Mt. Kouffé, Wari-Marou and Ouémé Supérieur, and in the Lokoli Forest (Nobimé *et al.* 2011), but surveys conducted in July 2016 found no groups in these forests (Matsuda Goodwin *et al.* 2016). The species' status in Upper Ouémé River swamp forests is uncertain. In Nigeria, this species is claimed to occur still in Old Oyo National Park (OONP) and Kainji Lake National Park (KLNP) (J. O. Orimaye pers. comm.). However, limited surveys conducted in OONP and several community forests at the western edge of Nigeria in December 2015 did not find this species (Matsuda Goodwin *et al.* 2016).

The decline of *C. vellerosus* populations has been relatively neglected in comparison with other Critically Endangered West African primates, such as the rolway monkey (*Cercopithecus rolway*). Yet the local extinction of *C. vellerosus* has been accelerating. Forest loss and degradation are increasing even at BFMS (Kankam *et al.* 2010; A. Elgart pers. comm.) and illegal logging and hunting has not abated in the Tanoé-Ehy forest (Osei *et al.* 2015). Furthermore, currently no protected areas in the region have any *in situ* conservation programmes for this species and there are no *ex situ* populations.

Unless urgent actions to safeguard the white-thighed colobus are taken, it could follow the path of Miss Waldron's red colobus (*Piliocolobus waldroni*) to extinction. To secure the future of this species, we recommend the following actions: (1) Enact and enforce tougher laws against hunting and other illegal activities in FRs and PAs and extend such laws to community forests where this species still occurs; (2) upgrade the Lama classified forest in Benin to national park status; (3) conduct an environmental impact assessment of the hydropower project on the Mono River that threatens to partially submerge Togodo NP in Togo; (4) conduct systematic surveys near Kikélé sacred forest and swamp forests in Benin, in FRs and NPs in Togo, along the southern Togo-Ghana border, in the Mole NP and Amazule Swamp Forest (near Kwabre) in Ghana, and in OONP and KLNP in Nigeria.

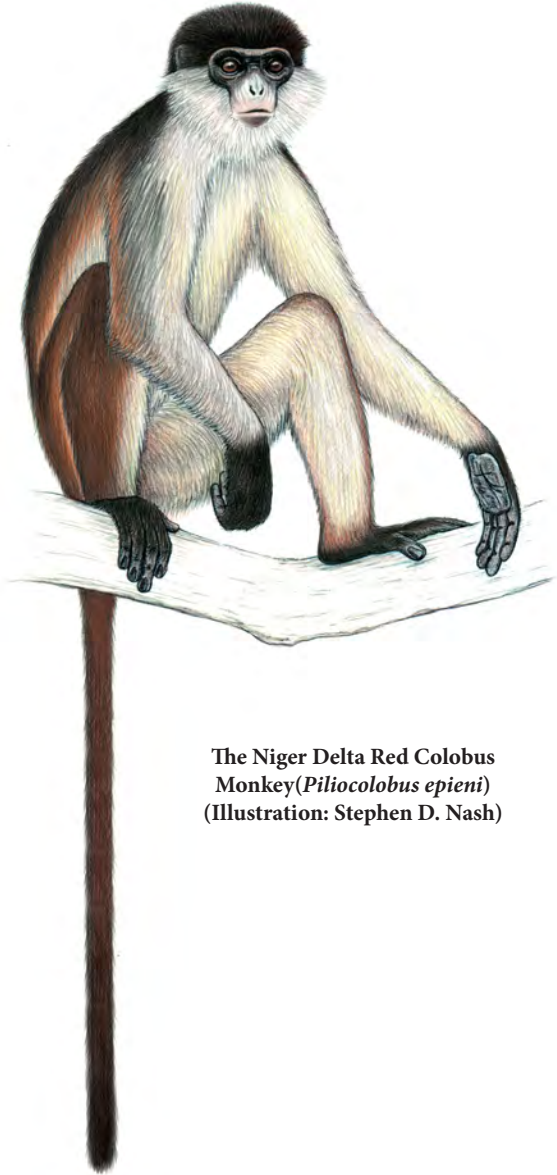
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Niger Delta Red Colobus Monkey
Piliocolobus epieni (Grubb and Powell, 1999)
Nigeria
(2008, 2010, 2016)

Rachel Ashegbofe Ikemeh & John F. Oates



**The Niger Delta Red Colobus
Monkey (*Piliocolobus epieni*)**
(Illustration: Stephen D. Nash)

mitochondrial DNA suggest that this population is not closely related to its closest geographic relatives, the Bioko red colobus (*Piliocolobus pennantii*) and Preuss's red colobus (*Piliocolobus preussi*) of E. Nigeria and W. Cameroon, leading Ting (2008) to treat this monkey not as a subspecies of *P. pennantii* (see Groves 2001; Grubb *et al.* 2003) but as a distinct species, *Procolobus epieni*. Groves (2007) regarded almost all the different forms of red colobus monkeys, including *epieni*, *pennantii* and *preussi* as separate species, in the genus *Piliocolobus* – a taxonomy that we follow here. Since 2008, *P. epieni* has been classified as Critically Endangered on the IUCN Red List of Threatened Species.

The marsh forests where the Niger Delta red colobus is found have a year-round high water table, but do not suffer deep flooding or tidal effects. The most intensive ecological study of this monkey, by Lodjewijk Werre (1994–1996), suggested that the clumped distribution of food species in the marsh forest is a key factor restricting *P. epieni* to its limited range, which is demarcated by the Forcados River and Bomadi Creek in the northwest, the Sagbama, Osiana and Apoi creeks in the east, and the mangrove belt to the south (Werre 2000). At the time of its discovery in the mid 1990s, this red colobus was locally common, especially in forests near the town of Gbanraun, but it was beginning to come under intense pressure from degradation of its habitat and commercial hunting. Important colobus food trees—especially *Hallea ledermannii*—were being felled at a high rate by artisanal loggers, the logs floated out of the Delta on rafts to processing centers in Lagos and elsewhere. In addition, large canals dug as part of oil extraction activities, as well as smaller canals dug by loggers into the interior swamps, were changing local hydrology (Werre and Powell 1997; Grubb and Powell 1999). The Ijaw people are traditionally fishermen but outside influences introduced by the oil industry have encouraged commercial bushmeat hunting and logging throughout the Niger Delta. The most recent range-wide assessment of *P. epieni* conducted in 2013

The Niger Delta red colobus (*Piliocolobus epieni*) is endemic to the marsh forests in the central part of the Niger Delta of Nigeria (Oates 2011). The species' name is derived from its name in the Ijaw language of the people who inhabit the limited area of about 1,500 km² in Bayelsa State where it occurs. *Piliocolobus epieni* only became known to science in 1993 in the course of a biodiversity survey co-ordinated by C. Bruce Powell (Powell 1994). Studies of vocalizations and

suggests that, as a result of habitat destruction and hunting, the population has declined significantly since the 1990s, and that it may now be around 90% lower than the previous estimate of ~10,000 (Ikemeh 2015). In the 2013 survey, the presence of *P. epieni* could be confirmed in only four community forests, and it was considered extirpated from 11 other forests where it had been reported in the 1990s by Werre (2000). Cumulative survey data indicate that the current number of individuals surviving in the wild may be only a few hundred (Ikemeh 2015). The Apoi creek forests flanked by the communities of Gbaraun, Apoi and Kokologbene, and forests near Kolotoro, were found in the 2013 survey to be the two most important remaining areas where this species can be conserved. Insecurity in the region and the consequences of corrupt governance are amongst factors that have exacerbated the major threats of habitat degradation and commercial hunting.

At present, there are no formal protected areas in the Niger Delta, even though it has great ecological significance and supports many rare, unique and/or threatened taxa. The Niger Delta red colobus shares its marsh forest habitat with two other threatened primates; the Nigerian white-throated guenon (*Cercopithecus erythrogaster pococki*) and the red-capped mangabey (*Cercocebus torquatus*), each listed as Endangered on the Red List. Also found in these forests are the putty-nosed monkey (*Cercopithecus nictitans*), the mona monkey (*Cercopithecus mona*), and the olive colobus (*Procolobus verus*). However, political instability in the Delta, related in the most part to disputes over the allocation of oil revenues, has prevented any progress in biodiversity conservation during the last decade. Because red colobus monkeys have been found to be highly vulnerable to habitat disturbance and hunting in other parts of Africa (Struhsaker 2005), it is feared that the Niger Delta red colobus is being driven to the edge of extinction.

The red colobus monkeys are probably more threatened than any other taxonomic group of primates in Africa (Oates 1996; Struhsaker 2005). Almost all those of western Africa are in a precarious position. *Piliocolobus badius temminckii* (Senegal to Guinea or Sierra Leone), *Piliocolobus badius badius* (Sierra Leone to western Côte d'Ivoire), *Piliocolobus waldroni* (eastern Côte d'Ivoire and western Ghana), *Piliocolobus preussi* (western Cameroon and eastern Nigeria), and *Piliocolobus pennantii* (Bioko Island, Equatorial Guinea) are all now considered to be Critically Endangered from

different combinations of habitat loss and hunting. It is probable that *P. waldroni* is now extirpated, although there has been some hope that a few individuals still survive in swamp forest in the far southeastern corner of Côte d'Ivoire and the southwestern corner of Ghana (McGraw and Oates 2002; McGraw 2005; Osei *et al.* 2015). *Piliocolobus bouvieri*, restricted to swamp forests in the Republic of Congo, and which has been considered a relative of *P. pennantii* and is very poorly known, is categorized as Endangered.

Because the security situation in the Niger Delta continues to be challenging, undertaking effective conservation actions remains difficult. Despite these challenges, and with the urgent need to help this species avoid extinction, a locally driven conservation advocacy initiative is underway, coordinated by R. Ikemeh and her team; much greater and more concerted efforts must be made to protect this species by law in Bayelsa State and to establish one or more protected areas for the fragile remaining populations.

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Grauer's Gorilla
Gorilla beringei graueri Matschie, 1914
Democratic Republic of Congo
(2010, 2012, 2014, 2016)

Stuart Nixon & Andrew J. Plumptre



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

The world's largest primate, Grauer's gorilla (*Gorilla beringei graueri*), is one of two subspecies of eastern gorilla and was recently classed as Critically Endangered on the IUCN Red List of Threatened Species (Plumptre *et al.* 2016a). Grauer's gorilla is endemic to the Democratic Republic of Congo (DRC) and distributed discontinuously throughout the lowland, mid-altitude and montane forests east of the Lualaba River to the western flanks of the Albertine Rift escarpment.

The first surveys of Grauer's gorillas were conducted in 1959 (Emlen and Schaller 1960). This landmark study observed that eastern gorillas were severely threatened by hunting and habitat destruction. Research on their ecology began in the 1970s in the highland sector of Kahuzi-Biega National Park (KBNP). Compared to mountain gorillas and western lowland gorillas, significantly less is known about Grauer's gorilla behaviour and ecology; however, their diet is known to be rich in herbs, leaves, bark, lianas, vines, seasonally-available fruit, bamboo (at higher altitudes) and insects, and they show a preference for regenerating vegetation (Schaller 1963; Yamagiwa *et al.* 2005; Nixon *et al.* 2006).

During the 1960s, through to the late 1980s, habitat conversion in the eastern part of their range destroyed almost all montane forest outside the protected areas and exterminated a number of important high-altitude gorilla populations. Widespread killing of gorillas for bushmeat or in retaliation for crop raiding is believed to have reduced populations across their entire range during this period (Mehlman 2008; Nixon *et al.* 2012). In the 1990s, efforts were made to determine the status of Grauer's gorilla in the Maiko National Park (MNP; Hart and Sikubwabo 1994), 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 were 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, with the Rwandan genocide of 1994 causing hundreds of thousands of refugees to flee into DRC, destabilising an already fragile region and leading to the First Congo War (1996–1997).

The Second Congo War began in 1998 and continued until 2003 with devastating consequences, including an estimated 3.8 million people dead, widespread insecurity, heightened illegal bushmeat trade and increased deforestation (Turner 2007). In eastern DRC, civil war resulted in the formation of many armed groups, including those born among local communities protecting their interests from other armed groups (such as Mai Mai militia), particularly over access to mining sites. Artisanal mining operations expanded rapidly in North and South Kivu provinces, with most mines controlled by armed militia or soldiers from the national army. Artisanal miners and militia often operate in remote forests, far from villages, and resort to hunting wildlife to feed themselves, targeting the larger species that provide more meat. Despite being legally protected, gorillas are highly prized as bushmeat because of their large size, and since they move in groups on the ground and can be tracked, they can be shot relatively easily. KBNP and MNP have been at the centre of this intense and illegal resource extraction for the past 20 years, which has severely impacted the capacity of these important parks to protect the gorillas.

Evidence from a limited number of surveys conducted across the Grauer's gorilla range between 2003 and 2010 indicated a severe population decline, with killing of gorillas recorded at all sites visited (Nixon *et al.* 2006; Amsini *et al.* 2008; Mehlman 2008; Nixon 2010). The KBNP highland population was almost halved between 1996 and 2000 (Spira *et al.* 2016) and several small populations first described by Emlen and Schaller (1960) in the Maiko region were extirpated (Nixon *et al.* 2012). Further evidence for severe declines came from an analysis of great ape habitat across Africa, which estimated that suitable environmental conditions for Grauer's gorillas had been halved since the 1990s (Junker *et al.* 2012).

A multi-stakeholder conservation action plan for Grauer's gorillas and eastern chimpanzees in eastern DRC was developed to address the conservation crisis facing this ape (Maldonado *et al.* 2012). This action plan details priority actions for Grauer's gorilla conservation, and 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. A principal recommendation of this plan was to conduct a range-

wide assessment of Grauer's gorilla status and threats. In a collaborative effort, Fauna & Flora International, the Wildlife Conservation Society, the Institut Congolais pour la Conservation de la Nature (ICCN) and local communities carried out surveys of this ape across its range between 2011 and 2015.

The results of this extensive effort show that Grauer's gorilla has suffered a catastrophic decline since the 1990s (Plumptre *et al.* 2016a, 2016b), due almost entirely to illegal hunting associated with artisanal mining activities. A recent study (Kirkby *et al.* 2015) investigating the impact of mining around KBNP and Itombwe revealed that many miners admit to having eaten gorillas. This finding is reflected by the survey results, which show that gorillas in the KBNP stronghold have declined by 87% since 1994. Gorilla nest encounter rates indicate declines of 82–100% at six of 10 additional sites. Only one population is currently increasing—that in the heavily-protected KBNP highland sector, which grew by 18% to 213 individuals between 2010 and 2015 (Spira *et al.* 2016). A spatial occupancy analysis of the survey data identified KBNP and the adjacent Kasese region (including the Reserve des Gorilles de Punia), together with the remote and previously undocumented Usala Forest, as critical sites for the remaining populations of this ape. An estimated 3,800 (95% CI 1,280–9,050) Grauer's gorillas now remain across their 19,700km² range, representing a 77% decline in a single generation (Plumptre *et al.* 2015, 2016b).

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. ICCN and partners made significant progress between 2014 and 2016, regaining control of parts of the KBNP lowland sector and increasing regular surveillance of the southern sector of MNP. Significant gains have also been made in other important regions, including the gazetting of the Itombwe Natural Reserve in 2016 to protect the core of the Itombwe Massif and its highly fragmented gorilla populations. Outside protected areas, regular community-based gorilla monitoring programmes have been developed in the Lubutu, Kasese and Walikale regions, and could be expanded to other remote regions outside the formally protected areas.

Despite these significant gains, ICCN faces continuing conflict with armed groups, and several highly dedicated ICCN personnel have been killed in the line

of duty while attempting to protect Grauer's gorillas and their habitat. Destruction of forest for timber, charcoal production and agriculture continues to threaten the isolated gorilla populations that persist in the North Kivu highlands and Itombwe Massif, while poaching presents a serious ongoing threat to Grauer's gorillas across their range. Since 2003, ICCN and partners have confiscated 15 Grauer's gorilla infants—casualties of poaching, now cared for by the Gorilla Rehabilitation and Conservation Education Centre (GRACE) near Lubero, North Kivu. The potential for reintroduction of these confiscated individuals to sites such as Mt. Tshiaberimu in Virunga National Park offers some hope for the future of small, isolated subpopulations in well-protected areas.

Conservation challenges are likely to increase in the coming decades 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 citizens 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 the further demise of this subspecies. The steady recovery of the KBNP highland population (Spira *et al.* 2016) is encouraging evidence that highly-targeted conservation efforts can succeed even in the face of acute and sustained human pressures.

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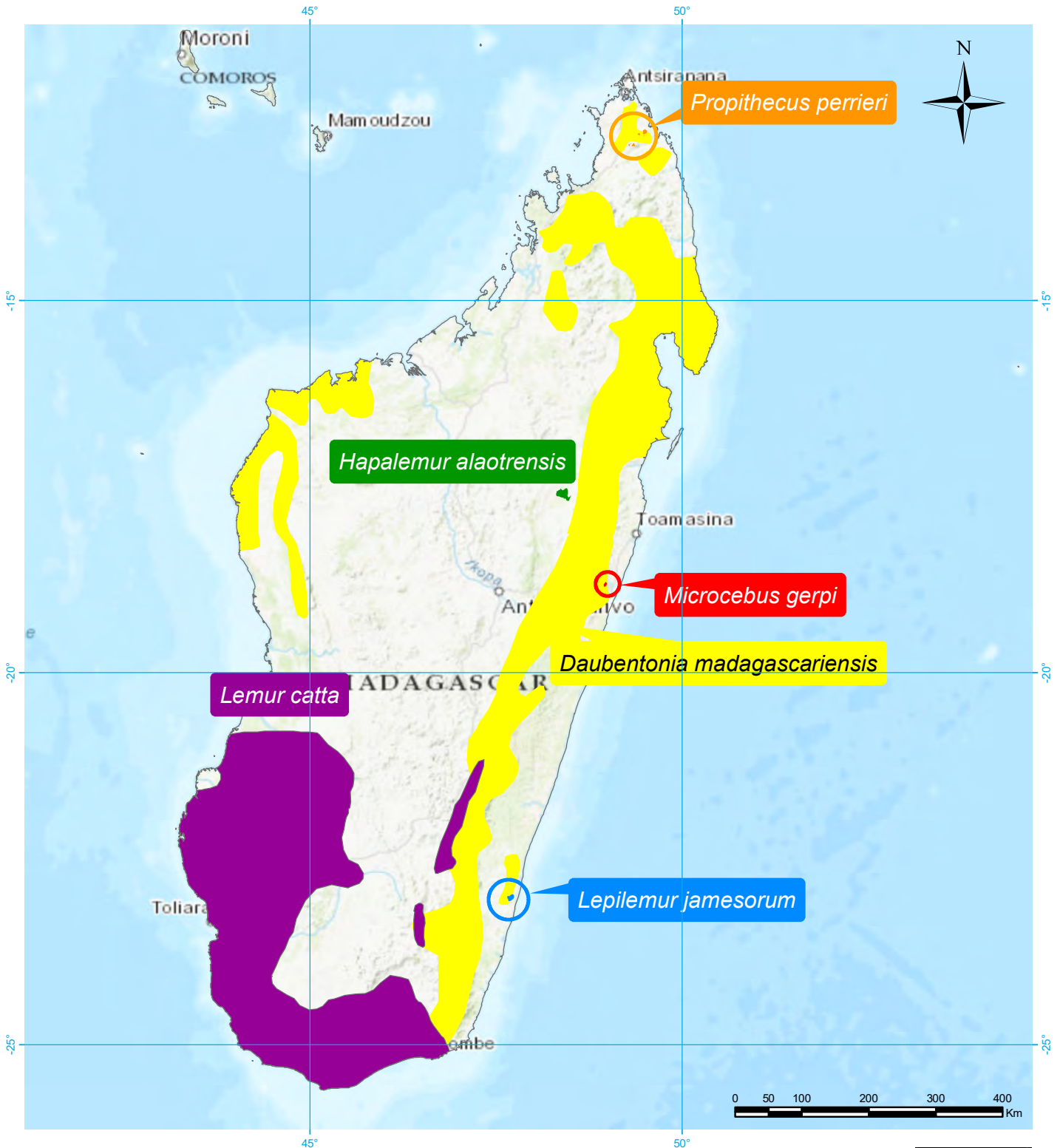
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Madagascar



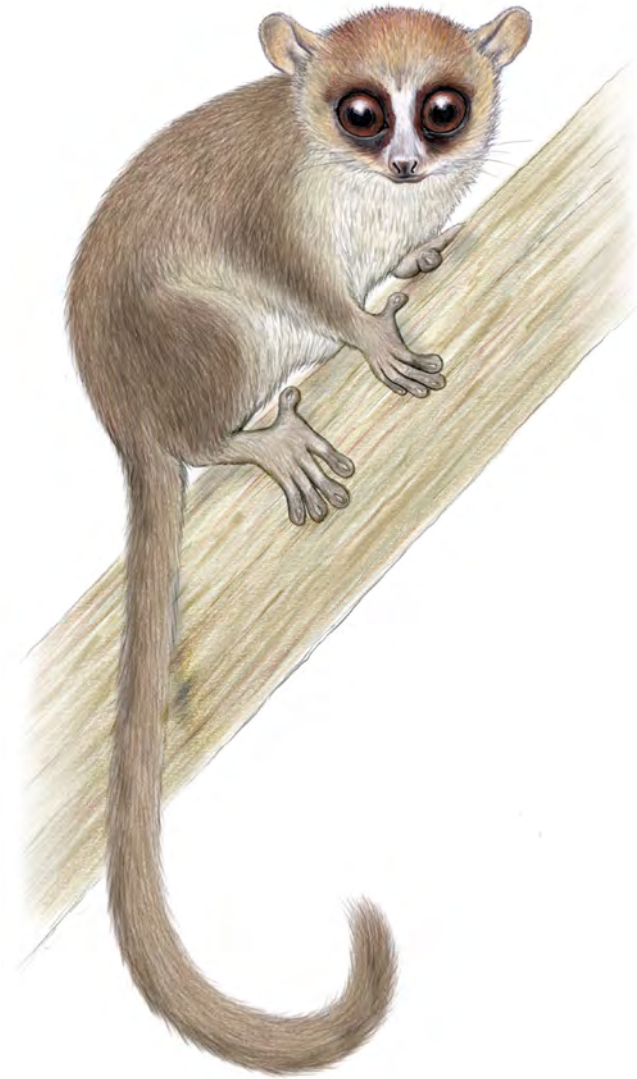
Madagascar Primates

- | | |
|---|---|
|  <i>Daubentonia madagascariensis</i> |  <i>Lemur catta</i> |
|  <i>Hapalemur alaotrensis</i> |  <i>Microcebus gerpi</i> |
|  <i>Lepilemur jamesorum</i> |  <i>Propithecus perrieri</i> |



Gerp's Mouse Lemur
Microcebus gerpi Radespiel *et al.*, 2011
Madagascar
(2016)

Ute Radespiel & Jonah H. Ratsimbazafy



Gerp's Mouse Lemur (*Microcebus gerpi*)
(Illustration: Stephen D. Nash)

Microcebus gerpi is one of the larger mouse lemur species (>50g average adult body mass), with an average head length of 36.4 ± 1.28 mm, an average body length of 84.14 ± 7.17 mm, and an average adult body mass of 68.14 ± 15.04 g (Radespiel *et al.* 2012). It differs morphologically from the larger western mouse lemurs by its small ears, typical for rainforest mouse lemurs, and from its immediate geographic neighbor, *M. lehilahytsara*, by its larger size. It has a brownish-gray body and a rufous head. The outer arms and legs are darker brown than the rest of the body. The head is darker brown around the eyes and it has a distinct white stripe between the eyes. The ears are small but eminent and dark brown towards the edges. The tail is densely furred, long and of brownish-gray color, and can store body fat (Radespiel *et al.* 2012). Gerp's mouse lemurs are nocturnal solitary foragers, but further data on their life history, ecology, and behavior are not available. A total of 14 individuals were caught, measured, sexed, photographed and released in the forest of Sahafina (29–231 m a.s.l.) and in the adjacent savoka, which is composed of transitional secondary vegetation that grows after the forest has been cut down for the cultivation of rice. The presence of seven young animals in the captured population of December 2008 confirmed successful reproduction there.

So far, *M. gerpi* has been confirmed only from this one fragment of lowland evergreen rainforest that is situated about 58 km east of the Mantadia National Park and about 87 km south of Toamasina. Eastern Madagascar shows very strong topographic heterogeneity. A narrow strip (<50 km) of coastal lowland habitat is delimited to the east by the Indian Ocean and to the west by a central mountain range that splits the island of Madagascar into a humid east and a rather dry west. Multiple large rivers form efficient physical barriers to gene flow and further subdivide the western and the eastern domain into a larger number of so-called inter-river systems. These have been shown to act as centers of endemism for a variety of lemurs, including mouse lemurs and other

Gerp's mouse lemur, *Microcebus gerpi*, was recently discovered in the Sahafina Forest in eastern Madagascar during two field trips in December 2008 and June 2009. The analysis of mitochondrial sequence data and a comparison of thirteen external standard morphometric measurements with values found in the literature for other mouse lemur species allowed the species to be formally described in 2012 (Radespiel *et al.* 2012).

species with limited dispersal capacities (Wilmé *et al.* 2006; Louis *et al.* 2006; Olivieri *et al.* 2007; Weisrock *et al.* 2010; Rasoloarison *et al.* 2013; Hotaling *et al.* 2016). In the case of *M. gerpi*, the northern and southern limits of its distribution are not yet known. The highland area to the west, however, is known to contain another mouse lemur, the highland specialist *M. lehilahytsara* (Kappeler *et al.* 2005; Weisrock *et al.* 2010; Radespiel *et al.* 2012).

Microcebus gerpi was assessed as Critically Endangered at an IUCN Red List assessment workshop held in Antananarivo in May 2012. This classification is justified by the combination of (1) its presumably very limited range, which could be as small as about 7,600 km² (Radespiel *et al.* 2012), (2) the very low estimated area of occupancy and extent of occurrence (Andriaholinirina, N. *et al.* 2014), (3) the high anthropogenic pressures in the lowland areas of Madagascar (Harper *et al.* 2007), and (4) the fact that there is no protected zone within its known range. In addition to the immediate threat of forest loss for the survival of this mouse lemur, mouse lemurs are also hunted opportunistically (Golden 2009; Borgerson 2016). This alarming situation calls for immediate conservation action.

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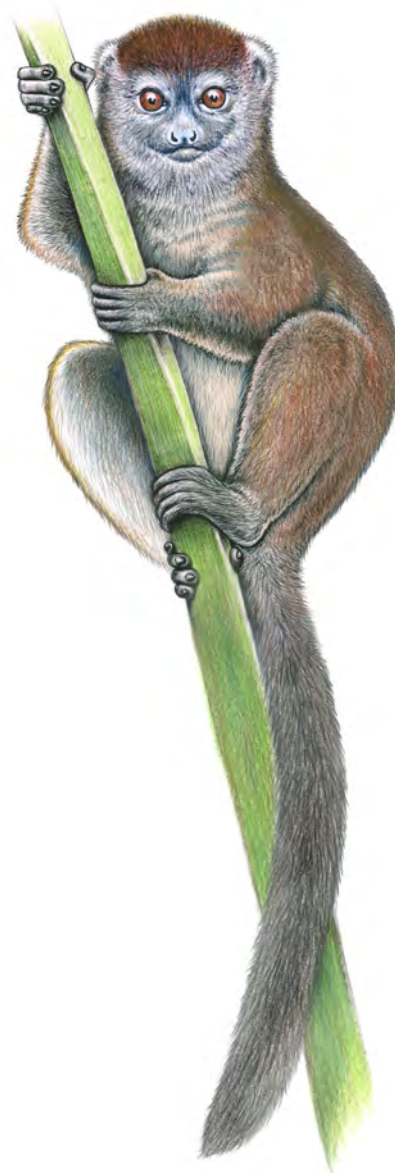
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Lac Alaotra Gentle Lemur
Hapalemur alaotrensis Rumpler, 1975
Madagascar
(2000, 2014, 2016)

Lena M. Reibelt, Jonah Ratsimbazafy & Patrick O. Waeber

The Lac Alaotra gentle lemur, *Hapalemur alaotrensis*, has been recognized as a distinct species – elevated from the subspecies *H. griseus alaotrensis* – since 2001 (Groves 2001). It is also known as the Lac Alaotra bamboo lemur. While all other bamboo lemurs are forest dwellers occupying a variety of forest habitats across Madagascar, *H. alaotrensis* is confined to the marshlands surrounding Lake Alaotra. The species has an average adult body length of about 30 cm (12 in), with a tail 30–40 cm (12–16 in) in length, a dark-grey coat on the face, ears and chest, and with a chestnut to golden-brown tinge on the crown and nape, back and tail. The head is rounded; the muzzle blunt and the ears are short and round. The grasping hands and feet and long tail are used for balance, and allow this wetland-adapted lemur to walk along the reed and cyperus stalks of its marshland habitat (Waeber *et al.* 2017b).

The Lac Alaotra gentle lemur is a cathemeral and highly specialized herbivore. It targets 11 plant species and 16 distinct plant parts, including shoots, stems, pith, leaves, seeds, buds, and flowers (Mutschler 1999). More than 95% of its feeding time is spent on *Cyperus madagascariensis* (Cyperaceae), and three Poaceae *Phragmites communis*, *Leersia hexandra*, and *Echinochloa crusgalli*, which are available year-round. The crepuscular lemur shows two distinct activity peaks, one in the early morning, the other in the late afternoon (Mutschler *et al.* 1998). The social groups of *Hapalemur alaotrensis* typically consist of two to nine individuals (Nievergelt *et al.* 2002), while groups of more than 13 have also been reported (Waeber and Hemelrijk 2003). Home ranges of *H. alaotrensis* vary from two to five hectares depending on group size (Nievergelt *et al.* 1998). Females are dominant in *H. alaotrensis*, leading group travel except for water channel crossings (Waeber and Hemelrijk 2003). The gestation period is similar for all bamboo lemurs, being between 137 and 149 days. For *H. alaotrensis*, the birth season starts in September and ends in February (Mutschler 1999), with births not being synchronized within groups (Nievergelt *et al.* 2002).



The Lac Alaotra Gentle Lemur (*Hapalemur alaotrensis*)
(Illustration: Stephen D. Nash)

Lake Alaotra, in the central highlands in the Alaotra-Mangoro region, is the biggest shallow fresh-water lake in Madagascar. It covers 20,000 ha of open water, and is surrounded by some 20,000 ha of marshes and 120,000 ha of rice-fields. Due to the high importance of the wetland for biodiversity and agro-economy, it was inscribed as a Ramsar site in 2003 and designated a temporary New Protected Area (NPA) in 2007; it was

granted permanent protection status in July 2015. The principal threats to *Haplemur alaotrensis* are habitat loss, habitat degradation, and hunting. Fisheries and rice production are the primary economic drivers in the region; marshes are burned mainly to establish irrigated rice fields and access to fishing ponds (Copsey *et al.* 2009a, 2009b; Guillera-Arroita *et al.* 2010; Ralainasolo *et al.* 2006). While the human population around the lake increased fivefold in the last fifty years to over 550,000 people, rice harvests and fish output are declining. In search of humid and arable land, rice paddies are established during the dry season within the boundaries of the NPA; these production areas within the marsh and shallow lake are growing from year to year.

The Lac Alaotra gentle lemur is the only primate living constantly in a wetland; it is classified by the IUCN as Critically Endangered based on its limited geographical range and the increasing pressures on its remaining lakeside habitat. In the 1990s, the total population was estimated at 7,500–11,000 individuals (Mutschler and Feistner 1995); in 2005, it had shrunk to an estimated 2,500 individuals (Ralainasolo *et al.* 2006). The lemur's habitat is threatened by annual fires, and in one year more than 40% of its remaining marshland was destroyed (Ratsimbazafy *et al.* 2013). The highest density of *H. alaotrensis* is sheltered in the high priority conservation zone and tourist focal area, Park Bandro, which is the highest conservation category in the Alaotra NPA. At the eastern lake shores, adjacent to Andreba Gare, the park officially covers 85 hectares. In 2013, Ratsimbazafy and colleagues estimated a subpopulation of more than 170 individuals in the isolated park. A recent reassessment of park boundaries revealed that only half of the original park remains and that the current carrying capacity of Park Bandro is just 40–80 individuals (Raveloarimalala and Reibelt 2016). Further subpopulations are found in the southwestern marshes of Lake Alaotra; subpopulations in the northern parts of Lake Alaotra have likely disappeared. Given the continuous marshland burning and ongoing poaching since 2006, total population numbers of *H. alaotrensis* are now assumed to be much lower than the last census estimates.

Conservation and management of the Alaotra NPA is challenging due to very limited technical and financial resources. There are four permanent technical agents from the ministries of Water and Forests responsible for more than 50,000 ha of marshes (also outside the NPA). In close partnership with the authorities, Durrell

Wildlife Conservation Trust and Madagascar Wildlife Conservation have been collaborating for more than 10 years on community-based efforts around the lake, focusing on ecological monitoring, environmental education, and eco-tourism (Rendigs *et al.* 2015; Waeber *et al.* 2017c).

Given the alarming rates of habitat destruction and the related collapse of lemur population numbers, habitat restoration and the reconnection of isolated subpopulations will be priority conservation actions for the years to come. Waeber *et al.* (2017a) showed that people around the lake appreciate clear zoning for conservation and livelihood activities, and are willing to respect park boundaries as long as their livelihoods are ensured. Future restoration efforts will be paralleled by increased social marketing and educational conservation actions, in order to ensure a long-term impact of the conservation efforts for *H. alaotrensis*.

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Ring-Tailed Lemur

Lemur catta Linnaeus, 1758
Madagascar
(2016)

Marni LaFleur, Tara A. Clarke, Jonah Ratzimbazafy & Kim Reuter



The Ring-Tailed Lemur (*Lemur catta*)
(Illustration: Stephen D. Nash)

The ring-tailed lemur (*Lemur catta*) is an iconic primate and a flagship species for the island of Madagascar. They are one of the most recognizable of all lemurs, due in part to their long conspicuous black and white striped tails, as well as their vast global prevalence in zoos and popular culture. Ring-tailed lemurs have been continuously studied for decades at a select few field sites in Madagascar (see Jolly 2012), yet new and remarkable findings continue with recent studies (i.e. cave use: Sauther *et al.* 2013; cathemerality: Donati *et al.* 2013; LaFleur *et al.* 2014).

Ring-tailed lemurs are strictly female dominant (Jolly 1984), live in social groups of up to 30 animals, and inhabit a plethora of habitat types throughout southern Madagascar (Goodman *et al.* 2006). They are a remarkably flexible 'edge' or 'weed' species and are able to adapt to ecological change (Sussmann 1977). Despite being behaviorally and ecologically plastic, ring-tailed lemurs are Endangered (Andriaholinirina *et al.* 2014), and their wild populations are experiencing rapid and alarming rates of decline (Gould and Sauther 2016; LaFleur *et al.* 2016). Habitat loss (Sussman *et al.* 2006; Brinkman *et al.* 2014), bushmeat hunting (Gardner and Davis 2014), and most recently, live capture for the domestic illegal pet trade (Reuter *et al.* 2016), have decimated wild populations and leave many perilously close to localized extinction (LaFleur *et al.* 2016). Moreover, the effects of genetic isolation (Clarke *et al.* 2015), continued habitat fragmentation (Waeber *et al.* 2015), and further habitat loss due to climate change (i.e. 63% before 2080; Brown and Yoder 2015) are likely to have long-term deleterious impacts on this charismatic species.

Madagascar's dry forests, where ring-tailed lemurs naturally occur (Goodman *et al.* 2006), have been reduced by 45% over the past 40 years (Brinkman *et al.* 2014). Deforestation for small-scale but widespread charcoal production, slash-and-burn agriculture, and livestock grazing, all impact remaining forests throughout southern Madagascar (Waeber *et al.* 2015). Though ring-tailed lemurs can persist in very degraded landscapes (see Kelley 2013; Gould and Gabriel 2015), it is not likely that they can survive for long in completely deforested areas. Furthermore, a recent examination of this species' historical range (Goodman *et al.* 2006) indicates a possible significant reduction (LaFleur *et al.* 2016). This suggests that ring-tailed lemur populations are shrinking along with their forest habitats, while also disappearing from certain forested areas, likely due to hunting and live capture for the pet trade.

Although hunting of lemurs is taboo in some areas of southern Madagascar (Loudon *et al.* 2006), hunting of ring-tailed lemurs may have led to localized extirpation in several forested areas north of Toliara (see Gardner and Davies 2014; LaFleur *et al.* 2015). Moreover, there are several additional areas where hunting of ring-tailed lemurs occurs (see Goodman 2003; Moniac and Heitmann 2007; Seirs 2007), and consequently where there are now very few or no animals remaining (LaFleur *et al.* 2016). Hunters often use dogs to track, chase and kill entire social groups of ring-tailed lemurs (see Gardner and Davies 2014), which does not appear to be sustainable.

In Madagascar, ring-tailed lemurs are frequently captured live from protected forest areas and kept as illegal pets (Reuter *et al.* 2016; Reuter and Schaefer 2016; LaFleur *et al.* 2016). These lemurs are primarily used to generate income from tourists, either through for-pay photo opportunities, or by way of increasing hotel/restaurant businesses (Reuter and Schaefer 2016). Ring-tailed lemurs used as photo props can generate as much as \$3 USD per interaction (LaFleur *et al.* 2015), while hotels that show lemurs on their websites charge higher prices and thus average \$25 USD more per night (Reuter and Schaefer 2016). Sadly, infant ring-tailed lemurs appear to be targeted for the pet trade, and likely suffer high mortality rates due to dehydration and starvation in the days and weeks following wild capture (LaFleur *et al.* 2016). Again, the pressure experienced by ring-tailed lemurs as a result of the domestic illegal pet trade, is likely to be unsustainable for the species and, besides, results in poor welfare in captured individuals.

Several urgent interventions are needed to ensure the survival of wild ring-tailed lemur populations. They include targeting the drivers of ring-tailed lemur extirpation, including: 1) habitat loss, 2) bushmeat hunting, and 3) live-capture for the illegal pet trade. Ultimately, these drivers can be tied to the extreme poverty found throughout Madagascar, particularly southern regions of the country where childhood growth stunting and malnutrition are prevalent (see Waeber *et al.* 2015). Improving general economic and food-specific security for individuals, along with programs increasing awareness and pride in local biodiversity are critically needed. Additionally, tourists need information regarding how their choices may impact endangered species, such as ring-tailed lemurs, and how to be a “lemur ally” when traveling.

The trend is clear: ring-tailed lemur habitats and populations are declining at unsustainable rates. If we don't act to protect this species swiftly, they are certain to become extinct in all but a select few locations in the wild. Ring-tailed lemurs are important within the habitats where they persist, as they are seed dispersers, and an emblematic species for Madagascar. Immediate funding to support community development, habitat preservation, and animal monitoring are needed to ensure the long-term survival of ring-tailed lemurs in Madagascar's unique southern forests.

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James' Sportive Lemur
Lepilemur jamesorum Louis Jr *et al.*, 2006
Madagascar
(2016)

Tsarafilamatra R. Andriamisedra, Jonah Ratsimbazafy, William Dreyer & Edward E. Louis, Jr

James' sportive lemur is a medium-sized nocturnal sportive lemur of Madagascar. It exists only in Manombo Special Reserve and Vevembe Classified Forest in the southeastern coastal region of the island, and is listed as Critically Endangered on the IUCN Red List (Andriaholinirina *et al.* 2014). James' sportive lemur is similar to other sportive lemurs in that it is folivorous (Ganzhorn 1993); but it supplements its diet by feeding on gums. The species weighs approximately 780 g, with a head-body length of 26 cm, and a tail length of 30 cm (total length of 56 cm). The pelage is short and smooth, primarily brown on the body and light grayish-brown on the belly and ventral portion of the extremities. The face is distinguished by the whitish-gray marking along the jaw and throat from the chin to the ears, forming a mask. The upper part of the head is brown with a black midline stripe that is continuous for almost the entire length of the body. The ears are large and cup-shaped, gray dorsally, with black borders and a small cream-colored patch on the region beneath (Louis *et al.* 2006; Mittermeier *et al.* 2010). In general, the tail is uniformly brown, but several individuals have been noted to have a whitish tip.

Lepilemur jamesorum is known largely from the Manombo Special Reserve, although it has been documented through molecular genetic data to exist in the Vevembe Classified Forest inland near Vondrozo. Although James' sportive lemur was initially described in 2006, there has been limited field research conducted on it before or since. Recent surveys in Manombo by the authors indicated very low densities, with only two individuals occupying an 800-ha survey plot within the special reserve. There is a limited availability of tree holes, and infrequent feeding traces have been found during recent surveys at Manombo.

Unfortunately, the distribution of James' sportive lemur in coastal littoral forests in southeastern Madagascar makes it especially vulnerable to stochastic events such as cyclones such as that of 1997. Anthropogenic

pressure is a significant problem in the Manombo area due to high levels of poverty, limited job opportunities, and inflation, which drive local communities to use the protected area to survive. Consequently, the primary threats for James' sportive lemur are significant habitat loss and hunting for bush meat by using traditional traps.



James' Sportive Lemur (*Lepilemur jamesorum*)
(Illustration: Stephen D. Nash)

Local hunters also catch sportive lemurs by taking them from tree holes after cutting into the tree. Deforestation is a significant problem in this region as people use the forest as their only resource for timber, firewood and charcoal production. These problems directly affect food availability for James' sportive lemur and also lead to a fragmented habitat.

The exact distribution and population numbers for James' sportive lemur are unknown. Its survival is dependent on initiating field surveys and research studies to ascertain its status and to establish known population parameters for this species. Concomitantly, reforestation efforts are indispensable to restore habitat loss, reconnect forest fragments, and provide alternative resources at the community level outside of the protected areas. Engaging the local communities in alternative livelihoods is also vital for sustainable practices and conservation in the Manombo region, as is the reinforcement of environmental education in the school system and local surrounding communities.

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Perrier's Sifaka
Propithecus perrieri Lavauden, 1931
Madagascar
(2000, 2002, 2004, 2014, 2016)

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



Perrier's Sifaka (*Propithecus perrieri*)
(Illustrations: Stephen D. Nash)

Perrier's sifaka (*Propithecus perrieri*) is a lemur of intermediate size relative to other members of the genus *Propithecus* (Ranaivoroisoa *et al.* 2006; Lehman *et al.* 2005). It is characterized by an all-black pelage, naked black face and striking orange-red eyes (Mittermeier *et al.* 2010). Perrier's sifaka occurs in dry deciduous forests on limestone karst and semi-evergreen transitional forests on sandstone soils. A preliminary dietary study found that it feeds predominantly on leaves, flowers and fruit (Lehman and Mayor 2004). Its current distribution is the smallest of all *Propithecus* species and *P. perrieri* has been recognized as Critically Endangered since 1996 (Mittermeier *et al.* 2010; Banks 2012; Salmona *et al.* 2013a; Andriaholinirina *et al.* 2014). Its geographic range is restricted to the extreme northeast of Madagascar, some 50 km south of Antsiranana (Diego Suarez). It extends from the

Analamerana Massif to the Irodo River and is bounded in the south by the Andrafiamaena mountain chain (Banks 2012; Zaonarivelo *et al.* 2007). Despite evidence of the species' presence in the Ankarana National Park in the 1980s and 1990s (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.* 2013a) failed to find Perrier's sifakas there. Furthermore, suggestions that its distribution might also extend south of the Andrafiamaena mountains and into the Andavakoera Forest (Schwitzer *et al.* 2006) could not be confirmed during two surveys in the area in 2006 and 2012 (Zaonarivelo *et al.* 2007; Salmona *et al.* 2013b). Also, *P. perrieri* is not being kept in captivity anywhere in the world (Andriaholinirina *et al.* 2014).

Earlier estimates of Perrier's sifaka population size suggested that less than 1,000 individuals persist in the wild (Banks *et al.* 2007). Estimates of the effective population size (N_e) from field data (~230 individuals; Banks *et al.* 2007) and from genetic data (N_e ~50–100; Salmona *et al.* 2015) further support that the population is small. However, extended survey efforts (across 85% of the total remaining habitat) by M. Banks between 2007 and 2012, and considering behaviour, observer,

and habitat type, showed a population size estimate of ~2,100 individuals. Sandstone forests, despite representing only ~12% of the habitat, likely host nearly 40% of Perrier's sifaka populations with densities up to an order of magnitude higher than in other forest types (Banks 2013).

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 larger distances of open habitat (Meyers and Wright 1993; Richard *et al.* 1993). Population level studies and occupancy patterns (Banks 2013) as well as genetic data (Salmona *et al.* 2015) indicate that the population is either not strongly influenced by the fragmentation of forests and the matrix of open grassland habitats or these effects have not yet been detected (i.e. no clear barriers could be identified by these authors). Surprisingly, Mayor *et al.* (2002), Salmona *et al.* (2015) and Bailey *et al.* (2016), identified relatively high levels of genetic diversity compared to other sifakas. Even though Perrier's sifaka may have the ability to cross open grassland, most sifakas encountered were elusive and fled from humans (Salmona *et al.* 2015). Studies of occupancy patterns showed 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.* 2015). With >2,200 individuals left (Banks 2013), not all of which are reproductive, and a long generation time of 10–20 years, the viability of the population is at a high risk of being compromised. Generation time estimates are tentative and partly based on data from Verreaux's sifaka, *P. verreauxi* (Lawler *et al.* 2009; Morris *et al.* 2011; Salmona *et al.* 2015; see Salmona *et al.* 2017 for details).

Decades ago, several authors reported the presence of Perrier's sifaka outside of its current distribution (Hawkins *et al.* 1990; Meyers 1996). Moreover, subfossils of *P. cf. diadema* (possibly *P. perrieri*) were reported in an Ankarana cave (Jungers *et al.* 1995; Godfrey *et al.* 1999) and far north from its current distribution in Montagne des Français, Andavakoera cave (Dewar *et al.* 2013). 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.* 2017) similar to the decline detected in the neighbouring golden-crowned sifaka (Quéméré *et al.* 2012; Salmona *et al.* 2017). 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 (Salmona *et al.* 2017). Indeed, the population contraction datings overlap with the first documented human presence in the region ~4,000 years ago (Dewar *et al.* 2013), and also with major dry periods such as the mid-Holocene Boundary (Salmona *et al.* 2017). Today, Perrier's sifaka is still strongly protected by a local Antankarana taboo (*fady*) which prohibits poaching and consumption and is shared by more than 95% of local residents (Anania *et al.* submitted). This suggests that the sifaka population may not have been under heavy direct human pressure. The recent range decrease – no observations in Ankarana NP since the 1990s – may be the result of indirect effects such as selective logging, deforestation, and loss of connectivity. While Landsat 7 imagery between 1994 and 2003 suggests a decline of sandstone forest cover of >60% over this period, loss of connectivity has been exacerbated by increasing traffic on the national road crossing the park, likely isolating part of the sifaka population.

Given the small total population size, persistence of local threats and the paucity of wildlife patrols, an appraisal of Perrier's sifaka population levels and an effective control of habitat loss are urgently needed. This requires a unified regional management plan, since the species' natural range and potential areas of migration/seasonal presence overlap with three protected areas of different protective status, independently managed by Madagascar National Parks (Analamerana and Ankarana) and Fanamby (Andrafiarana). Given the diverse group of stakeholders involved (e.g., park services, ministries, universities, tour operators, local businesses, farmers), *P. perrieri* conservation requires a clearly defined institution, committed to leading its conservation plan with incentives for inclusive action that take advantage of the strengths of the different participants.

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Aye-Aye
Daubentonia madagascariensis (Gmelin, 1788)
Madagascar
(2016)

*Doménico R. Randimbiharirinirina, Timothy M. Sefczek, Brigitte M. Raharivololona, Yves Rostant Andriamalala,
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The Aye-Aye (*Daubentonia madagascariensis*)
(Illustration: Stephen D. Nash)

The aye-aye, *Daubentonia madagascariensis*, is the only surviving representative of the Daubentoniidae, the oldest family of the living lemurs (Martin 1990; Simons 1995; Catlett *et al.* 2010). Aye-ayes have the widest distribution of any extant lemur, ranging from Montagne d'Ambre in the north, to Parc National d'Andohahela in the south, and Parc National Tsingy de Bemaraha in the west of Madagascar (Ganzhorn and Rabesoa 1986; Simons 1993; Schmid and Smolker 1998; Rahajanirina and Dollar 2004).

The aye-aye has a body mass of 2.5–2.6 kg, a head-body length of 30–37 cm and a tail length of 44–53 cm (total length 74–90 cm) (Glander 1994). As such, aye-ayes are the largest nocturnal primate and the largest

of the solitary lemurs (Oxnard 1981; Feistner and Sterling 1995). Aye-ayes have several unusual, derived traits. They include an elongated, thin, highly-flexible middle finger, continuously-growing incisors, the greatest encephalization quotient of any strepsirrhine or nocturnal primate, and a relatively slow life history that includes late weaning and a protracted learning period (Owen 1863; Jouffroy 1975; Martin 1990; Simons 1995; Barrickman and Lin 2010; Catlett *et al.* 2010).

Aye-ayes live in several types of forest, from primary rainforest to dry forest, and survive in habitats of varying qualities, including continuous forests, disturbed forests, and heavily degraded forests near plantations (Pollock

et al. 1985; Ganzhorn and Rabesoa 1986; Ancrenaz *et al.* 1994; Andriamasimanana 1994). Two populations of aye-aye were introduced on Nosy Mangabe, an island in the north east, and on the peninsula of Maroantsetra, respectively (Petter 1977; Sterling 1993). Part of their success across various habitats is due to their highly adaptable diet. Aye-ayes eat mainly insect larvae and *Canarium* seeds, but can expand their diet to include nectar from *Ravenala madagascariensis*, coconut, banana, mango, litchi, breadfruit, sugar cane and cankers (Petter and Petter 1967; Petter 1977; Iwano and Iwakawa 1988; Sterling 1993; Andriamasimanana 1994; Simons and Meyers 2001).

Despite the aye-aye's distribution and dietary flexibility, they are more susceptible to anthropogenic disturbance than other lemur species due to their huge individual home ranges and low population densities (Perry *et al.* 2012). As aye-ayes are solitary with home ranges from 20 to 200 ha (Sterling 1993), reliable population estimates remain elusive as most locality records are based on feeding traces only. Whether those traces are from one individual or more is unknown. The greatest threats to aye-ayes are the destruction of their habitat (forest degradation and fragmentation, slash-and-burn agriculture) and local communities that believe that the aye-aye is a harbinger of death, and kill them whenever they are seen near to their villages (Petter and Peyrieras 1970; Simons and Meyers 2001). Although many scientific and popular articles have been written on the aye-aye, Sterling's (1993) research on Nosy Mangabe is the only long-term field study conducted on multiple individuals. Further studies are needed to understand the aye-ayes' use of habitats across multiple sites, before accurate population estimates and conservation initiatives can be achieved for this species.

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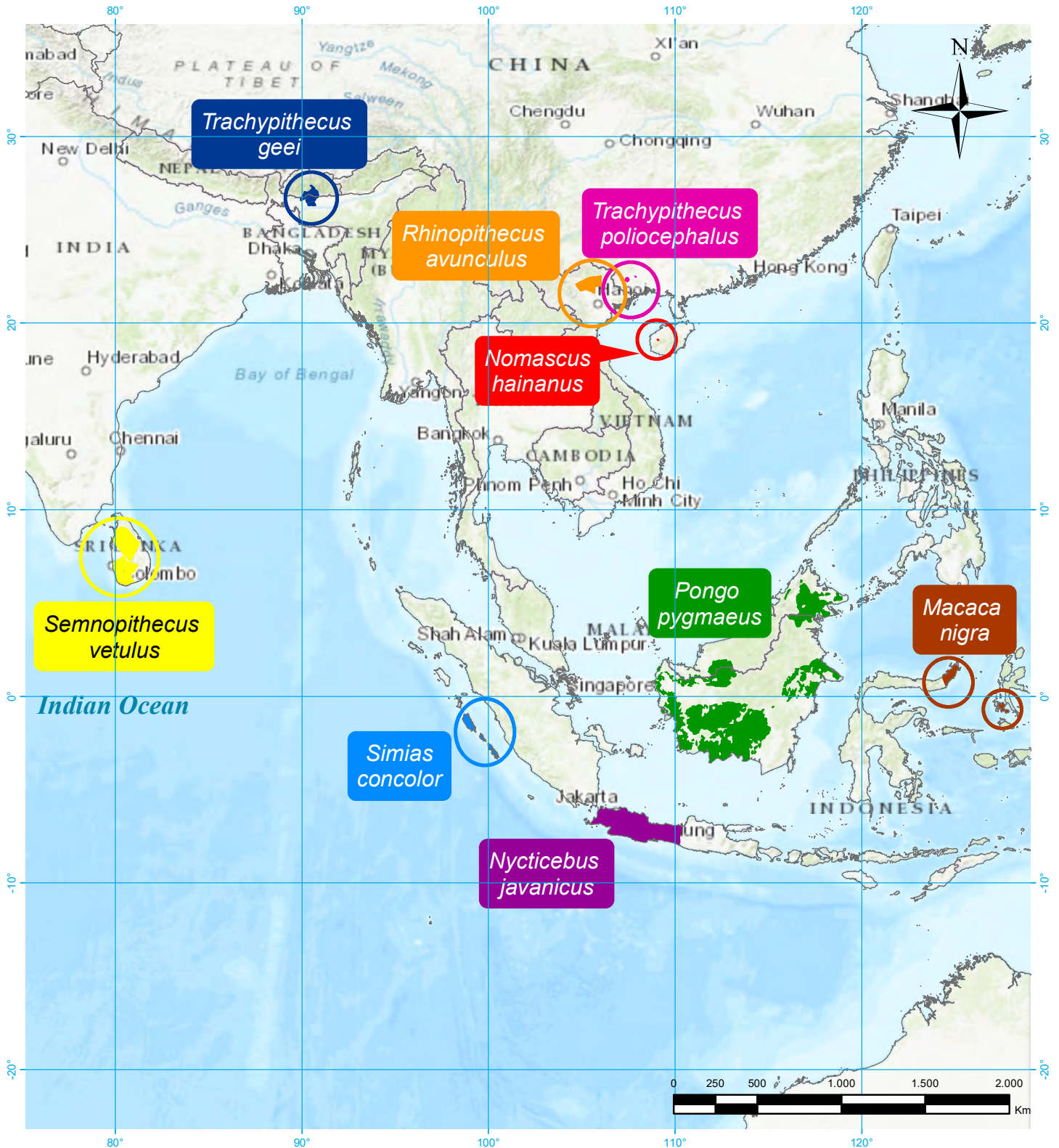
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Asia



Asian Primates

- | | | |
|---|--|---|
|  <i>Macaca nigra</i> |  <i>Rhinopithecus avunculus</i> |  <i>Trachypithecus poliocephalus</i> |
|  <i>Nomascus hainanus</i> |  <i>Semnopithecus vetulus</i> | |
|  <i>Nycticebus javanicus</i> |  <i>Simias concolor</i> | |
|  <i>Pongo pygmaeus</i> |  <i>Trachypithecus geei</i> | |



Javan Slow Loris
Nycticebus javanicus É. Geoffroy Saint-Hilaire, 1812
Indonesia
(2008, 2012, 2014, 2016)

K. Anna I. Nekaris & V. Nijman

Devastating habitat loss throughout southeast Asia threatens all nine species of slow loris with extinction, i.e. greater *N. coucang*, pygmy *N. pygmaeus*, Bengal *N. bengalensis*, Philippine *N. menagensis*, Bornean *N. borneanus*, Kayan *N. kayan*, Sody's *N. bancanus*, Sumatran *N. hilleri*, and Javan slow loris *N. javanicus* (Munds *et al.* 2013; Pozzi *et al.* 2014; Rowe and Meyers 2016). Slow lorises exhibit numerous unique traits, including slow life history, locomotion and digestion; ability to enter torpor and hibernate; and being the only venomous primates (Nekaris 2014). Still, wild slow lorises have only been studied for a year or more a handful of times (Malaysia *N. coucang*, Wiens *et al.* 2006; Cambodia *N. pygmaeus*, Starr *et al.* 2011; India, *N. bengalensis*, Das *et al.* 2014) with only *N. javanicus* the focus of long-term study (Rode-Margono *et al.* 2014). Many researchers and conservationists have only ever seen a slow loris in the illegal wildlife trade, either dried on bamboo sticks in preparation for traditional medicine, paraded as a photo prop on a tourist beach, or sold as a pet (Schulze and Groves 2004; Das *et al.* 2009; Nijman *et al.* 2015; Osterberg and Nekaris 2015). The extreme popularity of viral slow loris internet videos is a double-edged sword, to some extent making the public aware of their decline, but also causing the public to perceive that they are not threatened (Nekaris *et al.* 2013a). The extent of trade raised international concern, resulting in the transfer of the genus *Nycticebus* to CITES Appendix I in 2007 (Nekaris and Nijman 2007).

Sody's and Javan slow loris are now listed by the IUCN as Critically Endangered. No records of a living Sody's slow loris in the wild are available since a collecting trip by Dutch agronomist and taxonomist H. J. V. Sody in the 1930s on the island of Bangka, making it arguably the most threatened of the known slow loris species. Here we use the Critically Endangered Javan slow loris as the flagship for slow loris conservation (Nekaris *et al.* 2013b). Since being re-recognised as a species by the IUCN in 2006, work on the Javan slow



The Javan Slow Loris (*Nycticebus javanicus*)
(Illustration: Stephen D. Nash)

loris has increased and provides a sound example of understanding and mitigating the threats to a highly-threatened species. Both morphologically and genetically distinct, it weighs about 1 kg, and exhibits a 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).

Collecting individuals to meet the demand for pets is the most severe threat to the survival of Javan slow lorises. Despite being legally protected in Indonesia

since 1973, with its striking coloration and availability on Java, Indonesia's commercial centre, it is no wonder that Indonesian pet traders in the 1990s targeted Javan slow lorises above other endemic slow loris species. Since 2012, the numbers of Javan slow lorises in open trade in markets have decreased, with a stark rise in numbers of greater slow lorises from Sumatra, a species whose threat status must also be carefully monitored. Indeed, over three years of market surveys on Java between 2012 and 2015, four times more greater slow lorises than Javan slow lorises were counted, with traders claiming that Javan slow lorises are increasingly difficult to obtain (Nijman *et al.* 2015). In November 2013, nearly 300 greater slow lorises were confiscated in two raids. The smaller raid, where 76 slow lorises were confiscated, led to the death of 31 individuals in the weeks following; death rates of the larger raid are unknown. Successful prosecution of lawbreakers buying or selling slow lorises in Indonesia is a very rare occurrence, indeed so rare that we are not aware of a single slow loris trader having been sentenced in the last decade, despite hundreds of slow lorises having been confiscated from traders. Wildlife traders in Indonesia increasingly have turned to social media to advertise their illegal stock, including Javan slow lorises. The huge rise in Facebook and WhatsApp means that many are sold via social media without ever being seen in a wildlife market. An ongoing online monitoring programme by the Little Fireface Project suggests that in 2016 and 2017 an average of 25 Javan slow lorises are offered per month in online forums.

Javan slow lorises, both ones that are sold in the wildlife markets and the ones that are sold online often have their teeth removed. To avoid being bitten by venomous slow lorises, traders habitually cut or pull out an animal's lower front teeth (Nekaris *et al.* 2013c). Traders may also cut teeth prior to packing slow lorises tightly into crates as during transport they often damage each other with their venomous bites. Indeed, Fuller *et al.* (2017) showed that in a single confiscation of 77 slow lorises by Cikananga Wildlife Rescue Centre, nearly 30% died in the first 6 months, with wounds, mainly bites, being the main cause of death. Other causes of death due to dental removal include dental abscess or pneumonia (Nekaris and Starr 2015). Those that do survive are no longer able to eat their preferred food (gum) (Das *et al.* 2014), 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; in the major markets in Java not only Javan slow lorises are traded but also at least four of the other six Indonesian species, and in the markets in Sumatra at least three species are regularly traded, including ones that do not occur naturally on the island. The similar appearance of slow lorises to the untrained eye results in release of slow loris species from Sumatra and Borneo into Java and vice versa, with potential for disastrous effects from hybridization or displacement of native species by introduced ones (Nekaris and Starr 2015). The ability of 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, exacerbating these problems (Kumar *et al.* 2014).

Moore *et al.* (2014) assessed the success of reintroduction of Javan slow lorises, finding a death rate of up to 90%. Illness, hypothermia and exhaustion were all implicated in the death of slow lorises. Reintroductions were started before the basics were known about the Javan slow loris' behavior, ecology or distribution. No habitat suitability assessment could be made, since details were lacking on the type of habitat the species preferred and what it avoided. Subsequently it is reported by rescue centers that success of reintroductions of Javan slow lorises is improving, but unfortunately no published data are available to verify these claims. Newspaper reports show up to 30 slow lorises are released in one site at one time; the highly territorial and venomous nature of slow lorises means that such releases are destined to have a high failure rate. A related study of pygmy slow lorises in Vietnam found that the season in which slow lorises are to be released and the age of the animals should be considered to increase the likelihood of survival (Kenyon *et al.* 2014).

To obtain vital information on the Javan slow lorises, in 2011 the Little Fireface Project instigated a study of the species' behavioural ecology in Garut District, West Java, Indonesia (Rode-Margono *et al.* 2014). This multi-disciplinary project has obtained data on home range size, social organization, infant dispersal, and feeding ecology of the slow lorises. We now know 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), that the species goes into torpor, and that the diet of

lorises comprises mainly gum, supplemented with nectar and insects (Cabana *et al.* 2017). 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 provide essential data for a national slow loris action plan. At the local level, slow lorises are often totally dependent on local people for their protection, feeding on human planted tree species and residing in human farmlands. 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 distributions and it is 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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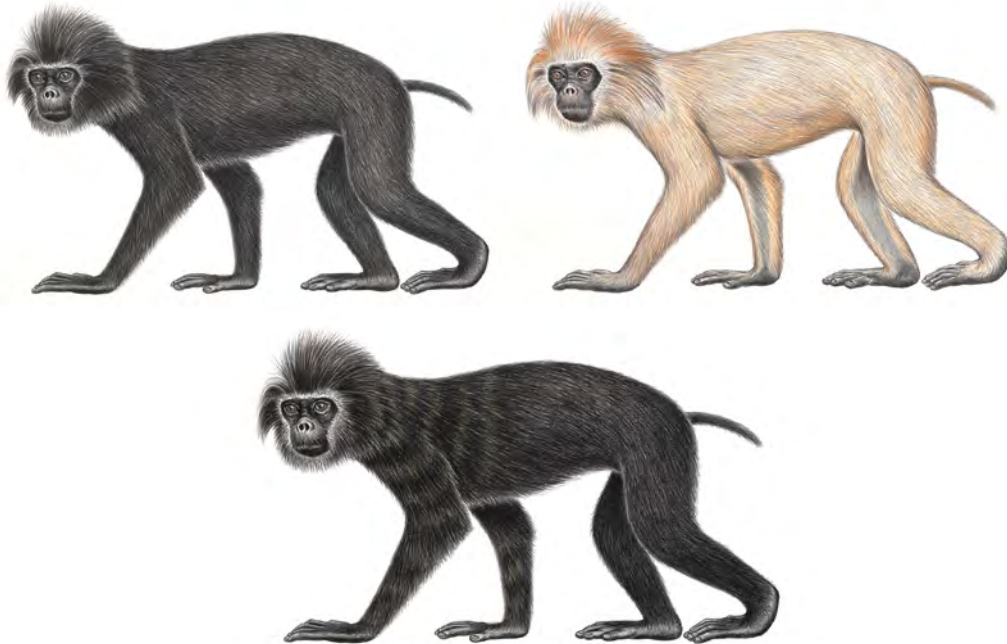
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Pig-Tailed Snub-Nose Langur
Simias concolor G. S. Miller, 1903
Indonesia

(2002, 2004, 2006, 2008, 2010, 2012, 2014, 2016)

Lisa M. Paciulli & Jessica L. Citrola



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

The pig-tailed snub-nose langur (*Simias concolor*) is again serving as the flagship species for the six Mentawai Island primates. The other three species inhabiting the 7,000 km² archipelago west of Sumatra are Kloss's gibbon (*Hylobates klossii*), the Pagai langur (*Presbytis potenziani*), the Siberut langur (*Presbytis siberu*), the Pagai macaque (*Macaca pagensis*), and the Siberut macaque (*M. siberu*). *Simias* is a monotypic genus with two subspecies: *S. c. concolor* G. S. Miller, 1903 / Masepsep that inhabits Sipora, North Pagai Island, and South Pagai Island; and *S. c. siberu* Chasen and Kloss, 1928 / Simakobu, 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 by hunting, commercial logging, and human encroachment (Whittaker 2006). The Pagai Island populations contend with forest conversion to oil palm plantations, forest clearings, product extractions by local people (Whittaker 2006), and opportunistic hunting (Paciulli 2004). In 2014, there were threats of plantation development

in Siberut, including a 1000-km² oil palm plantation, and in 2016, a 200-km² timber plantation for biomass energy production (Gaworecki 2016). Although both of these plans were canceled as a result of local opposition, protests, and environmental assessments, attempts to exploit the Mentawai Islands' natural resources are likely to continue since the national government has designated these bio-diverse, tropical islands as production forests (Gaworecki 2016).

Timber removal on a large scale is a concern as *Simias* has significantly lower densities in forests logged ~20 years previously – 2.54 individuals/km² compared to 5.17 individuals/km² in unlogged forests (Pagai Islands, Paciulli 2004). It is estimated that on the Pagais there are approximately 3,347 pig-tailed snub-nose langurs, 1,049 Kloss's gibbons, 1,545 Pagai langurs, 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/km² (Quinten *et al.* 2010).

Logging facilitates hunting by providing easier access to forested areas (Febrianti 2015). 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 2017). The locals consider pig-tailed snub-nose langur meat to be a delicacy (Febrianti 2015), and entire groups can be eliminated in a single hunt (Hadi *et al.* 2009). 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-nose langurs (Quinten *et al.* 2014). On Siberut, hunting reduces pig-tailed snub-nose langur group size, significantly impacts adult sex ratios, and affects the number of immatures in groups (Erb *et al.* 2012).

The uncertainty of Indonesian government land-use means that land function and protection level on the Mentawai Islands can change at any time with little notice, putting the species further at 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.* 2016), hunting is much more prevalent there than elsewhere, with ~4,800 primates being removed from the park each 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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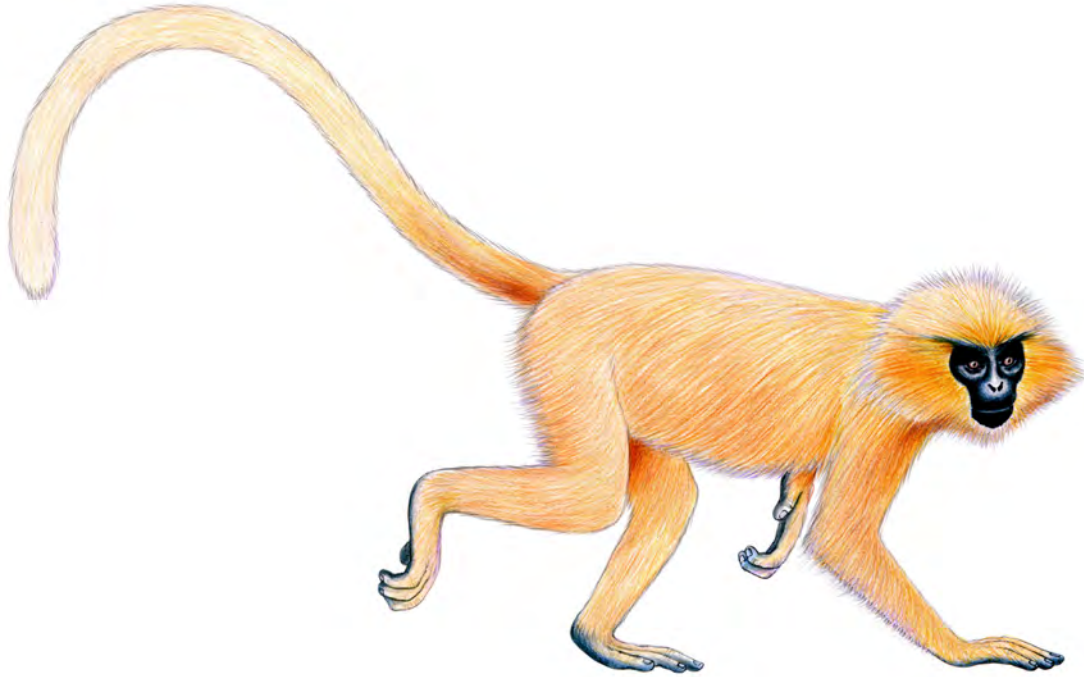
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Golden Langur
Trachypithecus geei Khajuria, 1956
India and Bhutan
(2016)

Rekha Chetry, Dilip Chetry & P. C. Bhattacharjee



The Golden Langur (*Trachypithecus geei*) (Illustration: Stephen D. Nash)

The golden langur (*Trachypithecus geei*) is a beautiful primate discovered in 1953 (Gee 1955), and found to be restricted to India and Bhutan. The distribution of golden langurs in India is restricted to an area bounded by the Manas River in the east, Sankosh in the west, and the Brahmaputra River in the south (Srivastava 1999). The distribution of the species in India extends over 2,500 km² between 26°15'N to 26°33'N and 89°47'E to 92°55'E. Its distribution is equally restricted in neighbouring Bhutan, where the golden langur is found between the Sankosh River and the Chamkhar-Mangde-Manas river complex (Wangchuk 2005). In Bhutan, the range of the species covers an area of 4,782 km² (Wangchuk 2005). There are two small introduced populations of golden langur outside this range; one on Umananda River Island in the Brahmaputra River at Guwahati in Assam, and the other in Sipahijala Wildlife Sanctuary in Tripura (R. Chetry and D. Chetry 2009).

Golden langurs primarily inhabit the wet evergreen forests and the tropical semi-evergreen forests of their range. The sal (*Shorea robusta*)-dominated forests of

western Assam and secondary forests also support the golden langur. In Bhutan, they generally inhabit warm broadleaf forests between 1,000 m and 2,400 m asl and sub-tropical forests between 200 m and 1,000 m asl. The species' elevational range has been ascertained to be up to 2,400 m asl. However, It has also been recorded from 2,600 m at Chendebji in Bhutan (Wangchuk 2005). The total potential habitat for golden langurs in Bhutan is 4,782 km², of which 3,475 km² is the actual available habitat (Wangchuk 2005, Choudhury 1990). The estimated available habitat in India is 1,255 km² (Srivastava *et al.* 2001).

Since its discovery, the golden langur has been considered to be a monotypic species. Wangchuk *et al.* (2003), however, strongly advocated that the golden langur has two subspecies based on mitochondrial DNA studies, namely a northern subspecies (*Trachypithecus geei bhutanensis*) and southern subspecies (*Trachypithecus geei geei*). Across their known range, golden langurs can be found in sympatry with the Assamese macaque (*Macaca assamensis*), the rhesus macaque (*Macaca*

mulatta) and the slow loris (*Nycticebus bengalensis*). Golden langurs maintain a peaceful co-existence with the sympatric diurnal species. Primarily arboreal, the golden langur spends 99% of its active time in trees (Biswas 2004). They generally explore the top and middle strata of the forest, but in degraded habitat they also descend to the ground (R. Chetry and D. Chetry 2009). The diverse social groups generally observed in the golden langur are – uni-male: multi-female troops or societies, bi-male: multi-female troops, multi-male: multi-female societies, all-male bands or all-male societies, and lone males (Biswas 2004). The most common social dynamic is the uni-male: multi-female troop or society (Biswas 2004). Social bonds between troop members are very strong. The annual home range is between 10 and 58 ha for diverse social troops in different habitats (R. Chetry and D. Chetry 2009). As far as day range is concerned, each golden langur troop covers around 200–700 m. Major activities of golden langurs are locomotion, resting, feeding and monitoring. Other activities include grooming and playing. Golden langurs spend 12.8–33% of time feeding, 40–63.1% resting, 6.3–19% on locomotion, 5–11.5% monitoring, 2–3.7% playing and 0.3–6% grooming and engaging in other social activities (Mukherjee 1996; D. Chetry 2002; Medhi and Chetry 2003; Biswas 2004; Medhi 2004).

Leaves (both young and mature) constitute 60% or more of their daily diet. In addition to leaves, they regularly eat other plant parts such as fruits, seeds and flowers, stem cortex and twigs. Gum feeding, soil feeding, algae feeding, snail feeding and alcoholic effluent feeding have also been observed (Medhi 2004; Biswas 2004). Identified plant species used as food by the golden langur number more than 200. They use tall trees for sleeping in order to avoid nocturnal predators. Leopards (*Panthera pardus*), wild dogs (*Cuon alpinus*) and pythons (*Python morulus*) are the prominent predators of golden langurs (R. Chetry and D. Chetry 2009). Domestic as well as stray dogs (*Canis familiaris*) attack golden langurs near human habitats (D. Chetry *et al.* 2005). Golden langurs are seasonal breeders, and June to January is the breeding season. The estimated gestation period is 168–180 days, and inter-birth interval is two years (Biswas 2004; R. Chetry and D. Chetry 2009). January to June is the birthing season. Male golden langurs attain sexual maturity at 5–7 years while the age of sexual maturity for a female is four years.

At present, 86% of the golden langur population is in Bhutan, with Wangchuk (2005) estimating the population for the entire country to be approximately

6,637 individuals. Ghosh (2009) and Biswas *et al.* (2010) directly sighted 5,141 golden langurs in 566 troops in Assam (India). The global population of the species was thus estimated at >12,000 (R. Chetry and D. Chetry 2009; Horwich *et al.* 2013).

The golden langur is a Schedule –I species in the Wildlife (Protection) Act of India 1972 (amended 2002). The Forest and Nature Conservation Act of Bhutan 1995 has also designated it as a Schedule–I species. It is classified as Endangered on the IUCN Red list and is an Appendix-I species in CITES. The main conservation threat the species is facing is loss of habitat. Srivastava *et al.* (2001) reported a 50% loss of original habitat for golden langurs in India. Parallel to habitat loss, habitat fragmentation and habitat shrinkage are also affecting the species. Golden langurs were extirpated in as many as eight forest patches between 1970 and 1990 (Choudhury 2002). There are reports of individuals dying due to electrocution and roadkill, which is a problem for the species in certain areas (R. Chetry and D. Chetry 2009). Domestic dogs have emerged as another threat for the species in forest fringe villages (D. Chetry *et al.* 2005). Hunting and smuggling, anthropogenic dependency on forests, social unrest, inbreeding and diseases are some other threats to golden langurs in India. Chakrashila Wildlife Sanctuary is the only protected habitat for this endangered species in India. It is more secure in neighbouring Bhutan, with almost 50% of its habitat in Bhutan falling in three protected areas, namely Royal Manas National Park, Black Mountain National Park and Phipsoo Wildlife Sanctuary. Yet in Bhutan, anthropogenic pressure, changing land-tenure systems, development, shifting cultivation and commercial logging are posing threats to golden langurs and their habitat (Wangchuk 2005). Construction of suspension bridges over the Chamkhar has opened the way for hybridization between capped langurs and golden langurs (Wangchuk *et al.* 2009; Choudhury 2008). Ram *et al.* (2016) also revealed possible hybridization of golden langurs with capped langurs. According to Wangchuk (2005), hybrids form 15% of the total golden langur population in Bhutan. The conservation scenario of the golden langur is, therefore, alarming and critical. Conservationists are extremely concerned that each of the major threats to golden langurs are increasing and likely to worsen considerably over the coming years, despite a number of conservation initiatives.

Golden langurs have undoubtedly drawn the attention of the public and law makers, yet they are still overshadowed by big charismatic species

such as the tiger and elephant. There is a clear and urgent need for further surveys through the entire range of the species in order to record the spatial and temporal demographic changes in dwindling forest patches. Given that in India most of its populations are outside protected areas, there is an urgent need to protect such as the proposed Ripu-Chirang Wildlife Sanctuary, and Kakoijana Wildlife Sanctuary in Assam. Chakrashila Wildlife Sanctuary, which supports a healthy population along with its adjacent reserve forests such as Sreegram Reserve Forest, Katrigacha Reserve Forest, Buxamara Reserve Forest and Nadangiri Hill Reserve Forest, should be upgraded to national park status. Forest fragments with golden langur populations can be connected with forest corridors in the near future. A future metapopulation management plan will need to incorporate the translocation of threatened and fragmented populations. New projects to address these conservation and research issues should be implemented to ensure the long-term conservation of the golden langur and its habitat.

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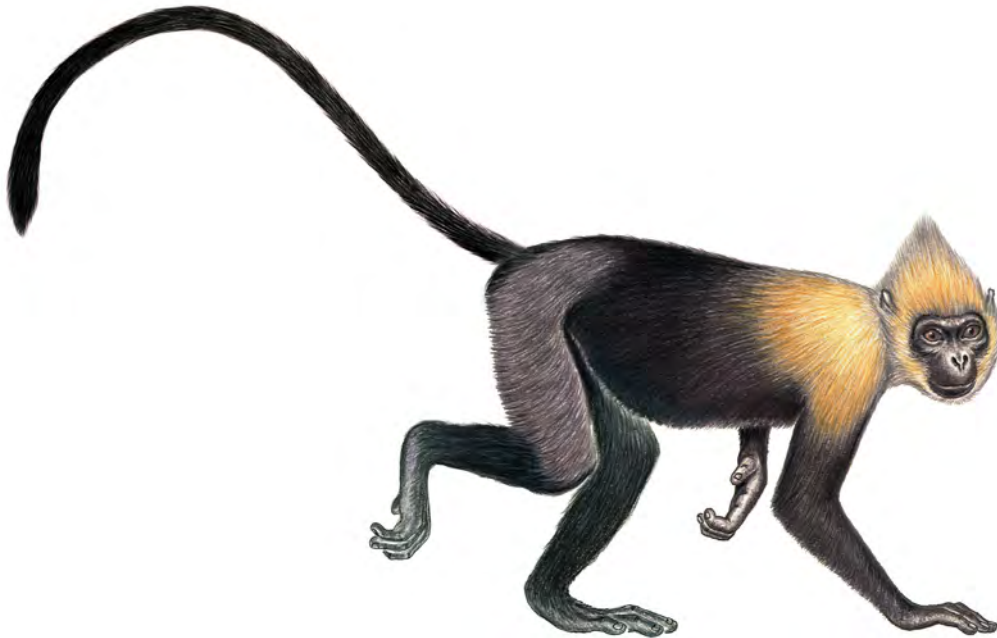
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Golden-Headed or Cat Ba Langur
Trachypithecus poliocephalus (Trouessart, 1911)
Vietnam
(2000, 2002, 2004, 2006, 2008, 2010, 2012, 2014, 2016)

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



The 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 six to seven 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 km²) 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. In the latter half of 2015, numbers fell from the mid-high 60s to the low 50s and have since been slowly recovering. This has raised concerns that as langur numbers recover, interest in poaching by people from adjacent regions may also revive.

Although the growth of the population is encouraging, the overall status of the species remains critical and the total population is 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 babies born in 18 months. The long inter-birth cycle results in an alternating high-low birth cycle with periodic back-to-back high birth years. The 2014/2015 years were followed by a low birth year in 2016. Births take place throughout the year, with a peak in January–April, just prior to the rainy season (N. Leonard pers. obs.).

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 Island. Unfortunately, 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 opportunity. 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 stopped years ago, although the 2015 drop in numbers calls this into question, 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 Island 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, 2016)

Le Khac Quyet, Dong Thanh Hai & Tilo Nadler



The 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 (*R. bieti*, *R. brelichi*, *R. roxellana*), the Myanmar snub-nosed monkey (*R. strykeri*) is found in Myanmar and China (Geissmann *et al.* 2011; Long *et al.* 2012), and *R. avunculus* occurs only in northeastern Vietnam. *Rhinopithecus avunculus* was discovered in 1911, and collected on perhaps no more than two further occasions over the course of the next 50 to 60 years. Consequently, it was presumed to be extinct by a number of primatologists until it was rediscovered in 1989 in Na Hang, Tuyen Quang Province (Ratajszczak *et al.* 1990). Historically the species occurs only east of the Red River between about 21°09'N and 23°N. Due to widespread deforestation and intensive hunting in recent decades, its distribution has become severely fragmented (Nadler *et al.* 2003; Nadler and Brockman 2014). The total population of the Tonkin snub-

nosed monkey is currently believed to be less than 250 individuals. *Rhinopithecus avunculus* is Critically Endangered (IUCN 2017). Recent evidence suggests there are only five known locations where Tonkin snub-nosed monkeys occur, and each is 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 most probably an overestimation (Thach Mai Hoang 2011). In 2007, the occurrence of 22 individuals was reported (Dong Thanh Hai 2007), and the most recent field surveys in 2010 estimated 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 a population. However, local reports indicated that a small group of 8–12 individuals possibly still occurs in the area (Dong Thanh Hai *et al.* 2006). The current threats to the population are hunting and habitat destruction. Conservation efforts should target reducing human activities inside the reserve.

In 2001, a population of about 60 Tonkin snub-nosed monkeys was discovered in Khau Ca, close to Du Gia Nature Reserve, Ha Giang Province. A census in April 2015 confirmed 125–130 individuals in the declared Tonkin Snub-nosed Monkey Species/Habitat Conservation Area Khau Ca, Ha Giang Province (Le Khac Quyet in prep.). This is the only population that is not immediately threatened. Population and habitat monitoring, scientific research, conservation education, public awareness and community participatory activities are being linked to increase protection efforts under the supervision of University of Colorado Boulder, Fauna and Flora International (FFI), Denver Zoo and Wildlife Reserves Singapore.

In 2007, a population of about 20 Tonkin snub-nosed monkeys was discovered in a small forest patch in Tung Vai Commune, Quan Ba District, Ha Giang Province close to the border with China (Le Khac Quyet and Covert 2010), and was confirmed by a census in 2014 (Nguyen Van Truong 2014). The most recent survey in November 2016 recorded four groups, in total 15–21 individuals in the area (Nguyen Van Truong pers. comm.). The population at Tung Vai is threatened through hunting and habitat loss due to timber exploitation, shifting cultivation and the collection of non-timber forest products for commercial purposes. Immediate conservation measures should include;

establishing and training of patrol groups, raising awareness and assessing the range of the population and the impact of cardamom production to the habitat.

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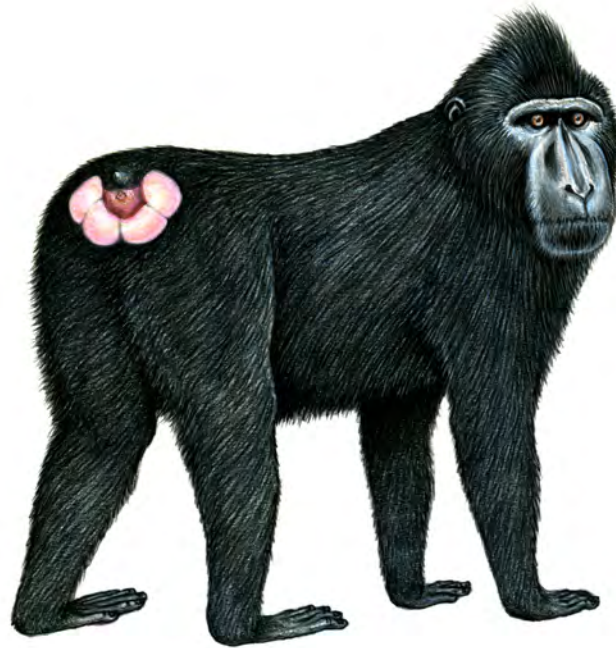
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Crested Macaque
Macaca nigra (Desmarest, 1822)
Indonesia (Sulawesi)
(2016)

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*Denotes equal contribution to this report



The Crested Macaque (*Macaca nigra*) (Illustration: Stephen D. Nash)

Crested macaques are one of the seven macaque species endemic to the island of Sulawesi, Indonesia (Fooden 1969), inhabiting only the very tip of the NE Sulawesi arm (Riley 2010). Sulawesi is a biodiversity hotspot in the Wallacea eco-region, characterized by a unique blend of Asian and Australian flora and fauna (Whitten *et al.* 1987) and an extremely high degree of endemism, especially among mammals (Musser 1987). The seven extant macaque species are of specific importance as they are living fossils of a primate adaptive radiation and speciation (Riley 2010). All species are threatened, but only the crested macaques have been classified as Critically Endangered by the IUCN (Supriatna and Andayani. 2008). Their population has decreased by 75–90% in the last 30 years, mainly due to poaching and illegal logging (Melfi 2010; Palacios *et al.* 2012).

Crested macaques are characterized by their distinctive thin strip of raised, longer hair extending from the forehead to the back of the head and their all black pelage; the only other color is on the ischial callosities,

which are pink. The crest separates the species from their also black sister taxon, the Gorontalo macaque (*M. nigrescens*). Their tails are short (22 mm), which in the past led to observers classifying them incorrectly as Celebes apes, instead of monkeys (Rowe 1996).

Crested macaques are diurnal and live in large multi-male, multi-female groups with female philopatry and male dispersal (O'Brien and Kinnaird 1997; Duboscq *et al.* 2013; Marty *et al.* 2017). Although females form matrilineal groups, their social interactions are characterized by a broad net of social partners, low intensity and often bi-directional aggression, and high levels of reconciliation (Duboscq *et al.* 2013, 2014). Males, in contrast, fight fiercely for dominance, and frequently migrate between groups, and mean alpha-male tenure is particularly short (Marty *et al.* 2017). Due to this high degree of male competition, crested macaques are sexually dimorphic, with males approximately twice as large as females (males: 12 kg, Marty *et al.* 2017; females: 6 kg, Thierry *et al.* 2004), and have larger canines (Thorén

et al. 2006). Males have two sexual signals, loud calls, and a red coloration of their scrotum that reflects their dominance status (Engelhardt *et al.* 2009; Neumann *et al.* 2009). Females have a cyclic swelling of their perianal skin that indicates quite reliably the time of ovulation (Higham *et al.* 2012). Although females typically mate with multiple males, male reproductive success is highly skewed towards alpha males (mean of 65% paternities, Engelhardt *et al.* in press).

Crested macaques survive best in primary and secondary forests (O'Brien and Kinnaird 1997) but can also be found sporadically in logged forests and plantations (Rosenbaum *et al.* 1998). Their habitat is highly seasonal, with annual rainfall of 1,550 mm – 2,400 mm, mostly falling from May to October (O'Brien and Kinnaird 1997). Crested macaques are semi-terrestrial, spending at least 60% of their day on the ground (O'Brien and Kinnaird 1997). Their diet consists primarily of fruit, supplemented with other plant parts as well as invertebrate and vertebrate prey. Female-female competition for food resources seems to have a significant effect on foetal survival, and infants seem to be threatened by the male reproductive strategy of infanticide (Kerhoas *et al.* 2014).

One stronghold of crested macaques, and likely the most viable surviving population, is on the northeastern coast of Sulawesi, in the 8,867-ha Tangkoko Reserve, formerly the Tangkoko-Batuagus Nature Reserve and Batu Putih Recreation Park (Supriatna and Andayani 2008; Riley 2010; Palacios *et al.* 2012). A survey by Palacios *et al.* (2012) indicated that one half of the park supported a population of 1,951, or 44.9 individuals per km². Another survey by Kyes *et al.* (2013) resulted in a higher estimate of 61.5 individuals/km², which is close to the population density of 76 individuals/km² estimated thirty years ago by Sugardjito *et al.* (1989). Kyes' survey was, however, conducted mainly close to a research station, and included a number of habituated groups that were under daily observation. The robust presence of those animals is likely due to the permanent presence of researchers, conservation efforts, and the connected ongoing ecotourism supported by locals (McKinnon and McKinnon 1980; O'Brien and Kinnaird 1997; Kyes *et al.* 2013; Macaca Nigra Project 2017; Selamatkan Yaki 2017; Tangkoko Conservation Education Programme 2017; Tasikoki Wildlife Rescue Center 2017). Areas

outside Tangkoko do not have such hopeful numbers. A census of 22 locations outside the park resulted in very few sightings of *M. nigra*, with most sites having from none to less than 10 individuals/km² (Melfi *et al.* 2007).

Significant reduction in population size in crested macaques is connected to habitat loss and hunting (Myers *et al.* 2000). Habitat loss is driven by mining, slash-and-burn agriculture, and cash crops (Melfi 2010). In addition, the consumption of macaque meat during holidays, weddings, and parties is a long-held tradition (Clayton and Milner-Gulland 2000), although hunting rates became unstable starting in the 1970's (O'Brien and Kinnaird 2000) and remain so to this day (Hilser *et al.* 2013). As farms expand into forests, macaques have started to include crops in their diet, resulting in conflict with farmers (Riley and Priston 2010).

While a number of research and community outreach programs have been conducted successfully around the Tangkoko Nature Reserve (see above), conservationists are concerned that threats to this species are increasing and likely to get worse in the coming years. For example, illegal fires, used to clear forest for cattle grazing, are encroaching on Tangkoko (Palacios *et al.* 2012). As this is the only protected area in the crested macaques' geographic range, urgent action is needed to stop this encroachment. While recent population genetic analysis indicates that the Tangkoko crested macaques still remain a genetically viable population despite the severe conditions they have faced (Engelhardt *et al.* in press), further population decline could jeopardize the health of their gene pool.

Since most population surveys have been in Tangkoko (Sugardjito *et al.* 1989; Palacios *et al.* 2012; Kyes *et al.* 2013), where numbers are also expected to be highest, we do not currently know the precise number of crested macaques left on Sulawesi. While the Tangkoko population remains genetically viable (Engelhardt *et al.* in press), we have no data regarding the degree of genetic inbreeding or health status of the overall population. Thus, a proper assessment is urgently needed. As human populations continue to grow in the region, so will the interactions between humans and macaques. How they will continue to survive in the face of this anthropogenic habitat change remains to be seen.

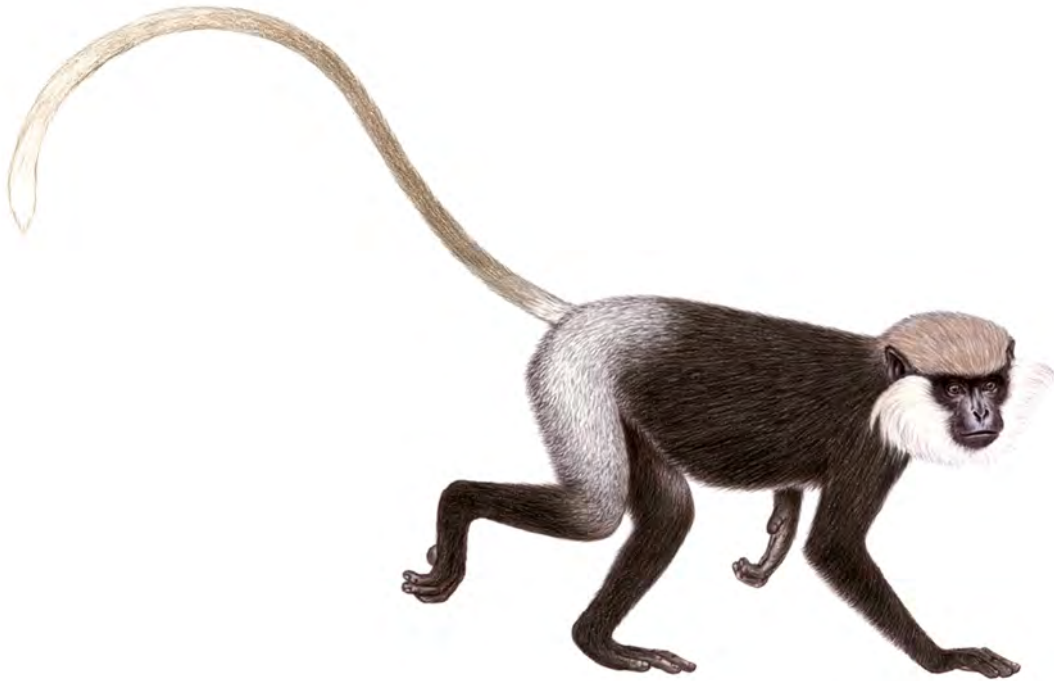
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Purple-Faced Langur
Semnopithecus vetulus (Erxleben, 1777)
Sri Lanka
(2004, 2006, 2008, 2010, 2012, 2014, 2016)

Rasanayagam Rudran & Surendranie J. Cabral



The Purple-Faced Langur (*Semnopithecus vetulus*) (Illustration: Stephen D. Nash)

At the 2016 IPS Congress in Chicago, it was decided to include all four subspecies of Sri Lanka's purple-faced langur (*Semnopithecus vetulus*) in the "Primates in Peril" report, whilst only the western purple-faced langur, *S. v. nestor*, had been included in previous iterations. Extensive deforestation and agricultural expansion occurred after Sri Lanka's twenty-six-year civil war ended in 2009, and this resulted in escalating conflicts between humans and monkeys. This escalation further undermined the long-term survival of the country's three endemic and already threatened monkeys (*Semnopithecus vetulus*, *S. priam thersites* and *Macaca sinica*). As public outcry and political pressure to resolve the conflicts increased, several government institutions and non-governmental organizations led by the SPEARS Foundation, helped the country's Department of Wildlife Conservation (DWC) with an action plan for people to conserve and coexist with all three monkey species. The plan was submitted to the Ministry of Sustainable Development and Wildlife in March 2016 for cabinet approval. While awaiting government approval, funds from foreign donors to the SPEARS Foundation helped its team to implement

some key elements of the action plan. One was the proposed establishment of Community Conservation Areas (CCAs), which would be administered and managed sustainably by local communities under the supervision of the DWC. To find suitable sites for CCAs, the SPEARS team analyzed 493 complaints of human-monkey conflicts received by DWC between 2007 and 2015. The analysis indicated country-wide existence of human monkey conflicts. Therefore, field surveys were conducted in different districts to collect data on human attitudes towards monkeys and the damage caused by these animals. Data were also collected on the population sizes of all monkey species and the extent of forests currently available for their survival. Information from the four purple-faced langur subspecies is presented below.

Western purple-faced langur (*Semnopithecus vetulus nestor*): The range of this subspecies includes the most densely populated region around Colombo, the country's capital. Urbanization poses a serious threat, therefore, to the long-term survival of this Critically Endangered and endemic subspecies (Dittus *et al.*

2008; Rudran *et al.* 2009). A survey conducted ten years ago (Rudran 2007) indicated that 81% of *S. v. nestor*'s historical range (Hill 1934; Phillips 1935) consisted of deforested and human dominated landscapes. Due to the reduction of *S. v. nestor*'s preferred habitat, much of its current population subsists mainly on fruit from home gardens (Dela 2007; Rudran 2007). The nutritional consequences of feeding on a low diversity diet of cultivated fruits are unclear, but considered detrimental to the folivorous *S. v. nestor* (Rudran 2015). Besides undermining the subspecies' diet, deforestation results in habitat fragmentation, and forces animals to travel on the ground and along power lines. These movements increase mortality due to attacks by dogs, speeding vehicles, and electrocution (Parker *et al.* 2008). In some parts of its range, *S. v. nestor* was occasionally shot and killed while feeding in home gardens (Dela 2004). Thus, deforestation increases human-induced fatalities, which reduces group sizes and appears to have led to the extirpation of *S. v. nestor* in certain parts of its range (Rudran 2007). Three districts surveyed last year were within this subspecies' range, and accounted for a large proportion of the complaints (30%) received by the DWC. However, 50% of a random sample of stakeholders interviewed in these districts said they were indifferent to damage caused by monkeys. This suggests that in districts with less conflict there would be a larger proportion of people indifferent to damage, which would be favorable for the establishment of CCAs. Besides the survey, other activities that focussed on garnering support from people were conducted specifically to help conserve *S. v. nestor*. These activities promoted environmental awareness among school children, economic stability among adults, and elder care among seniors of local communities. To promote environmental awareness among school children, lectures were presented to 1,060 students, and approximately 280 children participated in nature walks to appreciate the benefits people derive from wild plants and animals. To promote economic stability, 90 adults were taught to cultivate pepper, which is a cash crop with considerable demand in world markets. After the training, the trainees received pepper plants to grow in their gardens in order to generate income. Another initiative to promote economic stability involved an ecotourism program that is being developed at Thummodara. Several workshops have been conducted to train young adults as nature guides. A workshop was also held to invite the community to participate in ecotourism, by producing local handicrafts, organic food and drinks and other items that could be sold to

visitors. A website and a brochure have been prepared as well, to advertise the ecotourism program and attract local and foreign visitors.

Highland purple-faced langur (*S. v. monticola*):

This subspecies, also known as the bear monkey, was investigated for two years at Horton Plains by Rudran (1973a, 1973b) nearly forty-five years ago. When the area was surveyed again in 2016, Rudran noted appreciable changes to the vegetation. Many species previously recorded as important food plants of the bear monkey were dead or dying. This appeared to be primarily due to debarking of the adult trees by the sambar (*Cervus unicolor*) population that had increased because of the soft grass introduced to Horton Plains with the fertilizer used by a now defunct potato farm (Adikaram *et al.* 1999). Sambar also fed on the saplings of this subspecies' food plants, which has inhibited regeneration. The death and lack of regeneration of the food plants appears to have undermined this subspecies' survival. A census was not conducted in 2016, but early morning loud calls of harem males were considerably less frequent than before, indicating a population decline.

Southern purple-faced langur (*S. v. vetulus*):

This subspecies was recorded in all three surveys conducted in the Galle and Matara districts in 2017. Data from these surveys are still being analyzed, but *S. v. vetulus* was the most numerous monkey in these districts. A long-term study of *S. v. vetulus* (Roscoe *et al.* 2013) reported several threats to the future survival of this subspecies. These threats were the same as those experienced by *S. v. nestor*. Additionally, a major highway constructed through *S. v. vetulus*'s range is expected to create a permanent barrier to gene flow between the populations found along the coast and the interior of the country.

Northern purple-faced langur (*S. v. philbricki*):

This subspecies was investigated for two years in the late 1960s (Rudran 1973a, 1973b) when conflicts with humans were not a serious issue. In the late 1970s, however, the impact of the Accelerated Mahaweli Development Program (AMDP) on wildlife within *S. v. philbricki*'s range became a serious concern, and the US Agency for International Development (USAID) funded an Environmental Impact Assessment (EIA) of the program. After a year of field investigations, the EIA team recommended the establishment of four new National Parks around the development area, to mitigate the impact of agricultural expansion on wildlife (Tippetts-Abbott-McCarthy and Stratton

1980). While these areas may have historically been safe environments for *S. v. philbricki*, a study conducted in 2010–2011 showed that the area around Mihintale Sanctuary was severely fragmented (Wickremasinghe *et al.* 2016). This study also reported illegal hunting as a major threat for the future survival of *S. v. philbricki*. Other investigators have also found low frequency use of langur meat and body parts as food, and for medicinal purposes and rituals (Nahallage and Huffman 2013).

Final Remarks: Although Sri Lanka's monkeys face a perilous future (Rudran 2013), there is hope that they can be conserved. One reason for hope is that most Sri Lankans follow the Buddhist doctrine of compassion towards all living things. Therefore, promoting this doctrine and Buddha's own reverence of the forest present opportunities to deter deforestation in a country steeped in cultural traditions. Another reason for optimism stems from a government decision taken ten years ago, to increase Sri Lanka's forest cover from 27% to 36% using native plants, in order to achieve the country's economic development goals (Yatawara 2011). Although governments have changed since this decision was taken, the policy of increasing forest cover has remained. The political will to increase forest cover augurs well for the future. It is important that the Sri Lankan government approve the 2016 action plan in order to help ensure a steady flow of financial support to conserve Sri Lanka's monkeys.

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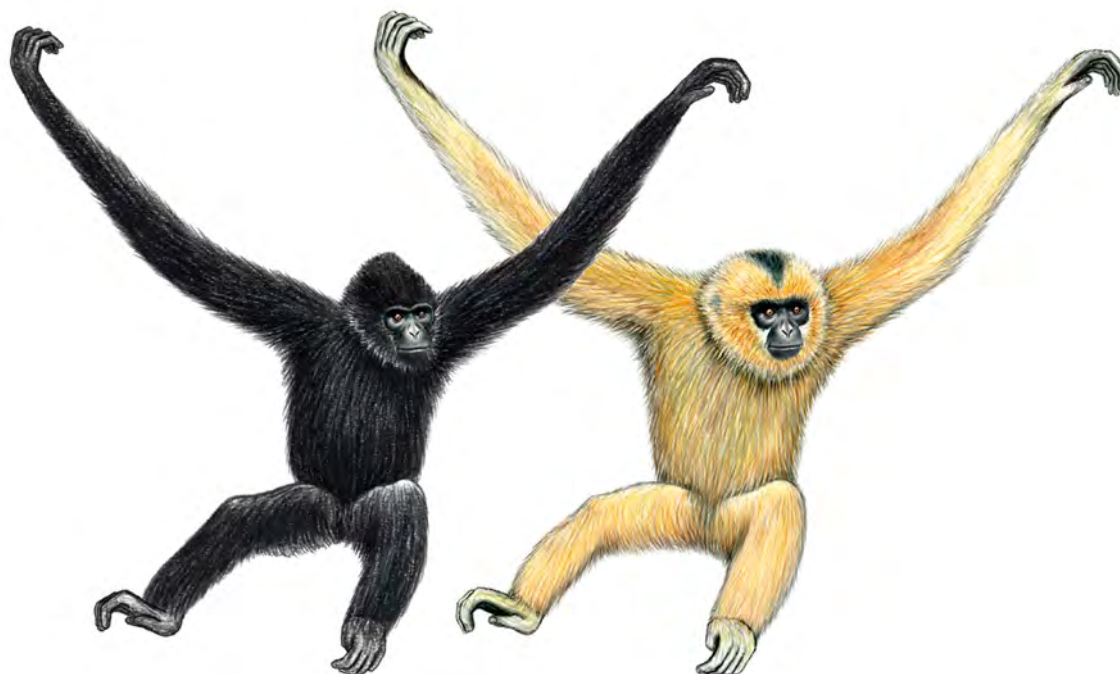
Hainan Gibbon

Nomascus hainanus (Thomas, 1892)

China (Island of Hainan)

(2000, 2002, 2004, 2006, 2014, 2016)

Bosco P. L. Chan



The 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 the most endangered of the gibbons and restricted to the island of Hainan (Geissmann 2003; Geissmann and Chan 2004; Wu *et al.* 2004). Adult male Hainan gibbons 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 as compared to 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 measuring 16 km² in Bawangling National Nature Reserve (BNNR) (Liu *et al.* 1989). In 1998, the population was said to be 17 (Kadoorie Farm & Botanic Garden 2001). A brief survey by a small research team estimated 12–19 individuals in four groups in 2001–2002 (Wu *et al.* 2004); a comprehensive gibbon survey organized by the provincial conservation authorities in October 2003 could only confirm 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). Determined conservation actions were launched in 2003 (Fellowes *et al.* 2008; Mootnick *et al.* 2012), and the population has been slowly recovering. Together with BNNR, the Hong Kong – based Kadoorie Farm and Botanic Garden (KFBG) have been conducting regular population monitoring, using the same methodology as in the 2003 comprehensive survey. The latest population census confirmed 4 breeding groups and 2 solitary gibbons, with a total of at least 27 gibbons; the four

well-monitored social groups are composed of 5, 7, 10 and 3 individuals, respectively. The exact number of individuals cannot be ascertained due to a number of dispersed subadults with erratic ranging patterns.

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 KFBG, BNNR, and the Hainan Wildlife Conservation Bureau of the Hainan Provincial Forestry Department. Their work includes regular monitoring of the gibbons, restoring the degraded lowland forest with gibbon food trees, community-based conservation work in villages adjoining the gibbon home range, and surveying for remnant Hainan gibbons in other potential sites throughout the island. In 2014, the Super Typhoon Rammasun hit Hainan and caused severe landslides throughout BNNR, including the home range of the Hainan gibbons. KFBG has been supporting the Reserve to restore damaged gibbon habitat by building a canopy bridge in a critical valley, and by planting over ten thousand saplings and young trees to enhance canopy connectivity. The team also successfully conducted the first sleeping tree study on Hainan gibbons, which has important implications for patrolling and forest restoration efforts.

Two research teams, of the Zoological Society of London (ZSL) and the Guizhou Normal University, are studying the gibbons intermittently and have published a number of scientific papers on various aspects of their biology and behavior, which enhances our understanding of the species (e.g. Bryant *et al.* 2016; Deng and Zhou 2016; Turvey *et al.* 2016).

With less than 30 Hainan gibbons confirmed, surviving in just one small forest block, the Hainan gibbon is one of the world's rarest primates. Following over a decade of determined conservation effort, support from the government and the surrounding ethnic community, conservation of the gibbons and their habitat has improved significantly. With the slow but 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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Bornean Orangutan
Pongo pygmaeus (Linnaeus, 1760)
Indonesia (Borneo)
(2016)

Simon J. Husson, Marc Ancrenaz, Elizabeth J. Macfie, Sri Suci Utami-Atmoko & Serge A. Wich



The Bornean Orangutan (*Pongo pygmaeus*)
(Illustration: Stephen D. Nash)

The Bornean orangutan, *Pongo pygmaeus*, appears in this list for the first time because of large population declines over the past fifty years, acceleration of this decline in the past twenty years, and persistence of threats faced by this species. Orangutans have long been considered susceptible to hunting because of their slow reproductive rate and low natural densities, but the former vastness of the Bornean forests meant that orangutans were not considered at risk of extinction. This notion has been challenged in the past two

decades with rapid expansion of oil palm plantations across Borneo, coupled with devastating fires in the orangutan's peatland stronghold, and the realization that in many parts of its range the species is still hunted for meat. Conservationists now forecast the orangutan population to decline by 86% over three generations, causing IUCN to upgrade the conservation status of the Bornean orangutan to Critically Endangered in 2016 (Ancrenaz *et al.* 2016), joining the Sumatran orangutan, *Pongo abelii*, at the highest level of extinction risk.

This species is endemic to the island of Borneo. Bornean orangutans are lowland specialists, rarely found above 500 m, but below that altitude they are found throughout many different dryland and swamp forested habitats on a range of soils. Borneo's geography is dominated by the Muller-Schwaner mountain chain that forms a spine down the centre of the island, and this, together with the headwaters of the largest rivers, has resulted in a bottleneck for dispersal and the evolution of three distinct subspecies. *Pongo pygmaeus wurmbii* is the most widespread, occurring in the south of the island in the Indonesian provinces of Central and West Kalimantan, with a few individuals in South Kalimantan. *Pongo p. morio* occurs in the north and east, in the Indonesian province of East Kalimantan and the Malaysian state of Sabah; and *P. p. pygmaeus* is the least numerous, with a restricted range in West Kalimantan north of the Kapuas river and in the Malaysian state of Sarawak south of the Rajang River. Orangutans are absent from northern Sarawak and Brunei, from large parts of the central and eastern Muller-Schwaner hills and are virtually absent from the south-east corner of Borneo.

While the most recent estimate of 46,952–72,941 Bornean orangutans (Utami-Atmoko *et al.* 2017) appears substantial, this masks very serious threats. There has been a dramatic decline in numbers over the past few decades, including the fragmentation of the current range into hundreds of forest remnants with very few large enough to hold 1,000 individuals or more.

Scientists estimate that there were 288,500 Bornean orangutans in 1973, and they numbered over 100,000 as recently as 2010 (Ancrenaz *et al.* 2016), equating to an average loss of over 5,000 per year. Between 1973 and 2010, 39% of Borneo's lowland forests were cleared (Gaveau *et al.* 2014), resulting in the loss of 98,730 km² of orangutan habitat (Ancrenaz *et al.* 2016). This was primarily for conversion to plantations and from forest fires. Of the remaining habitat in the year 2010, over 5 million ha was designated for future conversion (Wich *et al.* 2012) to meet increasing international demand for palm oil and to develop massive acacia plantations for pulpwood. A recent meta-analysis for Borneo revealed that as many as 10,000 orangutans were found in forests earmarked for oil palm agriculture but not yet converted (Meijaard *et al.* 2017). Even protected areas are shrinking at a rate of 1–1.5% per year in Indonesia (Curran *et al.* 2004; Husson *et al.* 2015) due to encroachment, smallholder plantation development and fire.

Massive forest fires ravaged the peat-swamp forests of southern Borneo in 2015, driven by a months-long drought linked to a huge El Niño event, and resulting from decades of peatland drainage for timber extraction and development. Drained peatland burns easily, and in 2015 fires started by companies and smallholders alike got out of control and spread, destroying huge areas of orangutan habitat. In peatland areas, the fires were only extinguished by the arrival of the wet season. Precise figures are not yet available, but one estimate suggests as much as 800,000 ha burned, an area that could support between 8,000 and 16,000 orangutans at normal peat-swamp densities (Glauber *et al.* 2016; Husson *et al.* 2009). This mirrored similarly destructive fires in 1997–1998, and it appears a pattern of periodic fires has become established, with large fires every 10 years or so linked to the occurrence of strong El Niño events. Worryingly, climate change is projected to increase the frequency and severity of dry season droughts, and the combined impacts of climate change, fire and agricultural development are predicted by Struebig *et al.* (2015) to cause the almost complete loss of this habitat in southern Borneo within 80 years.

Compounding this habitat loss is degradation of remaining forests, through logging in licensed timber concessions and illegal logging in protected areas. Selective logging can reduce carrying capacity by up to 30% (Husson *et al.* 2009). The other major threat is hunting. Traditional hunting for food has always been

commonplace in parts of Borneo, and the killing of females to capture infants for the pet trade has long been a problem (Rijksen and Meijaard 1999). The scale of hunting has recently been quantified and the numbers are much larger than previously thought. A mean estimate of 2,256 orangutans killed in Kalimantan each year (Meijaard *et al.* 2011) explains almost a fifth of the orangutan decline over the past four decades (Ancrenaz *et al.* 2016). Instances of human-orangutan conflict are increasing, as more forests are opened up and apes displaced from their natural habitat.

In summary, the picture is one of continuous, rapid decline of one of the world's most iconic primates. The coastal peatlands are burning, the hill forests of the interior are being logged and hunted, and everywhere else forest is being converted to plantations. To protect this species in the future, we must assess where orangutan populations are most resilient. The *P. p. wurmbii* subspecies is the most abundant, with 31,436–44,995 individuals living in some 6.5 million ha in 2016. Again, this seems not an immediate risk of extinction, but delving beneath the numbers gives cause for concern. We estimate that only 32–38% of these (11,891–14,399 individuals) occur in protected areas, although if we include privately protected populations within ecosystem restoration concessions this figure rises to 42–49%. That means that over half of this subspecies lives in forests that are either designated for conversion or allocated for timber extraction. Separately, over half of this subspecies, 57–60% (18,858–25,549 individuals) – and four of the six largest remaining populations – exist in areas that are predominantly peat-swamp forest habitat.

It is when we overlap these two factors that the true nature of the risk is revealed. For protected populations of *P. p. wurmbii*, 88–90% are in peat-swamp forest, a habitat which we know to be at huge risk in the future. Just 1,532–2,250 orangutans in this subspecies – a mere 5% of this taxon – are adequately protected in conservation forests in dryland habitats. This is the dichotomy that exists within orangutan conservation. We are disproportionately protecting peat-swamps from conversion, but this habitat appears destined to be lost to fire and climate change, whereas the dryland habitats that are most robust are being converted and orangutans are being hunted at unsustainable rates.

For the other two subspecies, the prognosis is slightly better. For *P. p. pygmaeus*, 87% of the 5,725 individuals

remaining are in protected forests, and just 9% of these are in peatland habitat, giving 4,553 protected individuals in non-peat habitat. For *P. p. morio*, 53–55% (5,154–12,257) live in protected areas, and nearly all of this is dryland habitat (although this is also susceptible to fire in eastern Borneo).

So, what is the hope for the future? Improving protection of peat-swamp forests through successful fire prevention strategies, rigorous law enforcement and landscape-level restoration is essential, but this looks a long way off under the current paradigm. The best long-term safety net for the Bornean orangutan is maintaining populations in dryland habitat that is either formally protected or under timber concession management, and placing focus on creating new protected areas in areas with dryland habitat. Reduced-impact logging following sustainable timber extraction practices has a less dramatic effect on orangutans as shown in Sabah (Ancrenaz *et al.* 2010) and in Kalimantan (Utami-Atmoko *et al.* 2017) if hunting is absent and if timber concessions can be kept safe from plantation development. Some orangutans use acacia and oil palm plantations as part of the larger forest landscape in which they exist, therefore maintaining corridors and preventing persecution within plantations is crucial for their survival (Meijaard *et al.* 2010; Ancrenaz *et al.* 2015; Spehar and Rayadin 2017). Improving spatial planning by restricting plantation development in orangutan habitat is a long-held goal, of which a recent Indonesian government moratorium preventing development of peatland habitat is one example. Finally, the peat-swamp forests won't be lost overnight. Although there are huge challenges involved in restoring thousands of square kilometres of drained, logged and partly-burnt peatland – challenges that have not been met before and are made even harder by climate change projections – the importance of protecting this carbon store is globally recognised and hope remains that the necessary political will and financial support will be made available to protect and restore this critical habitat.

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Neotropics



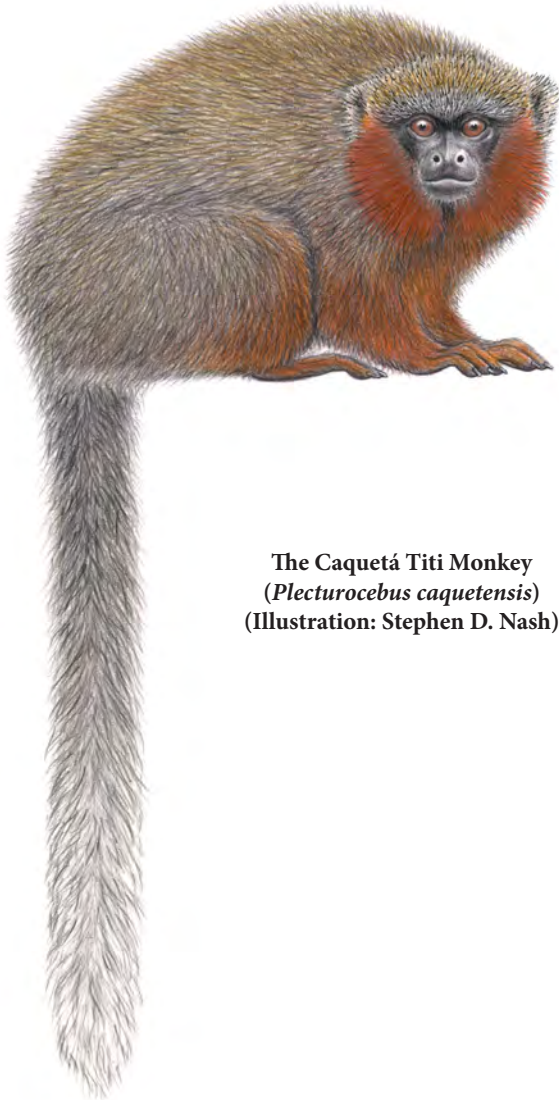
Neotropical Primates

- | | |
|---|--|
|  <i>Alouatta guariba guariba</i> |  <i>Cebus kaapori</i> |
|  <i>Ateles fusciceps</i> |  <i>Plecturocebus caquetensis</i> |
|  <i>Ateles geoffroyi</i> | |



Caquetá Titi Monkey
Plecturocebus caquetensis (Defler, Bueno & Garcia, 2010)
Colombia
(2016)

Thomas R. Defler, Javier García & Diana C. Guzmán-Caro



The Caquetá Titi Monkey
(Plecturocebus caquetensis)
(Illustration: Stephen D. Nash)

Endangered because of its restricted distribution and high rates of habitat loss throughout its range (Defler and Garcia 2012).

Plecturocebus caquetensis occurs only in the departments of Caquetá and Cauca in the Andean foothills of the southern Colombian Amazon, between the Orteguaza and Caquetá rivers (Defler *et al.* 2010; Garcia *et al.* 2010; Garcia and Defler 2011). It was recently recorded in the Piamonte Municipality in Cauca Department (Defler *et al.* 2016). Results of recent surveys in the south-western Colombian Amazon suggest that *P. caquetensis* is distributed south of the Fragua Chorroso, Pescado and Orteguaza rivers in the interfluvial area formed with the Caquetá River to about 350 m asl (Defler *et al.* 2016). The entire range of the species is estimated to be 4,029 km², of which 1,366 km² is heavily disturbed remnant habitat and only 839 km² (just 22% of its total distribution) remains forested (Defler *et al.* 2016). Historically, the upper Caquetá River is one of the most heavily impacted areas in the Colombian Amazon (Armenteras *et al.* 2006; Etter *et al.* 2006). Studies of forest fragmentation in the range of *P. caquetensis* show that the number of forest fragments of >1,000 ha has declined, with an increase in smaller fragments surrounded by an anthropogenic matrix dominated by pasture (Garcia and Defler 2013; Defler *et al.* 2016). Since 2013, the Caquetá Department has had the highest deforestation rates in Colombia (Colombia, IDEAM 2013, 2014).

The Caquetá titi monkey (*Plecturocebus caquetensis*) was first discovered in the 1960's in Valparaiso, Caquetá Department (Moynihan 1976). The presence of armed insurgents in the area meant that its formal description could not be made, however, until nearly 50 years later (Defler 2003, 2004, 2010; Defler *et al.* 2010). Recent phylogenetic analysis validated it as a distinct taxon (Defler *et al.* 2010; Hoyos *et al.* 2016) in the *Callicebus cupreus* group (Kobayashi 1995). In their revision of the Callicebinae, Byrne *et al.* (2016) placed *C. caquetensis*, together with the *C. cupreus*, *C. moloch* and *C. donacophilus* groups in the new genus *Plecturocebus*. *Plecturocebus caquetensis* is Critically

Because of the socio-political difficulties of working in this area, little is known about the species. Its fur is largely agouti-brown, with a lighter tail and reddish undersides, neck and cheeks as well as a distinctive red 'beard' (Defler *et al.* 2010). Their groups of 3 to 4 individuals are composed of an adult pair and their offspring. They are found in the middle to lower forest strata where they subsist on a varied diet of fruits, seeds, immature leaves, other plant parts and arthropods (Acero-Murcia *et al.* submitted).

The principle threat faced by *P. caquetensis* is anthropogenic habitat loss and fragmentation, primarily for pasture (Defler 2010; Defler *et al.* 2010; Garcia *et al.* 2010; Defler and Garcia 2012; Garcia and Defler 2013; Defler *et al.* 2016). Most of the human population in this area is impoverished (Colombia, DANE 2005) and extensive cattle ranching is the principle economic activity, combined with exploitation of forest resources for subsistence (Garcia *et al.* 2010). It has been estimated that 58% of the species' range is now pasture (Defler *et al.* 2016). Another major threat to the species' habitat in this area is the cultivation of illicit crops, primarily coca (*Erythroxylum coca*) for cocaine production (Garcia *et al.* 2010). The latest reports suggest that coca cultivation along the upper Caquetá River has increased in area and intensity (Colombia, UNOCD 2016).

In recent years, the Colombian government has followed a program of incentivized development of transport infrastructure, mining and energy production. Improvements to road systems have facilitated access to previously unexplored areas in the south-western distribution of *P. caquetensis*. Government statistics indicate that 99% of the range of *P. caquetensis* is designated for oil production or is already being explored and/or exploited (Defler *et al.* 2016). Similarly, in the Bota Cauca Region and along the Caquetá River there has been a recent growth in illegal gold mining (Javier Garcia pers. obs.).

There is no evidence of the presence of *P. caquetensis* in any protected area, either public or private (Garcia and Defler 2011; Defler and Garcia 2012; Defler *et al.* 2016). Eleven forest fragments along the Ortegua and Caquetá rivers have been identified as of conservation value for this species and potential areas for the establishment of reserves (Garcia *et al.* 2010; Garcia and Defler 2013; Defler *et al.* 2016). However, only six of these are larger than 2,500 ha, the minimum area estimated to support a viable population of this species in the long term (García and Defler 2013). Even if all suggested areas received protection they would cover just over 50% of remaining primary forest and only 10% of the species range (Defler *et al.* 2016).

The recent signing of a peace deal between the Colombian government and armed groups could lead to an acceleration of deforestation and forest fragmentation from further expansion of the agricultural frontier into the range of *P. caquetensis*, as well as an expansion of infrastructure to facilitate exploitation of oil and mineral resources. But the peace deal also provides a unique

opportunity for conservation and research on this species for the first time since its discovery. A National Conservation Plan is currently being developed for *P. caquetensis* (see Defler *et al.* 2016), in which key conservation issues will be addressed with the objective of gaining the support of environmental authorities in new initiatives to study and develop conservation actions for the species.

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Brown-Headed Spider Monkey
Ateles fusciceps Gray, 1866
Ecuador, Colombia, Panamá
(2006, [2012, 2014 (*A. f. fusciceps* only)], 2016)

Diego G. Tirira, Pedro G. Méndez-Carvajal & Alba-Lucía Morales-Jiménez



The Ecuadorian Brown-Headed Spider Monkey (*Ateles fusciceps fusciceps*), left, and the Darien Black Spider Monkey (*Ateles fusciceps rufiventris*), right (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 species lives in groups of up to 35 individuals; the group may divide into subgroups in search of food, the size of which varies from 1 to 10 individuals, and under such circumstances solitary individuals may be encountered (Gavilánez-Endara 2006; Estévez-Noboa 2009; Cueva and Pozo 2010; Moscoso 2010). *Ateles fusciceps* mainly inhabits large continuous forest patches in primary or secondary forest and prefers the highest levels of the canopy. The presence of this species in certain localities may be due to suitable habitat conditions such as continuous canopy cover and high abundance of large and tall trees (Tirira *et al.* 2011).

The species' diet comprises mainly ripe fruits (70–90% of its diet; Van Roosmalen and Klein 1988; Rylands and

Mittermeier 2013); its preferred fruit is generally hard with large piths, including various palms, but it also eats soft, small and multi-seeded fruits, mainly belonging to trees of the families Burseraceae, Caricaceae, Cecropiaceae, Clusiaceae, Fabaceae, Lauraceae, Malvaceae, Meliaceae, Moraceae, Myristicaceae, Olacaceae, Palmaceae, Rutaceae, and Sapotaceae, among others (Morelos-Juárez *et al.* 2015). When fruit is scarce, or when it wants to supplement its diet, this species eats fresh leaves, seeds, aerial roots, and some invertebrates, like insects; to a lesser degree it eats flowers, shoots and bark, fungi, decaying wood, and mud (Cuarón *et al.* 2008; Rylands and Mittermeier 2013; Morelos-Juárez *et al.* 2015). *Ateles fusciceps* is considered to be an effective disperser of seeds (Cuarón *et al.* 2008).

The brown-headed spider monkey is an active monkey that may move from several hundred meters up to three kilometers in a day. It often travels by brachiation, but may also run on all fours along thick branches. The

animals are typically shy and generally try to avoid humans. While feeding, they may sit or hang from one to three of their extremities (including the tail). The female becomes sexually mature at approximately four to five years of age, but usually does not give birth before it is seven or eight years old. The gestation period is 226 to 232 days, with the female giving birth to a single young every two to four years, and the young clings to its mother's underside for the first few months (Eisenberg 1973, 1976; Cuarón *et al.* 2008).

Defler (2004) and Defler *et al.* (2006) indicate that *A. fusciceps* is a subspecies of the Central American Spider Monkey (*Ateles geoffroyi*). However, the taxonomic status of both species was reviewed by Rylands *et al.* (2006) and Morales-Jiménez *et al.* (2015), who consider that both are valid and different species. Preliminary genetic analyses of samples from the south of Colombia and the north of Ecuador show two different monophyletic clades (Morales-Jimenez *et al.* 2015). Two subspecies are recognized:

Ecuadorian Brown-Headed Spider Monkey

Ateles fusciceps fusciceps Gray, 1866

The subspecies *Ateles fusciceps fusciceps* inhabits the Pacific Coast of Ecuador and possibly southern Colombia, in an altitudinal range of 100 to 2000 m above sea level, but usually below 1200 m (Tirira 2017). The subspecies is distributed in Ecuador from the northwestern Andes, in Esmeraldas Province to the northwest of Pichincha and Manabí Provinces, extending to the western borders of Imbabura and Carchi Provinces (Tirira 2017). The presence of *Ateles fusciceps fusciceps* in Colombia is uncertain, but it may be present south of the Mira River, in Nariño Department, southwestern Colombia (Defler 2004). It inhabits humid tropical and subtropical forest and is found in primary forest far from human habitation. The subspecies prefers the upper stratum of the forest but may be observed at mid-levels and occasionally in the understory (Tirira 2017).

Ateles fusciceps 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 (Ecuador, 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. Some recent observations have been made in the Los Bancos area, within Pichincha Province (Moscoso *et al.* 2011; S. Shanee, unpubl. data), and in the Flavio Alfaro area, in the northwest of Manabí Province (Cervera and Griffith 2016); nevertheless, it is uncertain if these two populations are connected with other subpopulations 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 that includes three small protected areas (El Pambilar Wildlife Reserve, Río Canandé Protected Forest, and Tesoro Escondido Protected Forest), and some surrounding unprotected forests; and the Awa Ethnic Forest Reserve, north of the Mira River and close to the Colombian border, and the buffer surrounding this reserve (Tirira *et al.* 2011). The buffer area of the Cotacachi Cayapas Ecological Reserve, especially de Tesoro Escondido Protected Forest within the Río Canandé area, is possibly the area that is harboring 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 0.2–13.2 individuals/km² (Madden and Albuja 1989; Gavilánez-Endara 2006; Cueva 2008; Estévez-Noboa 2009; Cueva and Pozo 2010; Moscoso 2010; Fuentes *et al.* 2015). It is estimated that there are less than 280 individuals surviving in the forests of Ecuador, and no subpopulation bigger than 50 individuals has been found.

Darien Black Spider Monkey

Ateles fusciceps rufiventris (Sclater, 1871)

This subspecies is restricted to eastern Panama and western Colombia, in an altitudinal range from sea

level to 2000–2500 m on the slopes of the Cordillera Occidental of the Andes (Defler 2004). They range from the Panamanian Darien and Bayano, in Panama (northern limit), including the central and southern areas of the Darien, as well as its flatlands and mountain chain, to Colombia, in the Urabá region in northwestern Antioquia, north through Córdoba, Sucre, and north Bolívar departments (northern distributional limit on the south bank of the Canal del Dique, Cartagena) and west of the Río Cauca to the coast. They also range east to the lower Río Cauca along the west bank to south-central Antioquia (the Cerro Pirre or the Río Tucutí marks the border with *Ateles geoffroyi griseus*), and south to the Cordillera Occidental of the Andes in southwestern Colombia, except La Serranía del Baudó, Chocó Department; the most southerly record is Barabacoas, Nariño Department (Defler 2004; Rylands and Mittermeier 2013; Morales-Jimenez 2005). This subspecies inhabits dry, humid, and cloud forest, and semi-deciduous and deciduous tropical, subtropical, and temperate forest, occupying the greatest range of forest habitats of any Colombian spider monkey (Defler 2004; Rylands and Mittermeier 2013).

Ateles fusciceps rufiventris is Critically Endangered on the IUCN Red list (Cuarón *et al.* 2008), and Endangered according to the *Libro Rojo de los Mamíferos de Colombia* (Defler *et al.* 2006), due to habitat loss and hunting (Rylands *et al.* 2006; Méndez-Carvajal 2012). The main threat the species faces is hunting as this is one of the species targeted for consumption (Defler 2004; Defler *et al.* 2006). In Colombia it has been wiped out in the northern part of the distribution area, in the departments of Bolívar, Sucre and Córdoba, due to habitat destruction (Miller *et al.* 2004). According to distribution models and analysis of the remnant habitat there is potential habitat left in the Chocó, Valle, Cauca, and Nariño departments. In Panama, the Darien forest has been severely fragmented in both legal and illegal ways. The desire of governments and business people of Panama and Colombia to complete the Panamerican highway, connecting both countries, has caused significant pressure to open the Darien forest gap. In addition, 50 ha of illegal oil palm plantations have been found inside the Protected Forest Alto Darien in Chupanuno, Boca de Cupe, near the buffer area of the Darien National Park.

Ateles fusciceps rufiventris is protected in 11% of the remnant area in seven national parks in Colombia

(Los Katios, Paramillo, Las Orquideas, Utria, Tatama, Los Farallones de Cali, and Munchique) (Morales-Jiménez 2005). There have also been sightings in the Reserva Natural Titi Cabeza de Algodón, located in the Municipio de Carmen del Darién (Chocó Department). However, hunting is allowed for local communities living within and around the national parks (Morales-Jiménez 2005).

No conservation projects have been established to date for this subspecies, and there is no basic information on population densities and the effects of hunting on the current populations in Colombia. In Panama, a conservation project started in 2009 to protect a wild population of this subspecies, led by the Fundación Pro-Conservación de los Primates Panameños (FCPP). The FCPP, together with Advantage Tour Panama and Adopt Panama Rainforest, protect about 60 individuals of *A. f. rufiventris* at the Chucanti Natural Reserve (3.5 km²), with subgroup structures of 2–4 individuals/subgroup, with a presence of 2.9 groups/km², and a density of 9.3 individuals/km² (Méndez-Carvajal 2012). The organizations are also conducting environmental education and tourism to promote the conservation of this subspecies, with the help of international and national visitors (Méndez-Carvajal 2014). There is a long-term mammal diversity monitoring project led by FCPP, calculating densities and circadian activity of *A. f. rufiventris* using camera traps. Data have been collected for remote places such as Situro, Cruce de Mono, Cana, Pirre, and Cocalito, among others, confirming low densities or absences for this primate. The hunting pressure from the Embera-Wounaan indigenous people is causing a dramatic decline in the subspecies' population. Most of the wild population of this subspecies has remained in the highlands, since lowland forest is increasingly fragmented for cattle ranches, and the migration of cattlemen from the Azuero Peninsula has influenced logging activities. On the Colombian side, this primate is also facing fragmentation and the population is declining (Defler 2004).

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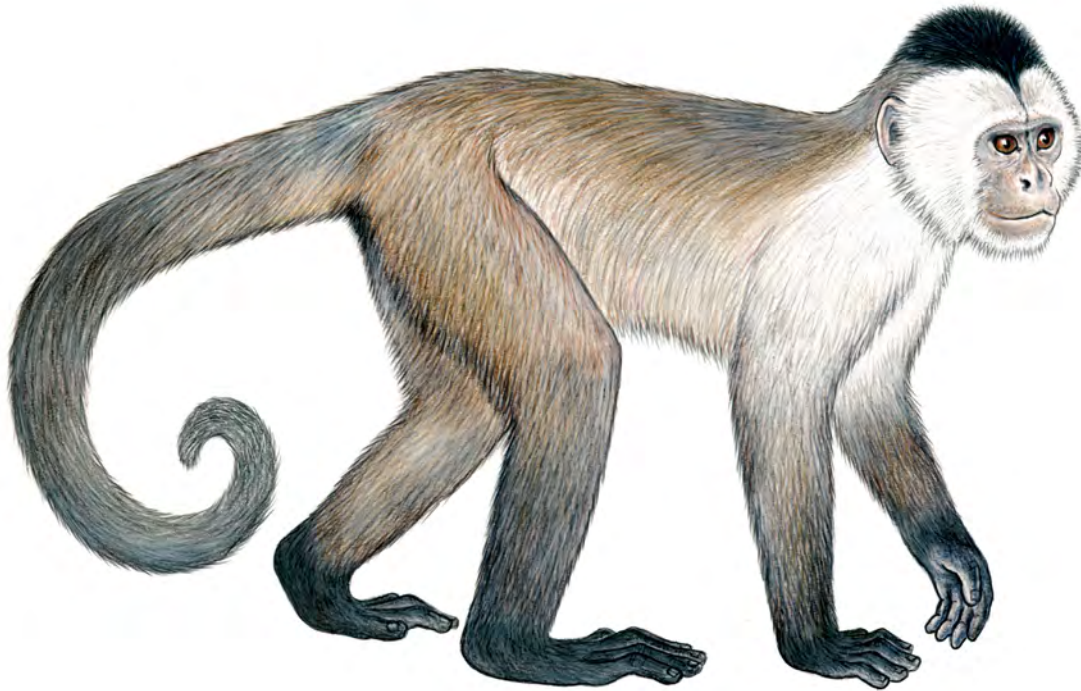
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Ka'apor Capuchin
Cebus kaapori Queiroz, 1992
Brazil
(2012, 2014, 2016)

Gerson Buss, Helder Queiroz, Fabiano R. Melo, Maurício Talebi & Leandro Jerusalinsky



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

The Ka'apor capuchin (*Cebus kaapori*), first described just 25 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 usually been observed in tall lowland terra firma forest, generally below 300 m above sea level, but has also been recorded in seasonally flooded forests and secondary forest (Silva Jr. *et al.* 2009; Silva Jr. *et al.* 2010; 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 conversion 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 Gurupi Biological Reserve has been logged and destroyed since its creation in 1988, and other pressures such as wildlife trafficking, drugs, plantations, and disputes over land ownership 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 Gurupi Biological Reserve, which translates to 0.06 groups/10 km. Recently, Buss *et al.* (2014) found 0.25 groups/10 km, in 320 km, at the same Gurupi Biological Reserve.

Results of a Population Viability Analysis with Vortex software indicated that only three populations can be considered viable over the next 100 years (Campos 2009). These populations are found in a complex of Indigenous Areas in Maranhão State (Caru, Awá, Alto Turiaçu, Araraibóia), and the Alto Rio Guama Indigenous Area in Pará State, besides the Gurupi Biological Reserve along the border between the two states.

Due to the threats of habitat loss and hunting, and a drastic population reduction (more than 80% over the past three generations or 48 years), *C. kaapori* is classified as Critically Endangered on the IUCN Red List (Fialho *et al.* 2017), the same category it received in the national assessment of Brazil (Brazil, MMA 2014; 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 Kaapor 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 found in only a few zoological institutions, such as the Centro de Primatologia do Rio de Janeiro (CPRJ) and Fundação Parque Zoológico de São Paulo (Marcos Fialho unpubl. data). Guajá Indians keep them as pets (Queiroz 1992). A recent initiative was carried out to establish a captive colony for an in-depth study of their reproductive biology and physiology in the Veterinary Department of Pará Federal University, at Castanhal, Pará State (Sheyla Domingues, pers. comm.).

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 (Buss *et al.* 2014). Partial results show that this species has a healthy population found in the reserve, despite anthropogenic pressures affecting the area. In

2015 (after these surveys), the reserve and contiguous indigenous areas were seriously affected by forest fires. The Chico Mendes Institute for Biodiversity Conservation (ICMBio) estimated that 133,000 ha of the Gurupi Reserve were impacted (Buss *et al.* 2017), which undoubtedly reduced the available habitat for *C. kaapori*. A monitoring program has been implemented by ICMBio, and further studies will be conducted in the coming years to assess the species’ situation in the area.

Field research is also currently ongoing, conducting long-term monitoring of the natural abundance of this species living in private forest remnants in eastern Pará State (Ana Cristina Mendes de Oliveira, pers. comm.) and developing occupancy models to understand habitat use and threats for the survival of fragmented subpopulations (Leticia Braga Gomes, unpubl. data).

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Central American Spider Monkey

Ateles geoffroyi Kuhl, 1820

Mexico, Guatemala, Nicaragua, Honduras, El Salvador, Costa Rica, Panama
(2016)

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Central American spider monkeys *Ateles geoffroyi* are distributed in Mexico, Guatemala, Nicaragua, Honduras, El Salvador, Costa Rica and Panama (Rylands *et al.* 2006). They are considered to be Critically Endangered because of habitat loss and fragmentation, and they are also heavily hunted for food and for the pet trade (Smith 2005). Kellogg and Goldman (1944) identified nine subspecies, but three have since been synonymized – *pan* Schlegel, 1876, and *yucatanensis* Kellogg and Goldman, 1944 (synonyms of *vellerosus* Gray, 1866) and *panamensis* Kellogg and Goldman, 1944 (synonym of *ornatus* Gray, 1870) (Rylands *et al.* 2006). Recent taxonomic studies using mitochondrial DNA have validated other subspecies which are mentioned here, but more information on identification and sample locations is needed to corroborate these conclusions (Ruiz-García *et al.* 2016).

The genus *Ateles* has long been considered the most threatened in the Neotropics (Mittermeier *et al.* 1989). *Ateles geoffroyi* has a long gestation period (226–232 days) compared to other Atelinae, such as *Alouatta*, *Brachyteles* and *Lagothrix* (Campbell 2000). It also spends more time foraging, compared to other Central American primates (Chapman *et al.* 1989), with 69–91% of fruit being a major dietary requirement (Campbell 2000). Spider monkeys have large home ranges and are more vulnerable than *Alouatta* when adapting to fragmented conditions (Méndez-Carvajal 2013). In addition to its ecological requirements, it is one of the main game species in indigenous regions (Smith 2005). This species possesses one of the larger distributions compared to other non-human primates in the Mesoamerican region, but it is threatened by high deforestation (McGrath 2014).

Ateles geoffroyi azuerensis (CR)

This subspecies was initially described as *Ateles azuerensis* Bole 1937, and was studied for the first time in its type locality (La Vaca, and Coto Region),

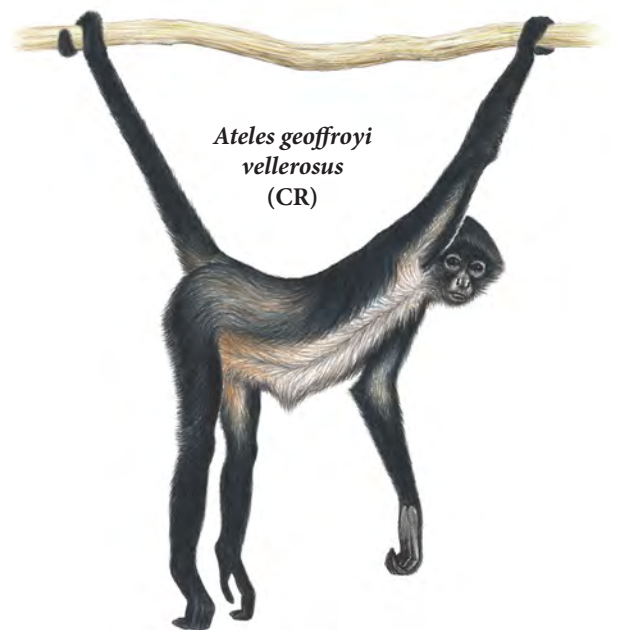
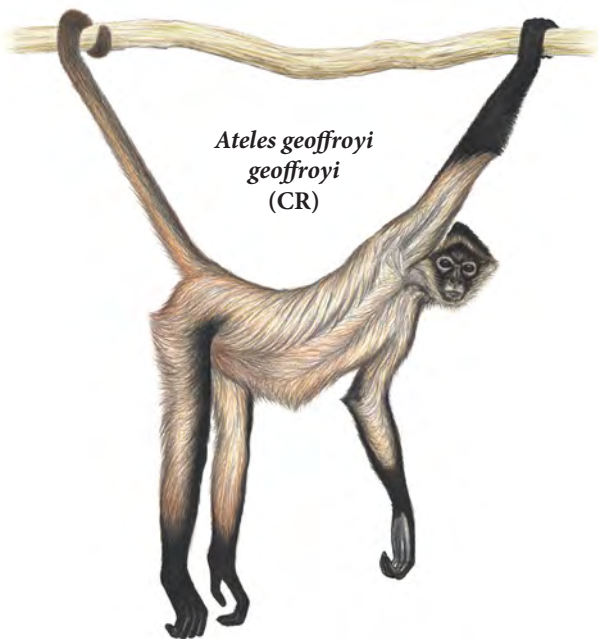
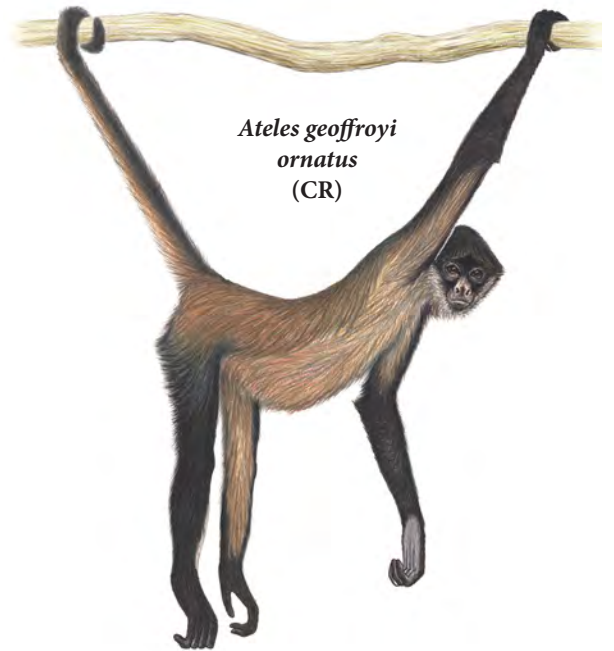
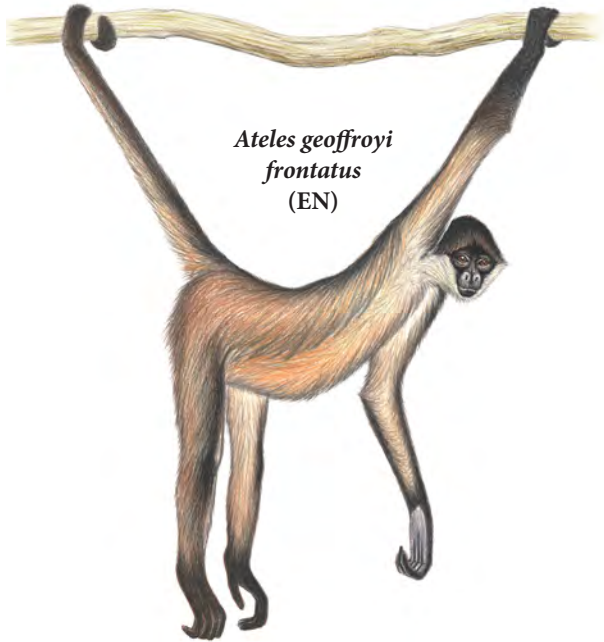
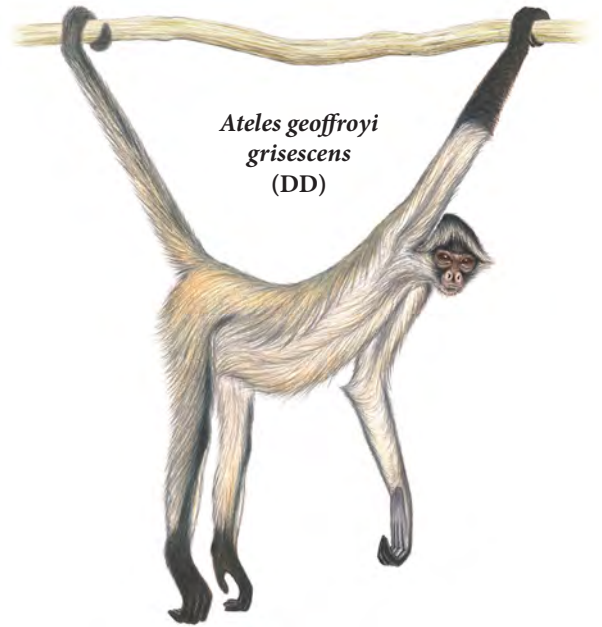
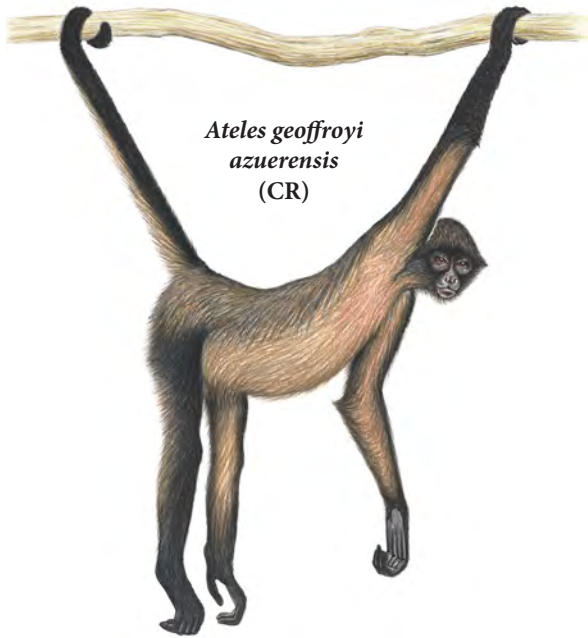
Chiriqui Province, by Carpenter 1935. The actual distribution and total population have been assessed by the Fundación Pro-Conservación de los Primates Panameños (FCPP), a Panamanian NGO that has been monitoring this primate since 2001. *Ateles g. azuerensis* has been extirpated in Chiriqui Province, west, north Veraguas and Herrera Province, and only appears to be present in the south-western Veraguas, and Los Santos Province (only on the Azuero Peninsula), in the southern areas near the Cerro Hoya National Park, and in the fragmented landscape between Punta Duarte, La Barra, Guanico, Quema, La Tronosa Forest Reserve, La Miel, and Pedasi. Only 10 subgroups and five complete groups have been detected, with a mean of 3.8 individuals/subgroup, SE ± 0.6 (range 2–7) and a mean of 12.5 individuals/group, SE ± 3.7 (range 10–22), with densities of 1.4 individuals/km² (for fragmented habitats), and an approximate total population of <150 individuals (Méndez-Carvajal and Ruiz-Bernard 2009; Méndez-Carvajal 2013). Conservation measures led by FCPP involve community volunteers from Azuero, environmental education and the creation of an educational Azuero primates guide, as well as monitoring diversity and surveying the Azuero Peninsula (Méndez-Carvajal *et al.* 2013).

Ateles geoffroyi frontatus (EN)

This primate was described by Kellogg and Goldman (1944), occurring in northern Nicaragua to the northwestern parts of Costa Rica, at Río Principolca, Tuma and Uluce. It has also been recorded in Metagalpa and the Nicaraguan highlands (Allen 1914; Cuarón *et al.* 2008).

Ateles geoffroyi geoffroyi (CR)

Kellogg and Goldman (1944) described this subspecies from San Juan del Norte, Martina Bay, southeastern Nicaragua, and it probably extends into northern Costa Rica (Cuarón *et al.* 2008).



The subspecies of the Central American Spider Monkey (*Ateles geoffroyi*) (Illustrations: Stephen D. Nash)

***Ateles geoffroyi griseus* (DD)**

Kellogg and Goldman (1944) reported this subspecies from the valley of the Río Tuira, Serranía del Sapo, Pirre, Tucuti in Darien Province, Panama (Elliot 1913; Gray 1865; Sclater 1875); it also occurs in Baudó, north-western Colombia (Cuarón *et al.* 2008). Recent studies reported that *A. g. griseus* can no longer be found in their original area (Tuira River), nor in Chucanti or the Maje Mountain Chain (Méndez-Carvajal 2012). However, the presence/absence of this primate from Panama is still in review (Méndez-Carvajal *et al.* 2016). A documentary related to the expedition to find *A. g. griseus* has been filmed by Barbara Réthoré and Julien Chapuis from Conserv-action and NatExplorers, in support of FCPP projects and the re-discovery of this subspecies.

***Ateles geoffroyi ornatus* (CR)**

This subspecies was identified for the first time in Cerro Bruja, Colon Province of Panama as *Ateles geoffroyi panamensis* by Goldman (1911, 1914). *Ateles g. panamensis* is still valid in Panama following Kellogg and Goldman (1944), Rylands *et al.* (1997) and Méndez-Carvajal *et al.* (2016). Its natural range is in Costa Rica and Panama. In Costa Rica, it is known to be in the Osa Peninsula, Carara Biological Reserve, Corcovado National Park (Matamoros and Seal 2001), and Cerro Chirripo, Cantón de Pérez Zeledón, at 1700 m asl, with a density of 0.012 individuals/km² (Rodríguez-Beitia pers. obs.). In Panama, it is present on the northern side of the Caribbean coast, in the low elevations and highlands of Bocas del Toro, the northern coast of Veraguas Province, Coclé (rare in Coclé and Donoso; Méndez-Carvajal, pers. obs.), Portobelo National Park, and San Blas mountain chain (Méndez-Carvajal *et al.* 2016). An isolated population has been introduced in Barro Colorado Island (Campbell 2000). In Panama, FCPP initiated a long-term monitoring project in 2010 in the San Blas mountain chain to understand better their actual distribution and population densities (Méndez-Carvajal 2014).

***Ateles geoffroyi vellerus* (CR)**

This subspecies is present in Mexico, Belize, Guatemala, Honduras and El Salvador (Cuarón *et al.* 2008). The population density of *A. g. vellerus* is between 2.9 individuals/km² and 9.3 individuals/km² at Montes Azules Biosphere Reserve in Marqués Comillas ejido, Chiapas, Mexico (Chaves *et al.* 2011). It also occurs in northern Veracruz, Oaxaca, Tamaulipas, Chiapas, Tabasco, Campeche, Quintana Roo and some other

regions on the Yucatan Peninsula (Chaves *et al.* 2011). It occurs in densities of 2 individuals/km² to 12 individuals/km² (Pozo-Montuy *et al.* 2015). In Guatemala, it occurs in Petén, Alta Verapaz, Baja Verapaz, Izabal, Sololá, Huehuetenango and Quiché (Ponce-Santizo *et al.* 2009). It has been reported in El Salvador at Chaguantique and El Tercio (Usulután Department), and Montecristo, Normandía, Cerro el Mono y Conchagua (Rodríguez-Menjívar 2007). *Ateles g. vellerus* is threatened by forest fires, the pet trade, habitat fragmentation due to farming activities such as oil palm, and road construction (McGrath 2014). Some conservation measures to protect this taxon include environmental education and building canopy bridges to facilitate canopy connection and reduce the number of animals killed on the roads. These activities have been implemented by the Mexican Primates Regional Monitoring System led by the project Conservación de la Biodiversidad del Usumacinta A.C. since 2013 (Pozo-Montuy *et al.* 2015). The Maya Biosphere Reserve (MBR) in the north of Guatemala, with 2.2 million ha, constitutes the largest and most important habitat for the subspecies (68.6% of its original forests). Conservation actions are maintained by several organizations to preserve this important forest block in Guatemala (Ponce-Santizo *et al.* 2009).

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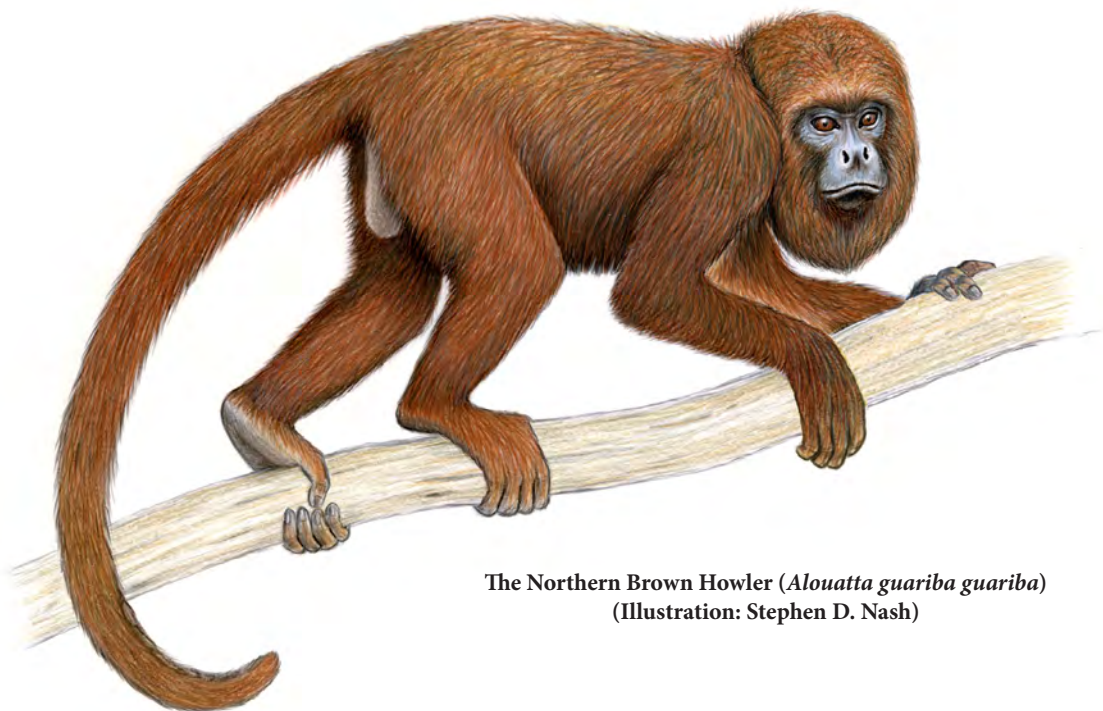
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Northern Brown Howler
Alouatta guariba guariba (Humboldt, 1812)
Brazil
(2012, 2014, 2016)

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



The Northern Brown Howler (*Alouatta guariba guariba*)
(Illustration: Stephen D. Nash)

The brown howler has two subspecies, the northern brown howler, *Alouatta guariba guariba*, and southern brown howler, *A. g. clamitans* (Rylands *et al.* 2000; Groves 2001, 2005). Studying 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), following the recommendation of Hershkovitz (1963). Rylands and Brandon-Jones (1998) argued, however, 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* north of the Doce, 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.* 2015a, 2015b; L. G. Neves, unpubl.

data). Gregorin (2006) argued that the original range of the northern brown howler in fact extended from Bahia (Rio Paraguaçu) 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, this being the case, some of the populations surveyed by Chiarello (1999) in Espírito Santo may have been of the northern subspecies *A. g. guariba*. Genetic studies are still inconclusive to support a taxonomic definition, although the karyotypic (e.g., Oliveira *et al.* 2002) and (scarce) molecular (e.g., Martins *et al.* 2011) data available suggest significant differentiation between the southern and the northern populations. Here, we maintain the names and subspecific classification as used by Rylands *et al.* (2000), Groves (2001, 2005), Glander (2013) and Cortés-Ortiz *et al.* (2015).

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 numerous plant species (Chiarello and Galetti 1994). *Alouatta guariba clamitans* is widely distributed, extending into northern Argentina, and is listed as Vulnerable on the Brazilian list of threatened species (Brazil, MMA 2014), while *A. g. guariba* has a considerably more restricted range and is classified as Critically Endangered. The primary threats are widespread forest loss and fragmentation throughout the species' range, due to logging, agriculture and cattle-ranching (Tabarelli et al. 2005), and hunting (Canale et al. 2012).

Disease epidemics, particularly yellow fever, that affect *A. guariba* and *A. caraya* (Holzmann et al. 2010; Almeida et al. 2012), are an additional and very serious threat to this taxon due to its already very reduced population. Indeed, in February 2017, a yellow fever outbreak affected several threatened primates in eastern Brazil, and we obtained strong indications that the epizootic had reached populations of *A. g. guariba* in the Jequitinhonha Valley, with at least five animals found dead in Ramaiana Farm, located in Felisburgo, Minas Gerais (André Botelho, pers. comm.). This population is close to the Mata Escura Biological Reserve, the only federal strictly protected area where *A. g. guariba* is known to occur (Melo et al. 2005).

A conservation project for *A. g. guariba* is now ongoing, following the national action plan for conservation of 27 threatened mammals of the Brazilian Atlantic Forest that includes this species (Escarlate-Tavares et al. 2016). Surveys carried out since 2012 by the Instituto de Estudos Socioambientais do Sul da Bahia (IESB), Instituto Uiraçu, the State University of Santa Cruz (UESC) and the National Center for Research and Conservation of Brazilian Primates (ICMBio/CPB), with the support of Conservation International (through the Margot Marsh Biodiversity Foundation's Primate Action Fund), the Rainforest Trust, and the Mohamed Bin Zayed Species Conservation Fund, have attempted to locate and count surviving populations, to better understand the threats to their survival and to clarify the limits to the species' geographic distribution. After three years and several field expeditions, ten populations have been

located in small and widely separated forest patches. Numbers total 27 groups and 55 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; 7) Pouso Alegre – one group, two individuals; 8) Itambé – two groups, vocalizations only; 9) Boa Nova – one group, two individuals; 10) Guaratinga – two groups, vocalizations only; 11) Santo Antônio do Jacinto – faeces only; 12) Sooretama – four groups, 10 individuals; 13) Jequitinhonha – three groups, three individuals; 14) Felisburgo – two groups, four 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 *cabruças* (an agroforestry system of cacao shaded with native trees) in southern Bahia, they have been largely hunted out. The most recent expedition to this region took place in July 2015 and confirmed a reduction in the geographic distribution of the taxon that now ranges north only to the Boa Nova municipality, which is about 200 km south from the Rio Paraguaçu, the historical 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 it coexists with the also Critically Endangered *Brachyteles hypoxanthus* and the Endangered *Sapajus xanthosternos* (see Melo 2005). This reserve is being constantly impacted by nearby rural settlements and *quilombolas* (ethnic groups, generally Afro-descendants, ancestors of refugee slaves) due to fires, logging, hunting, and disputes over land ownership. Adding the locations in the lower reaches of the 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. 2015a, 2015b; Neves et al. 2017). 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). Although surveys have been completed along the Rio Doce and Rio Paraguaçu valleys, future field expeditions will attempt to fill in the current gaps in the distribution, and especially to elucidate the western limits.

Two confiscated pets (a male and a female) have been successfully released into the Serra Bonita Private Reserve, Camacan, Bahia, owned by Vitor Becker, and managed by the NGO Instituto Uiraçú—an incipient reintroduction of the species that has not been seen or heard there for more than 50 years. There is also a promising initiative in the extreme south of Bahia in the Pau-Brasil National Park (19,027 ha, created in 1999), and surrounding private reserves. The project is led by researchers from the Federal University of São Paulo-Diadema, the park administrator and local stakeholders of the corporate sector. The aim is to study the geographical distribution and demographic parameters to establish the species as a conservation flagship for the region as well as to enable private stakeholders to contribute and become involved in the initiative.

Overall, the main conservation threats to the wild populations are hunting, habitat fragmentation and the very small sizes of the scattered populations. Now yellow fever, currently affecting howler populations throughout the south-east of the Atlantic forest presents an additional and extremely serious threat. Future measures for their protection will require a metapopulation management plan, which will include translocation to consolidate and reinforce the remnant populations.

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