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Front cover: Adult male Bioko drill *Mandrillus leucophaeus poensis*, a subspecies endemic to Bioko Island, Equatorial Guinea. This is an Endangered species and one of seven species of monkeys on Bioko Island. The animal in the photograph, Moka Boi, is housed at the Drill Rehabilitation & Breeding Center, Nigeria. Moka Boi's estimated age was 18 years when this photograph was taken in 2007. In captivity, male drills typically die of 'old age' at 16 to 19 years. Moka Boi was still alive as of October 2009. He was vasectomized in adolescence to enable him to enjoy a full life as part of a social group of mainland drills *M. l. leucophaeus* (E. L. Gadsby pers. comm.). Photograph by Jill Marty. See page 99.

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Illustrations by Stephen D. Nash

Introduction

Here we report on the fifth iteration of the biennial listing of a consensus of 25 primate species considered to be amongst the most endangered worldwide and the most in need of urgent conservation measures. The first was drawn up in 2000 by the IUCN/SSC Primate Specialist Group, together with Conservation International (Mittermeier *et al.* 2000). The list was subsequently reviewed and updated in 2002 during an open meeting held during the 19th Congress of the International Primatological Society (IPS) in Beijing, China (Mittermeier *et al.* 2002). That occasion provided for debate among primatologists working in the field who had first-hand knowledge of the causes of threats to primates, both in general and in particular with the species or communities they study. The meeting and the review of the list of the World's 25 Most Endangered Primates resulted in its official endorsement by the IPS, and became as such a combined endeavor of the Primate Specialist Group, the IPS, and Conservation International. A third revision was carried out at a meeting in August 2004, at the 20th Congress of the IPS in Torino, Italy (Mittermeier *et al.* 2006). The fourth, covering

the biennium 2006–2008, was the result of a meeting held during the 21st Congress of the International Primatological Society (IPS), in Entebbe, Uganda, 26–30 June 2006 (Mittermeier *et al.* 2007).

The list of the world's 25 most endangered primates for the biennium 2008–2010 was drawn up at an open meeting held during the 22nd Congress of the International Primatological Society, Edinburgh, UK, 3–8 August 2008. Our most sincere thanks to the organizers of the congress: Paul Honess (University of Oxford), Phyllis Lee (Stirling University), Hannah Buchanan-Smith (Stirling University), Ann MacLarnon (Roehampton University), and William Sellers (Manchester University).

As was the case for the 2004–2006 report, the texts for each species—reporting on their conservation status and threats—have counted on the extraordinary collaboration and expertise of those who know most about them; 85 contributors in all. We are most grateful for their time and dedication. Their contributions guarantee the authority of this report in describing the reasons why these primates are

in such danger, and we hope it will be effective in drawing attention to the plight of each and in garnering support for the appropriate concern and action by those who can contribute to saving them.

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The World's 25 Most Endangered Primates: 2008–2010

The 2008–2010 list of the world's 25 most endangered primates has five species from Madagascar, six from Africa, 11 from Asia, and three from the Neotropics—five lemurs, a galago and the recently described kipunji from Tanzania, two red colobus monkeys, the roloway monkey, a tarsier, a slow loris from Java, four langurs (the pig-tailed langur from Indonesia, two so-called karst species from Vietnam, and the purple-faced langur from Sri Lanka), the Tonkin snub-nosed langur and the gray-shanked douc, both from Vietnam, the cotton-top tamarin and the variegated spider monkey from Colombia (the latter also from Venezuela), the Peruvian yellow-tailed woolly monkey, two gibbons (one from China/Vietnam, the other from India, Bangladesh and Myanmar) and two of the great apes (the Sumatran orangutan and the Cross River gorilla from Nigeria and Cameroon).

The changes in the list compared to the previous one of 2006–2008 (see Tables 3 and 4) were not because the situation of the six species dropped has improved; unfortunately, far from it. Most of the changes were made so as to highlight other closely related species, which are also in dire straits regarding prospects for their future survival.

Lepilemur sahamalazensis was replaced by *Lepilemur septentrionalis*. Both are from the northernmost parts of Madagascar, both have minute populations in tiny, tiny geographic ranges, and both suffer from hunting pressure and habitat loss.

Table 1. The World's 25 Most Endangered Primates 2008–2010.

Madagascar		
<i>Prolemur simus</i>	Greater bamboo lemur	Madagascar
<i>Eulemur cinereiceps</i>	Gray-headed lemur	Madagascar
<i>Eulemur flavifrons</i>	Sclater's lemur	Madagascar
<i>Lepilemur septentrionalis</i>	Northern sportive lemur	Madagascar
<i>Propithecus candidus</i>	Silky sifaka	Madagascar
Africa		
<i>Galagoides rondoensis</i>	Rondo dwarf galago	Tanzania
<i>Cercopithecus diana roloway</i>	Roloway monkey	Côte d'Ivoire, Ghana
<i>Procolobus rufomitratus</i>	Tana River red colobus	Kenya
<i>Procolobus epieni</i>	Niger Delta red colobus	Nigeria
<i>Rungwecebus kipunji</i>	Kipunji	Tanzania
<i>Gorilla gorilla diehli</i>	Cross River gorilla	Cameroon, Nigeria
Asia		
<i>Tarsius tumpara</i>	Siau Island tarsier	Indonesia (Siau Is.)
<i>Nycticebus javanicus</i>	Javan slow loris	Indonesia (Java)
<i>Simias concolor</i>	Pig-tailed langur	Indonesia (Mentawai Is.)
<i>Trachypithecus delacouri</i>	Delacour's langur	Vietnam
<i>Trachypithecus p. poliocephalus</i>	Golden-headed or Cat Ba Langur	Vietnam
<i>Semnopithecus vetulus nestor</i>	Western purple-faced langur	Sri Lanka
<i>Pygathrix cinerea</i>	Gray-shanked douc	Vietnam
<i>Rhinopithecus avunculus</i>	Tonkin snub-nosed monkey	Vietnam
<i>Nomascus nasutus</i>	Cao Vit or eastern black-crested gibbon	China, Vietnam
<i>Hoolock hoolock</i>	Western hoolock gibbon	Bangladesh, India, Myanmar
<i>Pongo abelii</i>	Sumatran orangutan	Indonesia (Sumatra)
Neotropics		
<i>Saguinus oedipus</i>	Cotton-top tamarin	Colombia
<i>Ateles hybridus</i>	Variegated spider monkey	Colombia, Venezuela
<i>Oreonax flavicauda</i>	Peruvian yellow-tailed woolly monkey	Peru

Loris tardigradus nycticeboides from Sri Lanka (2004 and 2006) was replaced by the Javan slow loris, representing a crisis threatening all the Asian lorises. The massive and crushing trade in them for pets and for commerce in traditional medicines, compounded by widespread forest loss, is causing their rapid decline. The Javan slow loris, representing the plight of all, is evidently the hardest hit of any of the lorisiformes in this respect.

The 2008 IUCN Red List of Threatened Species recognizes 19 red colobus monkeys (*Procolobus*). Five were Not Evaluated (NE), two were ranked as Near Threatened (NT), seven were ranked as Endangered (EN), and three were ranked as Critically Endangered (CR). Only one of the red colobus monkeys, *Procolobus rufomitratus oustaleti*, from Central Africa, north of the River Congo, was ranked as of Least Concern (LC). These colobus monkeys are particularly susceptible to hunting—the widespread and insidious bushmeat trade—and also suffer from forest loss and fragmentation. Four red colobus monkeys have been listed over the five iterations of this list since 2000: *Procolobus badius waldroni*,

Procolobus p. pennantii and *Procolobus epieni*, from the widely destroyed, fragmented and hunted forests of West Africa, and *Procolobus r. rufomitratus* from the few small gallery forest patches remaining along the Tana River in Kenya. The Tana River red colobus has been on the list since 2002. The Niger Delta red colobus, first discovered only in 1993, was placed on the list in this biennium 2008–2010 because its range is very small, it suffers from bushmeat hunting, and there is widespread degradation of the Niger Delta's forests; there is every reason to suspect that its numbers are declining.

The Hainan gibbon, *Nomascus hainanus*, was taken off the list, despite the fact the world population of this species numbers less than 20 individuals. Considerable efforts are now underway to protect this species. The closely related eastern black crested gibbon, however, is also extremely threatened. It occurs in a very small region on the Vietnam/China border and numbers are estimated at around 100 in just 18 groups. The remaining few forest patches where it still survives are being destroyed (charcoal, firewood, and clearance for agriculture and pasture).

Table 2. The world's 25 most endangered primates 2008–2010 are spread through 17 countries. Those which stand out are Madagascar (five species), Vietnam (five species), and Indonesia (four species).

Madagascar	
Madagascar	<i>Prolemur simus</i> , <i>Eulemur cinereiceps</i> , <i>Eulemur flavifrons</i> , <i>Lepilemur septentrionalis</i> , <i>Propithecus candidus</i>
Africa	
Cameroon	<i>Gorilla gorilla diehli</i>
Côte d'Ivoire	<i>Cercopithecus diana roloway</i>
Ghana	<i>Cercopithecus diana roloway</i>
Kenya	<i>Procolobus rufomitratus</i>
Nigeria	<i>Procolobus epieni</i> , <i>Gorilla gorilla diehli</i>
Tanzania	<i>Galagoides rondoensis</i> , <i>Rungwecebus kipunji</i>
Asia	
Bangladesh	<i>Hoolock hoolock</i>
China	<i>Nomascus nasutus</i>
India	<i>Hoolock hoolock</i>
Indonesia	<i>Tarsius tumpara</i> , <i>Nycticebus javanicus</i> , <i>Simias concolor</i> , <i>Pongo abelii</i>
Myanmar	<i>Hoolock hoolock</i>
Sri Lanka	<i>Semnopithecus vetulus nestor</i>
Vietnam	<i>Trachypithecus delacouri</i> , <i>Trachypithecus p. poliocephalus</i> , <i>Pygathrix cinerea</i> , <i>Rhinopithecus avunculus</i> , <i>Nomascus nasutus</i>
Neotropical Region	
Colombia	<i>Saguinus oedipus</i> , <i>Ateles hybridus</i>
Venezuela	<i>Ateles hybridus</i>
Peru	<i>Oreonax flavicauda</i>

Table 3. The following primates included on the 2006–2008 list were removed from the 2008–2010 list.

Madagascar	
<i>Lepilemur sahamalazensis</i>	Sahamalaza sportive lemur
Africa	
<i>Procolobus pennantii pennantii</i>	Pennant's red colobus
<i>Procolobus badius waldroni</i>	Waldron's red colobus
Asia	
<i>Loris tardigradus nycticeboides</i>	Horton Plains slender loris
<i>Nomascus hainanus</i>	Hainan gibbon
Neotropics	
<i>Ateles fusciceps</i>	Brown-headed spider monkey

Table 4. The following six primates were placed on the list for the first time.

Madagascar	
<i>Eulemur flavifrons</i>	Sclater's lemur
<i>Lepilemur septentrionalis</i>	Northern sportive lemur
Africa	
<i>Procolobus epieni</i>	Niger Delta red colobus
Asia	
<i>Nycticebus javanicus</i>	Javan slow loris
<i>Nomascus nasutus</i>	Cao Vit crested gibbon
Neotropical Region	
<i>Saguinus oedipus</i>	Cotton-top tamarin

Of the two remaining species on the list for the first time, one was at the expense of one of the three red colobus monkeys, and the other at the expense of the Ecuadorean spider monkey, *Ateles fusciceps*, both on the 2006–2008 list. The loss of *A. fusciceps* of the Chocó region of Ecuador was due to the lack of a spokesperson on its behalf. The addition of the cotton-top tamarin, *Saguinus oedipus*, endemic to northern Colombia, was due to a recent distribution-wide survey of the species that had revealed a highly fragmented and severely diminished population, with even the few small protected areas where it occurs suffering extensive forest loss. Sclater's lemur, *Eulemur flavifrons*, is one of the least-studied of all *Eulemur* species. The single population that occurs on the Sahamalaza Peninsula of Madagascar is undergoing a very rapid decline because of hunting and trapping and the destruction of its forests due to slash-and-burn agriculture and selective logging.

New Species

Five of the world's 25 most endangered primates are species only recently described: the Rondo dwarf galago (*Galagoides rondoensis*) by Paul Honess in Kingdon (1997); the gray-shanked douc (*Pygathrix cinerea*) by Tilo Nadler in 1997; the Niger Delta red colobus (*Procolobus epieni*) by Peter Grubb and C. Bruce Powell in 1999; the kipunji (*Rungwecebus kipunji*) by Carolyn Ehardt and colleagues in Jones *et al.* (2005); and the Siau Island tarsier (*Tarsius tumpara*) that was first described by Myron Shekelle and colleagues in 2008. **Eighty-six primates—species and subspecies—**have been described since 1990; 47 from Madagascar, 10 from Africa, 11 from Asia, and 17 from the Neotropics (statistic current 25 July 2009). Fifty-four of the primates described since 1990 are prosimians, and 32 are monkeys. Many of these new primates have very restricted distributions (one of the reasons they were not discovered sooner) and some are known only from their type localities. With more information becoming available it is possible to predict that many will be future candidates for this list.

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Threats

The 2008 IUCN Red List of Threatened Species assessed the status of 634 primate taxa. Of these, 303 (47.8%) were ranked as threatened (Vulnerable, Endangered or Critically Endangered); 37% of the African primates, 43% of the lemurs, 71% of the Asian primates, and 40% of the Neotropical primates.

Nearly half of all the world's primates are threatened; principally due to habitat loss and hunting. In the face of habitat degradation and loss, factors which determine more precisely the status of each primate taxon include the following: the size of the geographic range of the taxon (extent of occurrence), the area actually occupied by the taxon (area of occupancy), the pattern of habitat loss (fragmentation, including fragment size and degree of fragment isolation), the extent and form of habitat degradation (for example, intensive logging, light selective logging, agroforestry, firewood collection, exploitation of non-timber products, understorey damage by cattle, and edge effects depending on fragment size), and the intrinsic resilience of the taxon to fragmentation and degradation. Hunting, of course, can vary in intensity (occasional, subsistence, for local, regional or international commerce [bushmeat]) and purpose (for food, traditional medicine, talismans and potions, for bait, pets or for biomedical research). Susceptibility to hunting pressure will depend on demographic (life history) variables, on overall population size and the geographic patterns of populations (some protected by remoteness, the degree to which populations are connected [sources and sinks]), and the ease with which they can be hunted (group size and habits, and accessibility, for example).

The depredations of hunting and habitat destruction on the populations of each primate taxon are behind the two principal parameters that result in them being placed on this list of the world's 25 most endangered primates—very, very small population sizes and very rapid declines in numbers. Of the 206 primates on the 2008 IUCN Red List that are classified as Critically Endangered or Endangered, fifty-four (26%) have at some time been placed on the world's 25 most endangered list. Seven of them have been on all five of the lists since 2000: the silky sifaka (*Propithecus candidus*), four Asian colobines—Delacour's langur (*Trachypithecus delacouri*), the Cat Ba langur (*T. p. poliocephalus*), the gray-shanked douc (*Pygathrix cinerea*), and the Tonkin snub-nosed monkey (*Rhinopithecus avunculus*)—the Cross River gorilla (*Gorilla gorilla diehli*), and the Sumatran orangutan (*Pongo abelii*) (Table 6).

Table 5 provides a summary of the threats to each of the world's 25 most endangered primates 2008–2010 as identified in the species profiles in this report.

Table 5. Threats to the world's 25 most endangered primates 2008–2010 as given in the species' profiles in this report.

	Estimated population size	Threats
Madagascar		
<i>Prolemur simus</i>	Not more than 100–160	Small isolated populations, slash-and-burn agriculture, mining, illegal logging, the cutting of bamboo, hunting with slingshots, reduced availability of drinking water due to climatic change. Intrinsic: extreme dietary specialization and dependency on giant bamboo.
<i>Eulemur cinereiceps</i>	7,265 ± 2,268	Very small range (c. 700 km ²), hybridization with <i>E. rufifrons</i> , low population densities, fragmented populations (small population effects, including parasitosis), cyclones, deforestation, hunting.
<i>Eulemur flavifrons</i>	450–2,300	Very small range (c. 2,700 km ²), forest loss, (slash-and-burn agriculture, selective logging), hunting and trapping and live capture for pet trade.
<i>Lepilemur septentrionalis</i>	Less than 100	Very small range, tree-felling for charcoal, hunting.
<i>Propithecus candidus</i>	100–1,000	Very small range, hunting, forest loss (slash-and-burn agriculture, selective logging, firewood).
Africa		
<i>Galagoides rondoensis</i>	-	Very small and fragmented range in remnant forest patches, loss of habitat (agricultural encroachment, charcoal production, logging).
<i>Cercopithecus diana roloway</i>	-	Hunting (bushmeat trade), forest loss, fragmented populations (numerous documented local extinctions).
<i>Procolobus rufomitrat</i>	Less than 1,000	Very small and fragmented range, forest loss (agricultural encroachment, selective logging for local use [houses, canoes]), exploitation of nontimber products, parasitosis of isolated populations.
<i>Procolobus epieni</i>	-	Very small range (c. 1,500 km ²), habitat degradation, bushmeat hunting, logging (important food trees for the species), change in hydrological regime of marsh forest due to construction of canals.
<i>Rungwecebus kipunji</i>	c. 1,117	Very small and fragmented range (area of occupancy c. 12.8 km ²), forest loss, hunting.
<i>Gorilla gorilla diehli</i>	200–300	Restricted range, agricultural encroachment, fires to clear forest or improve pasture, development activities (roads), hunting, wire snares set for other wildlife.
Asia		
<i>Tarsius tumpara</i>	Low thousands at best	Island population (active volcano), very small range (area of occupancy c. 19.4 km ²), high human density, hunting for snack food, habitat degradation.
<i>Nycticebus javanicus</i>	-	Massive trade (traditional medicine and pets), forest loss (agriculture), roads, human disturbance.
<i>Simias concolor</i>	c. 3,347	Island population, forest loss (human encroachment, product extraction, commercial logging, conversion to cash crops and oil palm plantations), hunting.
<i>Trachypithecus delacouri</i>	Less than 320	Restricted range (400–450 km ²), fragmented populations (60% occur in isolated populations of less than 20 animals), hunting (primarily for trade in bones, organs and tissues used in traditional medicine).
<i>Trachypithecus p. poliocephalus</i>	60–70	Island population (karst island of 140 km ²), seven isolated subpopulations, hunting (primarily for trade in bones, organs and tissues used in traditional medicine).
<i>Semnopithecus vetulus nestor</i>	-	Forest loss, more than 90% of forest in its range has been lost or fragmented (urbanization and agriculture), dependant on gardens for survival, electrocution (power lines), road kill, dogs, occasional hunting (for pet trade or persecution for crop-raiding).
<i>Pygathrix cinerea</i>	600–700	Restricted range and fragmented population, forest loss (agriculture logging, firewood), hunting, including use of snares.
<i>Rhinopithecus avunculus</i>	No more than 200	Restricted range and fragmented population (five isolated localities), forest loss (logging, shifting cultivation), hunting, dam construction (habitat loss and influx of thousands of people, increasing hunting pressure).
<i>Nomascus nasutus</i>	c. 110	Very small range (c. 48 km ²), habitat loss and disturbance (cultivation, pasture, firewood, charcoal production), fragmented populations (small population effects).
<i>Hoolock hoolock</i>	Less than 5,000	Recent very rapid declines in numbers, very fragmented populations (small population effects), forest loss (human encroachment, tea plantations, slash-and-burn cultivation), hunting for food and medicine, and capture for trade.
<i>Pongo abelii</i>	c. 6,600	Recent very rapid declines in numbers, restricted and fragmented range (10 fragmented habitat units), habitat conversion and fragmentation (fires, agriculture and oil palm plantations, roads, logging, encroachment), occasional killing as pests or for food, occasional pets.
Neotropics		
<i>Saguinus oedipus</i>	Less than 6,000	Forest loss and fragmentation (large-scale agricultural production [cattle] and farming, logging, oil palm plantations, hydroelectric projects), pet trade, capture for biomedical research (past).
<i>Ateles hybridus</i>	-	Restricted ranges of two subspecies, low population densities, forest loss and fragmentation (agriculture, cattle-ranching), hunting, pet trade.
<i>Oreonax flavicauda</i>	-	Restricted range, low population densities in tall premontane, montane and cloud forest, forest loss (agriculture, logging, roads, colonization), hunting (food, pets, fur).

Table 6. The following table shows the five lists produced to date. The seven species shaded are those which have remained on the list since 2000.

2000	2002	2004	2006	2008
Madagascar				
<i>Haplemur aureus</i>				
<i>Haplemur griseus alaotrensis</i>				
	<i>Haplemur simus</i>	<i>Prolemur simus</i>	<i>Prolemur simus</i>	<i>Prolemur simus</i>
		<i>Eulemur albocollaris</i>	<i>Eulemur albocollaris</i>	<i>Eulemur cinereiceps</i>
				<i>Eulemur flavifrons</i>
			<i>Lepilemur sahamalazensis</i>	
				<i>Lepilemur septentrionalis</i>
<i>Propithecus perrieri</i>	<i>Propithecus perrieri</i>	<i>Propithecus perrieri</i>		
<i>Propithecus candidus</i>	<i>Propithecus candidus</i>	<i>Propithecus candidus</i>	<i>Propithecus candidus</i>	<i>Propithecus candidus</i>
<i>Propithecus tattersalli</i>				
Africa				
		<i>Galagoides</i> sp. (Mt. Rungwe galago)		
			<i>Galagoides rondoensis</i>	<i>Galagoides rondoensis</i>
	<i>Cercopithecus diana rolaway</i>		<i>Cercopithecus diana rolaway</i>	<i>Cercopithecus diana rolaway</i>
<i>Cercopithecus sclateri</i>				
<i>Mandrillus leucophaeus</i>				
	<i>Cercocebus galeritus galeritus</i>			
<i>Cercocebus galeritus sanjei</i>	<i>Cercocebus galeritus sanjei</i>	<i>Cercocebus sanjei</i>		
<i>Cercocebus atys lunulatus</i>	<i>Cercocebus atys lunulatus</i>	<i>Cercocebus atys lunulatus</i>		
			<i>Rungwecebus kipunji</i>	<i>Rungwecebus kipunji</i>
<i>Procolobus badius waldroni</i>	<i>Procolobus badius waldroni</i>		<i>Procolobus badius waldroni</i>	
		<i>Procolobus p. pennantii</i>	<i>Procolobus p. pennantii</i>	
				<i>Procolobus epieni</i>
	<i>Procolobus rufomitratus</i>	<i>Procolobus rufomitratus</i>	<i>Procolobus rufomitratus</i>	<i>Procolobus r. rufomitratus</i>
<i>Gorilla gorilla beringei</i>	<i>Gorilla b. beringei</i>	<i>Gorilla b. beringei</i>		
<i>Gorilla gorilla diehli</i>	<i>Gorilla gorilla diehli</i>	<i>Gorilla gorilla diehli</i>	<i>Gorilla gorilla diehli</i>	<i>Gorilla gorilla diehli</i>
Asia				
			<i>Tarsius tumpara</i>	<i>Tarsius tumpara</i>
		<i>Loris tardigradus nycticeboides</i>	<i>Loris tardigradus nycticeboides</i>	
				<i>Nycticebus javanicus</i>
	<i>Simias concolor</i>	<i>Simias concolor</i>	<i>Simias concolor</i>	<i>Simias concolor</i>
	<i>Presbytis natunae</i>			
<i>Trachypithecus delacouri</i>	<i>Trachypithecus delacouri</i>	<i>Trachypithecus delacouri</i>	<i>Trachypithecus delacouri</i>	<i>Trachypithecus delacouri</i>
<i>Trachypithecus poliocephalus</i>	<i>Trachypithecus poliocephalus</i>	<i>Trachypithecus p. poliocephalus</i>	<i>Trachypithecus p. poliocephalus</i>	<i>Trachypithecus p. poliocephalus</i>
	<i>Trachypithecus leucocephalus</i>			
		<i>Presbytis hosei canicrus</i>		
<i>Pygathrix nemaeus cinerea</i>	<i>Pygathrix nemaeus cinerea</i>	<i>Pygathrix nemaeus cinerea</i>	<i>Pygathrix cinerea</i>	<i>Pygathrix cinerea</i>
<i>Rhinopithecus avunculus</i>	<i>Rhinopithecus avunculus</i>	<i>Rhinopithecus avunculus</i>	<i>Rhinopithecus avunculus</i>	<i>Rhinopithecus avunculus</i>
	<i>Rhinopithecus bieti</i>			
	<i>Rhinopithecus brelichi</i>			
		<i>Semnopithecus vetulus nestor</i>	<i>Semnopithecus vetulus nestor</i>	<i>Semnopithecus vetulus nestor</i>
<i>Hylobates moloch</i>				
<i>Hylobates concolor hainanus</i>		<i>Nomascus hainanus</i>	<i>Nomascus hainanus</i>	
	<i>Nomascus nasutus</i>			<i>Nomascus nasutus</i>
			<i>Hoolock hoolock</i>	<i>Hoolock hoolock</i>
<i>Pongo abelii</i>	<i>Pongo abelii</i>	<i>Pongo abelii</i>	<i>Pongo abelii</i>	<i>Pongo abelii</i>
Neotropics				
				<i>Saguinus oedipus</i>
<i>Leontopithecus rosalia</i>				
<i>Leontopithecus chrysopygus</i>				
<i>Leontopithecus caissara</i>	<i>Leontopithecus caissara</i>	<i>Leontopithecus caissara</i>		
<i>Cebus xanthosternos</i>	<i>Cebus xanthosternos</i>	<i>Cebus xanthosternos</i>		
			<i>Ateles hybridus</i>	<i>Ateles hybridus</i>
		<i>Ateles hybridus brunneus</i>		
			<i>Ateles fusciceps</i>	
<i>Lagothrix flavicauda</i>			<i>Oreonax flavicauda</i>	<i>Oreonax flavicauda</i>
<i>Brachyteles hypoxanthus</i>	<i>Brachyteles hypoxanthus</i>	<i>Brachyteles hypoxanthus</i>		

The World's 25 Most Endangered Primates 2008–2010 Profiles

Madagascar

Greater Bamboo Lemur

Prolemur simus (Gray, 1871)

Madagascar

(2002, 2004, 2006, 2008)

The greater bamboo lemur (*Prolemur simus*) is the largest of Madagascar's bamboo-eating lemurs (Albrecht *et al.* 1990) and the most critically endangered lemur in Madagascar (Ganzhorn *et al.* 1996, 1997; Konstant *et al.* 2006; Mittermeier *et al.* 2006; Ganzhorn and Johnson 2007; Wright *et al.* 2008). Although its placement in *Prolemur* has been questioned (for example, Tattersall 2007), it now represents a monospecific genus, based on a suite of distinctive dental and chromosomal characteristics (Vuillaume-Randriamanantena *et al.* 1985; Macedonia and Stanger 1994) that support its separation from the genus *Hap Alemur* (cf. Groves 2001). Genetic studies further suggest that *Hap Alemur* may, in fact, be more closely related to the genus *Lemur* (Rumpler *et al.* 1989; Macedonia and Stanger 1994; Stanger-Hall 1997; Fausser *et al.* 2002). *Prolemur simus* also differs from other bamboo lemurs in behavioral and ecological variation.

Greater bamboo lemurs are cathemeral and gregarious, with observed group sizes ranging up to 28 individuals (Santini-Palka 1994; Tan 1999, 2000). Their extensive vocal repertoire of at least seven distinct calls is thought to be linked to their relatively large group size (Bergey and Patel 2008). It is the only male-dominant lemur species known (Tan 1999, 2000). Home ranges are large (60–97 ha; Sterling and Ramarason 1996; Tan 1999, 2000; Dolch *et al.* 2008) and are primarily influenced by the distribution of bamboo and the availability of drinking water during the dry season (Wright *et al.* 2008). The species' stark reliance on giant bamboo makes it ecologically unique among primates. Throughout its range, *P. simus* has diets consisting almost exclusively of just one species of bamboo; *Cathariostachys madagascariensis* in the north (Tan 1999, 2000; Dolch *et al.* 2008) and a lowland species in the south (Wright *et al.* 2008). *Prolemur simus* is able to manipulate live bamboo culm with specializations in its teeth and jaws (Jernvall *et al.* 2008) that allow it to strip the outside of the live stalk and consume the pith, which is especially crucial for subsistence in drier months, while it relies on its shoots and leaves at other times of the year. *Prolemur simus* supplements its diet with fruits, flowers, soil and fungi (Meier and Rumpler 1987; Tan 1999, 2000; Wright *et al.* 2008; R. Dolch, J. L. Fiely, J. Rafalimandimby, E. E. Louis Jr. unpubl. data).

Historical records (Schwarz 1931) and sub-fossil remains confirm that it was once widespread throughout the island (Godfrey and Vuillaume-Randriamanantena 1986; Wilson *et al.* 1988; Simons 1997; Godfrey *et al.* 1999, 2004). Today, *P. simus* occupies as little as 1–4% of its former range, and remaining populations are very patchily distributed. It has

only been confirmed to occur at 12 sites; all of them in the eastern rainforests. Most of them are restricted to SE Madagascar, including those in the national parks of Ranomafana (Miaranony, Talatakely, and Ambatolahy Dimy) and Andringitra (Manambolo, possibly Korokoto, and Camp 2). Five sites are located in unprotected and often degraded forests at Kianjavato, Morafeno, Karianga (near Vondrozo), Mahaso, and Evendra (near Ivato) (Meier and Rumpler 1987; Wright *et al.* 1987; Sterling and Ramarason 1996; Goodman *et al.* 2001; Irwin *et al.* 2005; Ratelolahy *et al.* 2006; Wright *et al.* 2008). Recent surveys have confirmed the species' presence in the forests of Torotorofotsy in the region of Andasibe-Mantadia (Dolch *et al.* 2004, 2008).

Wild populations occur in genetically isolated ranges with critically low numbers. Based on available data, the total wild population of *P. simus* is estimated not to exceed 100–160 individuals (Wright *et al.* 2008; R. Dolch unpubl. data). The largest populations are thought to occur in Torotorofotsy (R. Dolch, J. L. Fiely, J. Rafalimandimby, E. E. Louis Jr. unpubl. data) with up to 60 individuals, and in Ranomafana with up to 50 individuals. As of 2007, only 22 individuals of *P. simus* were held in captivity (Wright *et al.* 2008; D. Rouillet pers. comm.).

The greater bamboo lemur is threatened by slash-and-burn agriculture, mining, illegal logging, the cutting of bamboo, and hunting with slingshots (Meier 1987; Meier and Rumpler 1987; Arrigo-Nelson and Wright 2004; Dolch *et al.* 2008). Presumed causes of its decline are its extreme dietary specialization and dependency on giant bamboo. Reduced availability of drinking water due to climatic change has also been cited as a limiting factor for the species' distribution (Wright *et al.* 2008).

Prolemur simus occurs mainly outside protected areas. It has been found in two national parks, Ranomafana and Andringitra. Suitable microhabitat within these protected areas is limited, and stochastically elevated mortality has contributed to the recent decline of these groups (Wright *et al.* 2008). The recent discovery of new groups raises hopes for the survival of the species. Yet, declines in known groups have raised new concern. Efforts are underway to declare important *Prolemur* sites as protected areas, and there are plans also for Torotorofotsy, Mahaso, and Kianjavato. Conservation research projects have been initiated to study additional populations to provide behavioral, ecological, and genetic data necessary to implement an immediate large-scale conservation management plan. Moreover, microhabitat preferences of *P. simus* at known localities should be used to identify suitable habitats within the eastern rainforest, within which it is presumed other greater bamboo lemur populations could be found.

Ranomafana region. The population in and around Ranomafana National Park (RNP) is 26 individuals, with a maximum estimate of 50 individuals, a number of which live outside the park boundaries. Madagascar National Parks (former *Association Nationale pour la Gestion des Aires Protégées*—ANGAP) and Centre ValBio/Institute for the Conservation of Tropical Environments (ICTE) at Stony Brook

University have achieved long-term behavioral data on the group in Talatakely. Further research initiatives are being conducted to monitor, protect and collect data on the two subpopulations just outside the park, in addition to conducting further surveys throughout the park. The major threats to the RNP population are its small size, genetic isolation, ranging into unprotected areas and opportunistic hunting. Participating institutions active in conservation efforts in this region are Centre ValBio, Madagascar National Parks, and the *Madagascar Institut pour la Conservation des Ecosystèmes Tropicaux* (MICET).

Torotorofotsy region. One of the most recently discovered, this site has one of the largest known contiguous populations with at least 4–5 groups and up to 60 individuals. Its discovery (Dolch *et al.* 2004, 2008) extended the known range of *P. simus* 400 km north of any known populations. The Torotorofotsy groups live almost entirely outside both Torotorofotsy Ramsar site and Andasibe-Mantadia National Park, and are squeezed in between mining concessions. Only one of the known groups occurs entirely within the boundaries of the Torotorofotsy Ramsar site. Mineral exploitation (nickel, cobalt and graphite) is the most prominent threat to the Torotorofotsy population, while it also remains a discrete population, genetically isolated from other known localities by a vast distance. The Torotorofotsy population was discovered by and has since been studied by members of Association Mitsinjo, a local NGO that has subsequently grown to an organization responsible for the management and research-based conservation of the Torotorofotsy Ramsar site. The Torotorofotsy population has been continuously tracked and monitored on a daily basis since July 2007. Data collection on ranging and behavioral ecology and efforts for the formal protection of the unprotected groups are ongoing by members of Association Mitsinjo, with the support of Omaha's Henry Doorly Zoo Madagascar Biodiversity and Biogeography Project (MBP-HDZ). Association Mitsinjo leads efforts to extend the Torotorofotsy Ramsar site to include all *P. simus* groups and to make it a new protected area in its own right.

Ivato and Karianga region. This population occurs in the southeastern part of the species' range in an extremely fragmented landscape that is completely unprotected and severely threatened by habitat disturbance. ICTE and MICET have begun working in Ivato commune, concentrating on one group of *P. simus* in Mahasoia agricultural plantation, just near the village of Ivato. Thus far, this subpopulation consists of one group of 27 individuals that is restricted to a 150-ha forest fragment. While an individual was sighted on a trail between Ivato and Evendra, additional groups between Mahasoia and the corridor remain unknown, but additional surveys are underway. The project is working to protect, monitor and collect behavioral, ecological and genetic data on the known group. This area is threatened mainly by slash-and-burn agriculture and fragmentation. Conservation efforts include working with the local community on more sustainable agricultural practices, and an endemic reforestation program to connect current forest fragments to the corridor c.10 km to the west,

where other subpopulations have been sighted. ICTE and MICET are trying to implement formal protection of the area extending from Karianga/Morafeno to the corridor. Virtually no forest persists near these sites, with a landscape consisting largely of agricultural land and anthropogenic grasslands, interspersed with small, isolated bamboo patches. Current initiatives will be expanded to Karianga commune, which contains a group of at least three individuals in Morafeno agricultural plantation. Mining concessions and hunting also threaten this population, which subsists in extremely small numbers and in genetically isolated forest fragments that are being actively degraded. Participating institutions in conservation efforts in this region include ICTE, MICET and Stony Brook University.

Kianjavato. Since 1986, individuals have been observed in bamboo patches at the edge of Kianjavato coffee plantation, isolated from the eastern forest escarpment by about 50 km. This area contains at least three groups, with at least 7 individuals each, and an estimated population size of 30 individuals. The MBP-HDZ is researching the behavioral ecology and genetic composition of this population.

Corridor. Surveys within the Vondrozo corridor have been ongoing to try and find additional individuals between Ranomafana (north) and the Manapatrana River (south). Although a recent survey found two individuals between Ivato commune and Andringitra National Park (K. Delmore, unpubl.), further surveys are needed. Meanwhile, Conservation International has been working to protect the biodiversity within the remaining habitat in the corridor. Protection of this tract of intact forest will be crucial to provide a natural link between the remnant populations in south and central-eastern Madagascar. Participating institutions in the surveys in the corridor include ICTE and Centre ValBio.

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Gray-headed Lemur

Eulemur cinereiceps (Milne-Edwards and Grandider, 1880)
Madagascar
(2004, 2006, 2008)

The gray-headed lemur (*Eulemur cinereiceps*) has a complicated taxonomic history. It is closely allied with the brown lemurs (*Eulemur* spp.), particularly the neighboring collared lemur (*E. collaris*). This taxon was until recently classed as a subspecies of *Eulemur fulvus* (Tattersall 1982; Mittermeier *et al.* 1994; Pastorini *et al.* 2000). However, cytogenetic and molecular genetic analyses, as well as infertility in crosses with collared lemurs, suggest full species status (Djletati *et al.* 1997; Wyner *et al.* 1999). The name derives from plates in Milne-Edwards and Grandidier (1890) and was applied to museum specimens from the southeastern coast near Farafangana by Schwarz (1931). Groves (1974) also used this name for “white-cheeked” specimens from southeastern Madagascar, distinguishing them from *E. collaris*. Subsequently, Rumppler (1975) made a similar distinction based on karyotypes, but adopted the presumably junior synonym *E. albocollaris*

(or “white-collared lemur”). The latter nomenclature was supported by Tattersall (1979, 1982) and others, and came to prevail in the literature. It was later suggested that *E. cinereiceps* and *E. albocollaris* might represent separate taxa (Groves 2001; Mittermeier *et al.* 2006); in this scenario, *E. cinereiceps* would likely be found in coastal forests—the localities for specimens discussed by Schwarz (1931)—whereas *E. albocollaris* would be restricted to interior forests (for example, near Vondrozo). This idea was tested with available evidence from genetic sampling and population surveys (Johnson *et al.* 2008). Although not all original localities for *E. cinereiceps* could be sampled due to extensive fragmentation and lemur extirpations in this region during the last century, there is no evidence to date from mtDNA or phenotypes for a coastal-interior division. For now it seems most likely that the region contains just one species and that the name *E. cinereiceps* has priority (Johnson *et al.* 2008). Further ground surveys and genetic sampling should be conducted to confirm these findings.

The gray-headed lemur has one of the most restricted distributions of any *Eulemur* species. It occurs only in southeastern Madagascar from just north of the Manampatrana River to near the Mananara River in the south (Petter and Petter-Rousseaux 1979; Tattersall 1982; Irwin *et al.* 2005). This range includes a continuous forest corridor in the interior escarpment and small forest relicts in the coastal plain. In the north, there is a hybrid zone with *E. rufifrons* centered in Andringitra National Park, extending south to the vicinity of Karianga and north beyond Ankarimbelo (Sterling and Ramarason 1996; Wyner *et al.* 2002; Irwin *et al.* 2005). This encompasses an area of up to 50% of the range of “pure” *E. cinereiceps*. The southern boundary of the species is not well established, and could extend to Vohipaho Forest near Vangaindrano (where *E. cinereiceps* may be sympatric with *E. collaris*; H. Andriamaharoa unpubl. data). Other than Andringitra National Park, *E. cinereiceps* is only found in two protected areas: Manombo Special Reserve and the recently established conservation project at Mahabo Forest, both near Farafangana. The large Andringitra population consists almost entirely of hybrids (Wyner *et al.* 2002), whereas degraded coastal forests at Manombo and Mahabo contain only *c.* 750 *E. cinereiceps* individuals (C. Ingraldi in prep.). Population densities across the range tend to be low relative to other *Eulemur* species (Johnson and Overdorff 1999; Johnson and Wyner 2000). Recent analyses combining ground surveys and Landsat imagery indicate that the total habitat remaining within the gray-headed lemur range is approximately 700 km², with an estimated remaining population of $7,265 \pm 2,268$ individuals (Irwin *et al.* 2005).

Information regarding the natural history of the gray-headed lemur derives largely from recent studies conducted at the interior Vevembe Forest, along with new long-term studies currently underway at Manombo and Mahabo. This species has a highly frugivorous diet, supplemented with flowers, leaves, and fungi. *Pandanus* spp. flowers are an especially important food late in the dry season at Vevembe

(Johnson 2002). *Pandanus* fruit also comprises a major component of the diet at Mahabo, along with *Noronhia*, *Pyrostria*, and *Uapaca* (H. Andriamaharoa, C. Birkinshaw, A. Rued unpubl. data). At Manombo, *E. cinereiceps* has been observed eating non-native plants like *Aframomum angustifolium* and a shelf fungus that grows on invasive *Cecropia* (Ralainasolo *et al.* 2008). Feeding on such items may enable *E. cinereiceps* to cope with habitat disturbance, and perhaps in part to avoid competition with other lemurs such as *Varecia* for native plants. The species is cathemeral (active both day and night) throughout the year. It is an adept arboreal quadruped with frequent use of leaping behaviors, and its limb kinematics corresponds closely with those of *Eulemur collaris* (Stevens *et al.* in review). Social groups tend to be multi-male/multi-female and regularly exhibit fission-fusion (Overdorff and Johnson 2003; Johnson 2006). Group size may reach as many as 16 individuals (Johnson 2002). Coastal populations have smaller social groups, with apparently frequent dispersal of individuals among groups (H. Andriamaharoa, S. Martin, C. Ingraldi, A. Rued unpubl. data). Like other lemurs, reproduction is highly seasonal, although extra-seasonal copulations (with at least one birth) have been recorded at Mahabo (A. Rued in prep.).

Deforestation and hunting present the greatest threats to the survival of the gray-headed lemur. Populations in the Manombo lowland rain forest and Mahabo littoral forest are particularly vulnerable to these pressures due to the fragmentation and isolation of the coastal landscape, as well as possible small population effects. They are also susceptible to powerful stochastic climatic events: a cyclone that struck this region in 1997 reduced lemur populations by approximately 50% (Ratsimbazafy *et al.* 2002). These coastal populations have apparently undergone a significant genetic bottleneck, and effective population size (number of breeding individuals) falls well below total population estimates (R. Brenneman, E. E. Louis Jr., S. Johnson in prep.). The extensive hybrid zone with *E. f. rufus* may also pose a risk to the gray-headed lemur; research is presently being conducted to assess the direction and magnitude of gene flow across the contact zone and “pure” populations (K. Delmore in prep.). Current research is also underway to investigate disease ecology in *E. cinereiceps*. Preliminary evidence suggests heavy infestations of some parasites (for example, pinworms) that could reduce fitness, particularly if degraded environmental conditions compromise immune response (S. Martin in prep.).

The Malagasy government, conservation NGOs, and researchers are together taking steps to counter these alarming trends. The Durrell Wildlife Conservation Trust is working in partnership with Madagascar National Parks to strengthen protection at Manombo, including possible expansion of the Special Reserve. Conservation education and tree-planting programs have also been established in the Manombo communities. Missouri Botanical Garden has supported community-based initiatives to preserve Mahabo Forest within the new framework for protected areas in Madagascar; similar programs are underway at Vohipaho, which may also

maintain a small *E. cinereiceps* population. Conservation International is presently initiating programs for the management of the Fandriana-Vondrozo forest corridor. This will be critical for the long-term survival of the gray-headed lemur, as the vast majority of populations are found within this corridor and few are presently protected. Researchers from Université d'Antananarivo, University of Calgary, Stony Brook University, Henry Doorly Zoo, Ohio University, and other institutions are active in studying gray-headed lemur ecology, social systems, population dynamics, and genetics to better understand the risks and baseline requirements for this still poorly known species.

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Sclater's Black Lemur, Blue-Eyed Black Lemur

Eulemur flavifrons (Gray, 1867)

Madagascar
(2008)

The blue-eyed black lemur or Sclater's black lemur was rediscovered by science only in 1983 after more than a century of uncertainty about its existence (Koenders *et al.* 1985; Meier *et al.* 1996). Its taxonomic validity was thereafter confirmed independently by Rabarivola (1998) and Pastorini (2000). The species was until recently regarded as a subspecies of *Eulemur macaco*, but was elevated to full species status on the basis of the consistency of the morphological differences between the black lemur and the blue-eyed black lemur and the pairwise genetic distances between *macaco* and *flavifrons* of 68–72 bp (which are in the same range as between the former *E. fulvus* subspecies, i.e., 29–90 bp, according to Pastorini 2000). Furthermore, the fact that the hybrid zone between the two taxa is restricted to just the north-eastern part of the distribution of *E. flavifrons* (Andrianjakarivelo 2004; Schwitzer *et al.* 2005, 2006; Mittermeier *et al.* 2008) is in favour of this new taxonomy.

Eulemur flavifrons is still one of the least-studied of all *Eulemur* species. The species occurs only in northwest Madagascar in a very small area of about 2,700 km², south of the Andranomalaza, north of the Maevarano, and west of the Sandrakota rivers, where it inhabits primary and secondary forest fragments (Koenders *et al.* 1985; Meyers *et al.* 1989; Rabarivola *et al.* 1991; Mittermeier *et al.* 1994). The area of repartition of *Eulemur flavifrons* lies within a transition zone between the humid Sambirano region in the north and the western dry deciduous forest region in the south, harboring semi-humid forests with tree heights of up to 30 m on ferruginous alkaline and alkaline soils based on sandstone, basalt or clay (IRNT 1991a). Average annual precipitation is around 1,600 mm (IRNT 1991b).

There is only a small population of *Eulemur flavifrons* remaining, the largest part of it living in forest fragments on and adjacent to the Sahamalaza peninsula (Mouton 1999). Rakotondratsima (1999) estimates the population of the Sahamalaza peninsula to be about 450–2,300 individuals and

to have declined about 35.3% in three years (see also Andrianmanandratra 1996). Andrianjakarivelo (2004) found the mean density of *E. flavifrons* in eight inventoried forest fragments to be 24 individuals per km² (range: 4–85 ind./km²). A total count in two different fragments of the Ankarafa forest on the Sahamalaza Peninsula yielded a density of 60 individuals per km² (Schwitzer *et al.* 2005, 2007a). However, the density of the species in Ankarafa seems to be higher than in any other forest in the range of *E. flavifrons* (Randriatahina and Rabarivola 2004). Extrapolating the two density estimates of Andrianjakarivelo (2004) and Schwitzer *et al.* (2005) to the total surface of the terrestrial core zones of the recently created Sahamalaza – Iles Radama National Park (115.8 km²) yields a remaining, severely fragmented population of 2780–6950 blue-eyed black lemurs. *Eulemur flavifrons* was assessed as Critically Endangered (CR A2cd) by the International Union for Conservation of Nature (IUCN) at their most recent Red List Assessment in April 2005, based on an 80% population reduction during the last 25 years. The principal threats to its survival are forest destruction due to slash-and-burn agriculture and selective logging, continued hunting and trapping, especially by the Tsimihety in the eastern part of its distribution, and live capture for the local pet trade (Gerson 1995; Rakotondratsima 1999). Andrianjakarivelo (2004) found a density of up to 570 traps/km² in certain areas where *E. flavifrons* occurs.

The blue-eyed black lemur's home range size and use differs between primary and secondary forest fragments, indicating that it is somewhat able to adapt to different types of habitat. Larger home ranges and lower densities of *E. flavifrons* in secondary forest as compared to primary forest, however, suggest that the former is less suitable habitat for the species (Schwitzer *et al.* 2007a). During a 12-month study, *E. flavifrons* consumed parts of 72 different plant species from 35 families; 52.3% of these were fruits and 47.7% were leaves. The animals also fed on flowers, insects, insect exudates and fungi (Polowinsky and Schwitzer in press). *Eulemur flavifrons* exhibits a bimodal activity pattern, which peaks during the morning and evening twilight. It shows activity bouts during the day and night year-round. Nocturnal illumination and the proportion of illuminated lunar disc are positively associated with the amount of nocturnal activity. Total daily activity, as well as nocturnal activity, is higher in secondary forest than in primary forest (Schwitzer *et al.* 2007b).

Blue-eyed black lemur groups are multi-male multi-female, ranging in size from 6 to 10 individuals, including 4 to 7 adults (G. H. Randriatahina and J. J. Roeder in prep.). Both sexes disperse, but only males have been seen moving into a foreign social group. The sex ratio at birth varies strongly between years and could be male-biased (G. H. Randriatahina and J. J. Roeder in prep.). Births occur between late August and October, at the end of the dry season. During two successive birth seasons, infant mortality was 22.7%. Infants start to become independent at around eleven weeks of age (S. Volampeno in prep.).

Parts of the Sclater's black lemur's range officially received protected area status in June 2007 (Parc National Sahamalaza–Iles Radama), including the Sahamalaza Peninsula and some mainland forests to the north and east (Moisson *et al.* 1999; Lernould 2002; Schwitzer and Lork 2004; Schwitzer *et al.* 2006). The Sahamalaza Peninsula is also a UNESCO Biosphere Reserve. The *Association Européenne pour l'Etude et la Conservation des Lémuriens* (AEECL) is a consortium of European zoos that have joined forces to conserve Madagascar's lemurs, with the involvement of representatives of local communities from the Sahamalaza Peninsula and representatives of WCS and several other environmental institutions. AEECL implemented a natural resource management programme in Sahamalaza in December 2000 in order to protect the remaining lemur habitat and to improve the living standards of the local human population. AEECL also maintains a field station in Sahamalaza, which serves as a basis for studying the conservation ecology of *E. flavifrons* and of other lemur species in the area.

As of 2008, there were 30 blue-eyed black lemurs living in European zoos (Moisson and Prieur 2008). The European captive population of the subspecies is being managed in a European Endangered Species Programme (EEP) coordinated by Mulhouse Zoo.

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Guy H. Randriatahina, Sylviane Volampeno,
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Northern Sportive Lemur

Lepilemur septentrionalis Rumpler and Albignac, 1975
Madagascar
(2008)

The northern sportive lemur (*Lepilemur septentrionalis*) was originally described based on cytogenetic and morphometric characteristics (Rumpler and Albignac 1975, see also Rumpler *et al.* 2001). Its taxonomic status has been supported by more detailed cytogenetic, morphogenetic and especially molecular data (Rumpler *et al.* 2001; Ravaorimanana *et al.* 2004; Andriaholinirina *et al.* 2006), and accepted in recent taxonomic revisions of primates (Groves 2001, 2005) and lemurs (Mittermeier *et al.* 2006, 2008). It is believed to be strictly limited to a few small patches of dry forest in the far north of Madagascar, just to the south of Antsiranana on the east coast. Mittermeier *et al.* (2006) list the following localities: very small remnant forest patches near the villages of Madirobe and Ankarongana in the Sahafary region, and in the immediate vicinity of Andrahona, a small mountain about 30 km south of Antsiranana, east of Route Nationale 6 (observations by Yves Rumpler, Russell A. Mittermeier).

It is nocturnal, spending the day sleeping in tree holes, and very little is known about its ecology and behavior. Total numbers are unknown but, taking into account the limited distribution of *L. septentrionalis* in the forests of Sahafary, Andrahona and Andranomadiro, there are probably only about 100–150 individuals remaining. Tree-felling for charcoal continues at an alarming rate and the animals

suffer from hunting. Surveys of five areas in 2007 provided the following population estimates: 1) Area of Andrahona (forest patches and gallery forests of Andrahona, Analajana, and Analanjavavy)—20 individuals in the entire area; 2) Area of Ankarakataova (forests of Ankarakataova Be and Ankarakataova kely)—none found; and 3) Area of Sahafary (degraded forest patches in Western Sahafary, Sahafary East, Sahafary North, Andravina, Sahandrano, Andranomadiro, and Analalava)—about 100 individuals. None of these areas is protected.

The combination of a very small range containing little and rapidly decreasing suitable habitat with high pressure from hunting makes this species especially threatened. A consortium of the Association Européenne pour l'Etude et la Conservation des Lémuriens (AEECL), the University Louis Pasteur of Strasbourg and the Fondation Nature et Découverte supported the field work and the genetic study. Socio-economic studies are under way to determine the anthropogenic effect on the remaining population (Lernould 2006).

Iary Ravaorimanana, Alphonse Zaramody,
Clément Rabarivola & Yves Rumpler

Silky Sifaka

Propithecus candidus Grandidier, 1871
Madagascar
(2000, 2002, 2004, 2006, 2008)

Propithecus candidus is a large white sifaka from north-eastern Madagascar. Silky sifakas have recently been raised to full species (Mayor *et al.* 2002, 2004; Mittermeier *et al.* 2006), though some still consider this taxon to be a subspecies of *Propithecus diadema* (see Groves 2001; reviewed in Tattersall 2007). It has a head-body length of 48–54 cm, a tail length of 45–51 cm, a total length of 93–105 cm, and a weight of 5–6.5 kg (Lehman *et al.* 2005). The pelage is long, silky and white, which gives this species its common English name. In some individuals, silver-gray or black tints may appear on the crown, back and limbs, and the pygal region (at the base of the tail) is sometimes yellow. The muzzle and face are bare, the skin a mix of pink and black, with some individuals having all pink or all black faces. The tips of the naked black ears protrude just beyond the white fur of the head and cheeks. This species does not occur with any other sifakas and cannot be confused with any lemurs within its range.

Unlike *Propithecus perrieri* and *P. edwardsi*, where adult males and females are difficult to distinguish, adult male and female *P. candidus* can be readily distinguished from one another by the pelage coloration of the upper chest. Adult males possess a large brown “chest patch” that results from chest scent marking with the sternal gular gland. As rates of male chest scent marking increase during the mating season, male chest patches become larger and can cover the entire front torso to the abdomen (Patel 2006a).

The most recent IUCN Red List assessment (2008) lists *P. candidus* as Critically Endangered. This is one of the rarest and most critically endangered lemurs. Global population size is estimated between 100 and 1,000. Silky sifakas are

hunted throughout their range as there is no local taboo, or *fady*, against eating them. Habitat disturbance, such as slash-and-burn agriculture (*tavy*), logging of precious woods (for example, rosewood) and fuel wood, also occurs in and adjacent to the protected areas where they are found (Patel *et al.* 2005b; Patel 2007b; Nielson and Patel 2008).

The silky sifaka has a very restricted range in northeastern Madagascar that includes the humid forest belt extending from Maroantsetra to the Andapa Basin and the Marojejy Massif. Marojejy National Park marks the northern limit of its current distribution, although at one time it occurred as far north as Bemarivo River near Sambava. The Androranga River may represent the northwestern range limit within the Tsaratanana Corridor. The Antainambalana River, within the Makira Conservation Site, is believed to be the southern limit. Silky sifakas may occur in northeastern Makira (Amparihobe, Bezavona), although they have not yet been observed there (Milne-Edwards and Grandidier 1875; Tattersall 1982; Wilme and Callmander 2006; Patel and Rasolofoson *et al.* 2007; Andrianandrasana 2008). Recent unconfirmed reports have identified several groups of silky sifakas just outside of north-eastern Makira in the unprotected Maherivaratra forest (Mosesy, Marojejy National Park Guide Association Chief pers. comm., February 2009) which would represent a slight enlargement of their known geographic range. In Maherivaratra and Andaparaty, silky sifakas may actually be sympatric with *Varecia*, which had never been suggested before 2008.

Surveys have documented the presence of silky sifakas in Marojejy National Park (Humbert 1955; Guillaumet *et al.* 1975; Benson *et al.* 1976, 1977; Duckworth *et al.* 1988; Nicol and Langrand 1989; Sterling and McFadden 2000; Goodman *et al.* 2003), Anjanaharibe-Sud Special Reserve (Nicol and Langrand 1989; Schmid and Smolker 1998; Goodman *et al.* 2003), the Makira Conservation Site (Rasolofoson *et al.* 2007; Ratelolahy and Raivoarisoa 2007; Patel and Andrianandrasana 2008), the Betaolana Corridor (Goodman *et al.* 2003), and the Tsaratanana Corridor (WWF Andapa Projet Simpona pers. comm.).

The majority of the remaining population of *P. candidus* is found in just two protected areas managed by Madagascar National Parks (Andapa): Marojejy National Park and Anjanaharibe-Sud Special Reserve. A few groups have recently been found in the Makira Forest Protected Area (managed by the Wildlife Conservation Society) at two sites: Andaparaty (central-east Makira) and Manandriana, 44 km to the north-west, adjacent to the Anjanaharibe-Sud Special Reserve). Silky sifakas are also found in the Betaolana Corridor that connects Anjanaharibe-Sud and Marojejy, as well as the unprotected Tsaratanana Corridor to the northwest. Further surveys are needed in Makira and in the western part of Anjanaharibe-Sud, which has recently been extended. Approximately 16 groups were found during a recent survey in western Marojejy near Antsahaberoaka (December 2008, pers. obs.).

A 14-month study (Patel 2006a; Patel *et al.* 2006) and two short studies (Kelley and Mayor 2002; Queslin and Patel 2008) have examined the behavioral biology, communication,

and feeding ecology of silky sifakas in Marojejy National Park. Silky sifakas show the greatest elevational range of any of the sifakas; as low as 300 m in the Makira (Andaparaty) and as high as 1,875 m in Marojejy. Thus, they inhabit several types of elevation-specific habitats including primary montane rainforest, sclerophyllous forest, and even low ericoid bush at their highest elevations. Their social structure is variable; they can be found in male-female pairs, one-male groups, and multi-male/multi-female groups. Groups range in size from two to nine. Home ranges (95% Kernel) vary by site from 34 to 47 ha (Patel 2006b; Patel and Andrianandrasana 2008).

Approximately 25% of the day is spent feeding, 44% resting, and the remainder is devoted to social behavior (16.8%), traveling, and sleeping. Long bouts of terrestrial play involving adults are not uncommon. Rates of aggression are low, and occur mainly during feeding. Females have feeding priority over males. As in other eastern sifakas, *P. candidus* is a folivorous seed predator eating fruits, seeds and leaves from a very large number of plant species. A recent two-month study documented feeding from 76 species across 42 families (mainly trees, but many lianas as well). During this short study, the most important plant families in their diet were Moraceae (20.3%), Fabaceae (12.9%), Myrtaceae (12.6%), Clusiaceae (10.1%) and Apocynaceae (9.5%). The four most preferred foods accounted for 37.1% of total feeding time: fruit from *Pachytrophe dimepate* (16.1%), seeds from *Senna* sp. (8.4%), young leaves from *Plectaneia thouarsii* (6.5%), and fruit from *Eugenia* sp. (6.0%). Fifty-two percent of feeding time was spent eating leaves, 34% fruit, and 11% seeds. Flowers and soil were eaten rarely (Patel 2006b; Queslin and Patel 2008).

Mating is believed to occur on a single day each year in December or January. Infants are born in June or July. Females generally give birth to a single offspring every two years, although they have been seen to give birth in consecutive years (Patel 2006b). Infants initially grasp the fur on their mother's belly, and only about four weeks later begin to ride "jockey style" on their mother's back. As is typical of *Propithecus*, all group members interact affiliatively with infants. Grooming is the most frequent form of non-maternal infant care, followed by playing, occasional carrying, as well as nursing in a few remarkable instances (Patel *et al.* 2003a; Patel 2007a). Dispersal has been observed only once, when a young adult male immigrated in 2007, aggressively forcing the older resident male out of the group he had been a member of for at least seven years. Although eastern sifakas generally exhibit male and female group transfer, female transfer in *P. candidus* has yet to be observed.

Other than humans, only the fossa (*Cryptoprocta ferox*) has been documented as a predator of the silky sifaka (Patel 2005). No aerial predation attempts by raptors have ever been observed, although these sifakas sometimes stare skyward and emit loud "aerial disturbance" roars in the presence of the large Madagascar buzzard (*Buteo brachypterus*), which does not, however, eat lemurs, only small birds. Loud

sneeze-like “zzuss!” vocalizations are their second type of alarm call, and are emitted in response to terrestrial disturbances and to lost calls by other group members, as well as after receiving aggression. Acoustic analyses have revealed sex and individual differences in the acoustic structure of the silky sifaka “zzuss” vocalization (Patel *et al.* 2003b; Patel *et al.* 2006).

As in all prosimians, olfactory communication is well developed. Eastern sifakas have several specialized scent-marking glands that include a sebaceous chest gland only found in males, and mixed apocrine-sebaceous genital glands in both sexes (Schilling 1979). Sifakas do not allomark, as in *Eulemur*, by directly scent-marking conspecifics. Females scent-mark trees by rubbing their genital glands in a rhythmic vertical motion. Males scent-mark trees in a number of ways, by rubbing them with their chest gland, genital glands, or a combination of the two. Males routinely gouge trees with their toothcombs just prior to chest-marking, which leaves long-lasting visible marks. Silky sifakas do not eat bark or gum, so such non-nutritive male tree-gouging is likely communicative in function (Patel and Girard-Buttoz 2008). Both sexes often urinate while scent-marking. Although males scent-mark two or three times as often as females, female scent-marks are responded to far more often and more quickly than male marks. A one-year study found that only 17% of male *P. candidus* marks are responded to by other group members, but 71% of female marks received a response, on average within 61 seconds (Patel 2006a). In both *P. edwardsi* and *P. candidus*, male overmarking of a female's mark is the most common response, followed by males overmarking the scent-marks of other males. Male eastern sifakas preferentially use one type of scent-marking, combined chest-anogenital marking, when depositing an overmark (Andrianandrasana *et al.* 2007). The high rates of overmarking practiced by male eastern sifakas lead to totem-tree marking, in which certain trees are covered with male scent-marks and gouge marks. Extensive scent-marking of the home range border has not been observed in *P. candidus* (Patel 2006a; Ritchie and Patel 2006; Patel and Girard-Buttoz 2008).

Erik R. Patel

Africa

Rondo Dwarf Galago

Galagoides rondoensis (Honess in Kingdon, 1997)

Tanzania

(2006, 2008)

Weighing approximately 60 g, this is one of the smallest of the galagos (Honess 1996b). It is distinct from other dwarf galagos in its bottle-brush-shaped tail, its reproductive anatomy, and its distinctive “double unit rolling call” (Bearder *et al.* 1995; Honess 1996a, 1996b; Perkin 2007). Current knowledge indicates that this species occurs in two distinct areas, one in southwest Tanzania near the coastal towns of Lindi and Mtwara, the other approximately 400 km further

north, above the Rufiji River, in pockets of forest around Dar es Salaam. One further population occurs in Sadaani National Park, approximately 100 km north of Dar es Salaam. Rondo dwarf galagos have a mixed diet of insects and fruit, often feed close to the ground, and move by vertical clinging and leaping in the shrubby understorey. They build daytime sleeping nests, which are often in the canopy (Bearder *et al.* 2003). As with many small primates, *G. rondoensis* is probably subject to predation from owls and other nocturnal predators. Among these, genets, palm civets and snakes invoke intense episodes of alarm calling (Honess 1996b).

On the IUCN Red List, the status of *G. rondoensis* has changed from Endangered (IUCN 2006) to Critically Endangered (IUCN 2008). It has an extremely limited and fragmented range in a number of remnant patches of Eastern African Coastal Dry Forest (*sensu* Burgess and Clarke 2000, p.18) in Tanzania, namely those at Zaraninge forest (06°08'S, 38°38'E) in Sadaani National Park (Perkin 2000), Pande Game Reserve (GR) (06°42'S, 39°05'E), Pugu/Kazimzumbwi (06°54'S, 39°05'E) (Perkin 2003, 2004), Rondo (10°08'S, 39°12'E), Litipo (10°02'S, 39°29'E) and Ziwani (10°20'S, 40°18'E) forest reserves (FR) (Honess 1996b; Honess and Bearder 1996). Two new sub-populations were identified in 2007 near Lindi town in Chitoa FR (09°57'S, 39°27'E) and Ruawa FR (09°44'S, 39°33'E) (Perkin *et al.* in prep.). Specimens of *G. rondoensis*, originally described as *Galagoides demidovii phasma*, were collected by Ionides from Rondo Plateau in 1955, and Lumsden from Nambunga, near Kitangari, (approximately 10°40'S, 39°25'E) on the Makonde Plateau in Newala District in 1953. Doubts surround the persistence of this species on the Makonde Plateau, which has been extensively cleared for agriculture. Surveys there in 1992 failed to detect any extant populations (Honess 1996b).

No detailed surveys have been conducted to assess population sizes of *G. rondoensis*. Limited distribution surveys have been conducted, however, in the southern (Honess 1996b; Perkin *et al.* in prep.) and northern coastal forests (27 surveyed) of Tanzania and coastal Kenya (seven surveyed) (Perkin 2000, 2003, 2004). Absolute population sizes remain undetermined but recent surveys have provided estimates of density (3–6/ha at Pande Game Reserve [Perkin 2003] and 8/ha at Pugu Forest Reserve [Perkin 2004]) and relative abundance from encounter rates (3–10/hr at Pande Game Reserve and Pugu/Kazimzumbwi Forest Reserve [Perkin 2003, 2004] and 3.94/hr at Rondo Forest Reserve [Honess 1996b]). There is a clear and urgent need for further surveys to determine population sizes in these dwindling forest patches. The total area of forest in which *G. rondoensis* is currently known to occur does not exceed 101.6 km² (Pande GR: 2.4 km², Rondo FR: 25 km², Ziwani FR: 7.7 km², Pugu/Kazimzumbwi FR: 33.5 km², Litipo FR: 4 km² and Zaraninge forest: 20 km², Chitoa FR: 5 km² and Ruawa FR 4 km² [Minimum area data source: Burgess and Clarke 2000; Doggart 2003; Perkin *et al.* in prep.]). The major threat this species is facing is loss of habitat. All sites are subject to some level of agricultural encroachment, charcoal manufacture and/or logging. All sites,

except Pande GR and Zaraninge forest, are national or local authority forest reserves and as such nominally, but in practice minimally, protected. Given current trends in charcoal production for nearby Dar es Salaam, the forest reserves of Pugu and Kazimzumbwi will disappear over the next 10–15 years (Ahrends 2005). Pande, as a Game Reserve, is perhaps more secure, and Zaraninge forest, being in a National Park, is the most protected part of the range of *G. rondoensis*. In the south, the Chitoa population is the most secure, as it is buffered by tracts of woodland. The type population at Rondo is buffered by woodland and *Pinus* plantations managed by the Rondo Forestry Project. Litipo, Ziواني and Ruawa FRs are under threat from bordering village lands.

Conservation action is urgently needed, and more research is required to determine the continuing rate of habitat loss at these sites and to survey new areas for remnant populations. There is emerging evidence (from vocalizations and penile morphology) that the northern and southern populations may be phylogenetically distinct with important taxonomic implications. As such the conservation of all populations is important.

Across its known range, the Rondo galago can be found in sympatry with a number of other galagos, including two much larger species in the genus *Otolemur*: Garnett's galago, *O. garnettii*, and the thick-tailed galago, *O. crassicaudatus*. The Rondo galago is sympatric with the Zanzibar galago, *Galagoides zanzibaricus*, in the northern parts of its range (for example, in Zaraninge forest, Pugu/Kazimzumbwi FR and Pande GR). *Galagoides zanzibaricus* was classified as Lower Risk (Near Threatened) in the 2006 IUCN Red List (IUCN 2006) due to threats to its habitat (in 2008 it was ranked as Least Concern). In the southern parts of its range (for example, at Rondo, Litipo and Ziواني FRs), the Rondo galago is sympatric with Grant's galago, *Galagoides granti*, (listed as Data Deficient in 2006, but Least Concern in 2008). The Mountain dwarf galago, *Galagoides orinus*, ranked as Data Deficient by IUCN in 2006 (considered Near Threatened in 2008), is restricted to areas of sub-montane and montane forest in the Eastern Arc Mountains further inland in Tanzania. As such *G. orinus* also has a very restricted range, although areas of its preferred habitat are believed to be at less risk of degradation because they are relatively inaccessible.

Paul E. Honess, Andrew Perkin & Simon K. Bearder

Roloway Guenon

Cercopithecus diana roloway (Schreber, 1774)

Ghana and Côte d'Ivoire
(2002, 2006)

There are two subspecies of *Cercopithecus diana*, both highly attractive, arboreal monkeys that inhabit the Upper Guinean forests of West Africa (Grubb *et al.* 2003). The roloway subspecies is distinguished by its broad white brow line, long white beard and yellow thighs. Groves (2001) considers the two subspecies to be sufficiently distinct to be regarded as full species. Of the two forms, the roloway, which is known from Ghana and eastern Côte d'Ivoire, is more seriously

threatened with extinction. In fact, along with the white-naped mangabey (*Cercocebus atys lunulatus*) and Miss Waldron's red colobus (*Procolobus badius waldroni*), it is among the three most endangered monkeys of the Upper Guinea forest block and a target species of the relentless bushmeat trade (Oates 1996).

As primatologists have searched the forests of Ghana and Côte d'Ivoire for evidence of living red colobus, they have also documented the continued decline of both the roloway guenon and white-naped mangabey, which seem to be found in (or absent from) many of the same forests (Struhsaker and Oates 1995; Oates *et al.* 1996/1997; McGraw 1998a; Koné 2004; Oates 2006). In Ghana, roloway guenons have been steadily extirpated from both unprotected and protected areas (for example, Bia National Park) and the monkey is nearing extinction in that country, if it has not disappeared already. Several recent surveys have failed to find roloways in any reserves in western Ghana. It is possible that the Ankasa Conservation Area still contains a few roloway individuals (Magnuson 2003), but in 2006 a wildlife guard reported to J. F. Oates (unpubl.) that he had not seen the monkey for several years. Careful surveys of Ankasa and Bia Conservation Areas and Cape Three Points Forest Reserve in 2007–2008 by West African Primate Conservation Action did not locate any roloways, but unconfirmed reports of their continued survival at Ankasa were received (S. Gatti pers. comm.). A thorough survey of the Dadieso Forest Reserve (where the monkey was also reported in the recent past) should be a high priority.

In neighboring Côte d'Ivoire, the Roloway guenon is not known from any protected areas and the monkey's status is equally dire. Surveys made ten years ago documented roloways in two forests, the Yaya Forest Reserve and the Tanoé forest adjacent to the Ehy Lagoon (McGraw 1998b, 2005; Koné and Akpatou 2005). Hunters had also reported small numbers of roloways in the Parc National des Iles Ehotilé (Koné and Akpatou 2005). Subsequent surveys of eighteen areas made between 2004 and 2006 confirmed the presence of roloways only in the Tanoé forest (Gonedélé Bi *et al.* 2008). This evidence suggests 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 decade and that the guenon's distribution in Côte d'Ivoire is now restricted to the Tanoé forest (Koné and Akpatou 2005). In 2007, local informants reported the presence of roloways in the Dassioko, Niouniourou, Port Gautier, Mabi and Yaya forest reserves, however surveys of these areas yielded no direct evidence of their presence (G. Campbell pers. comm.). If roloways have been eliminated from Ghana's Ankasa Conservation Area, then the Tanoé forest could be a final refuge for this guenon. This wet forest also harbors one of the few remaining populations of white-naped mangabeys in Côte d'Ivoire and, perhaps, a small number of Miss Waldron's red colobus. The Tanoé forest is under direct threat from a large palm oil company (PALMCI) and several organizations (CEPA, WAPCA) are lobbying against the company and have sponsored local awareness campaigns (Koné 2008). As the

potential last refuge for roloways, white-naped mangabeys, and Miss Waldron's red colobus, the protection of the Tanoé forest should be the highest conservation priority.

W. Scott McGraw and John F. Oates

Tana River Red Colobus

Procolobus rufomitratus (Peters, 1879)

Kenya

(2002, 2004, 2006, 2008)

Gallery forests found in the lower Tana River, Kenya, appear to be remnants of a previously continuous forest that extended from Central Africa to East Africa 25,000–30,000 years ago). The forests are part of the East African Coastal Forests Biodiversity Hotspot and for this, and other reasons, are of great conservation value. In particular, they are the only habitat for two endemic primate species; Tana River red colobus, *Procolobus rufomitratus* (Peters, 1879), and Tana River mangabey, *Cercocebus galeritus* Peters, 1879. These two species inhabit the forests along a 60-km stretch of the lower Tana River from Nkanjonja to Mitapani (01°55'S, 40°05'E). All of these forests are small, ranging in size from <1 ha to c. 500 ha. Six other species of nonhuman primates are found in this area. However, the Tana River red colobus and Tana River mangabey are forest dependent, and account for the bulk of the primate biomass in these forests.

The Tana River red colobus and the Tana River mangabey are both greatly threatened by forest loss and fragmentation caused by a growing human population. Forest is cleared mainly for agriculture; an estimated 50% of the original forest has been lost in the last 20 years. In addition, people use the remaining forest for materials to build homes and canoes, and for other non-timber forest products. Consequently, the current population of the Tana River red colobus is less than 1,000 individuals and declining, while the population of the Tana River mangabey is not much larger and declining. Furthermore, it has recently been found that the forest loss and fragmentation causes high levels of parasitism in these two primates (Mbora and McPeck 2009). The effect of this on the status of these two populations is currently unknown.

The long-term survival of the two endemic Tana River primates looks very bleak. In January 2007, the High Court of Kenya ruled that the Tana River Primate National Reserve (TRPNR), where 13 km² of forest were protected, was not established in accordance with the law. The TRPNR must, therefore, be degazetted, which means that none of the habitat of the Tana River red colobus and Tana River mangabey is legally protected. Furthermore, habitat loss outside the TRPNR has been exacerbated by the failure of the Tana Delta Irrigation Project's (TDIP) rice-growing scheme (under the administration of the Tana and Athi Rivers Development Authority [TARDA], with financing from Japan International Cooperation Agency [JICA]) to protect forest patches on their land. Now TARDA is in the process of expanding its activities in the region by establishing a 110 km² sugar cane plantation. In addition, a further 500 km² of land in and around the delta are earmarked for the development of sugarcane plantations

by Mat International Sugar Limited. These new plantations will result in a large influx of people and an increase in the demand for forest resources.

Curiously, despite the dire circumstance of Tana River red colobus and the species being on the list of The World's 25 Most Endangered Primates since 2002, not one conservation agency is working in the forests of the lower Tana River. A five-year Kenya Wildlife Service (KWS) and Kenya Forest Department project, funded by the World Bank/GEF, was initiated in 1996 to enhance conservation and protection of the primates and forests. Unfortunately, this potentially important project was terminated prematurely due to poor project management. This left the responsibility for the conservation and protection of the Tana River's forests and primates entirely to the KWS.

Despite the troubles highlighted above, the Tana River situation is not hopeless. One of us (DNMM) has maintained a (relatively poorly funded) research project in the area over the last five years. He has thus been able to monitor developments on the ground. In addition, more than 250 families who farmed within the TRPNR were voluntarily relocated in 2005 to Kipini (about 90 km away) by the KWS. At the moment, there appears to be growing concern for forest and biodiversity conservation among local people. For example, several local leaders have expressed a desire to convert the now degazetted TRPNR into a community wildlife sanctuary. However, there is need for strong support and encouragement from conservation organizations for a community-based conservation effort.

David N. M. Mbora & Thomas M. Butynski

Niger Delta Red Colobus Monkey

Procolobus epieni Grubb and Powell, 1999

Niger Delta, Nigeria

(2008)

This colobus monkey is listed as Critically Endangered on the 2008 IUCN Red List of Threatened Species. It only became known to science in 1993 in the course of a biodiversity survey co-ordinated by C. Bruce Powell (Powell 1994). The monkey's scientific name is based on its name in the Ijaw language of the people who inhabit the limited area (1,500 km²) where it occurs in the central Niger Delta. Studies of vocalizations and mitochondrial DNA suggest that *epieni* is not closely related to its closest geographic relatives, the Bioko red colobus (*Procolobus pennantii pennantii*) or Preuss's red colobus (*Procolobus preussi*), leading Ting (2008) to treat this monkey not as a subspecies of *pennantii* (see Groves 2001, 2005; 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*.

There has been only one field study of this red colobus. Werre (2000) established that *epieni* occurs only in the so-called "marsh forest" zone of the Central Delta, an area that has a year-round high water table, but which does not suffer

deep flooding or tidal effects. The study suggested that the more clumped distribution of food species in the marsh forest was a key factor restricting the monkey to its limited range, which is demarcated by the Forcados River and Bomadi Creek in the northwest, the Sagbama, Osiama and Apoi Creeks in the east, and the mangrove belt to the south. At the time of its discovery the Niger Delta 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, and 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.

As part of his research Werre (2000) formulated a conservation plan that was initially to protect 500 ha of forest near the settlement of Gbanraun through a leasehold arrangement with community landholders. It was hoped that this could eventually be expanded to a full protected area based on the proposed Apoi Creek Forest Reserve. 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 Vulnerable on the Red List. Also found in these forests are the putty-nosed monkey (*Cercopithecus nictitans*), the mona monkey (*Cercopithecus mona*), and possibly 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. *Procolobus badius waldroni* (eastern Côte d'Ivoire and western Ghana), *Procolobus preussi* (western Cameroon and eastern Nigeria), and *P. pennantii bouvieri* (Republic of Congo) are also now Critically Endangered. *Procolobus badius temminckii* (Senegal to Guinea or Sierra Leone), *Procolobus badius badius* (Sierra Leone to western Côte d'Ivoire) and *Procolobus pennantii pennantii* (Bioko Island, Equatorial Guinea) are listed as Endangered. There has been evidence of a few *P. badius waldroni* surviving in swamp forest in the far southeastern corner of Côte d'Ivoire

(McGraw and Oates 2002; McGraw 2005), but it is feared that this population may now be extinct. *Procolobus pennantii bouvieri* of the Republic of Congo has not been observed alive by scientists for at least 25 years, raising concerns that they may be extinct (Oates 1996; Struhsaker 2005). *Procolobus pennantii pennantii* is just hanging on in the southwestern corner of Bioko Island, where it has been decimated by bushmeat hunting (Hearn *et al.* 2006) in what is, theoretically, a protected area.

Although the security situation in the Niger Delta is challenging, a pilot survey is planned for early 2009 to gather information on the present status of forest and primates near Gbanraun, and to assess what options may be available for conserving any remaining *P. epieni*. A survey is also urgently needed for Bouvier's red colobus in Congo. In all the protected areas where red colobus monkeys occur, much greater efforts must be made to improve management, especially the enforcement of laws against hunting.

John F. Oates & J. Lodewijk Werre

Kipunji

Rungwecebus kipunji (Ehardt, Butynski, Jones & Davenport in Jones *et al.*, 2005)

Tanzania

(2006, 2008)

The discovery of the kipunji (*Rungwecebus kipunji*), a monkey endemic to southern Tanzania (Jones *et al.* 2005; Davenport *et al.* 2006), demonstrated how much there is still to learn about Africa's forests, as well as the continent's primate fauna. Kipunji were first found by teams working in the Southern Highlands and Udzungwa Mountains in 2003 and 2004, respectively (Jones *et al.* 2005; Davenport 2005, 2006; Davenport and Jones 2005; Davenport *et al.* 2005, 2006), sites that are some 350 km apart. Although initially placed in the genus *Lophocebus* (Jones *et al.* 2005), subsequent molecular and morphological analyses led to the monkey's placement in a new monospecific genus *Rungwecebus*, making it the first new genus of African monkey to be described in 83 years (Davenport *et al.* 2006). Further molecular studies have corroborated the validity of the genus (Olson *et al.* 2008) and anatomical investigations are under way.

More importantly, however, the kipunji is one of the world's most threatened primates, as demonstrated by a recent census that provided the first systematically-derived data on the animal's abundance and distribution (Davenport *et al.* 2008). Kipunji are cryptic, rare, primarily arboreal and in urgent need of conservation attention (Davenport *et al.* 2006; Davenport and Jones 2005), and consequently a complete count after a long-term survey was made, ensuring a much more accurate population estimate (Davenport *et al.* 2008). The census demonstrated that the kipunji is probably Africa's rarest monkey, and provided empirical data in support of its official designation as 'Critically Endangered' on the 2008 IUCN Red List, with the genus facing an extremely high risk of extinction in the wild (Davenport *et al.* 2008; Davenport and Jones 2008).

The kipunji is restricted to a number of discrete portions of the forests of Mt. Rungwe and the adjacent Livingstone (in Kitulo National Park) in the Southern Highlands, and the Vikongwa area of the Ndundulu forest (in the new Kilombero Nature Reserve) in the Udzungwa Mountains. The Mt. Rungwe-Livingstone population occupies degraded submontane and montane forest between 1,750 and 2,450 m above sea level, whereas the Ndundulu population lives between 1,300 and 1,750 m above sea level in submontane forest (Davenport *et al.* 2006, 2008). Kipunji have not been recorded in the Udzungwa Mountains National Park itself, the closest record being 1.9 km outside the park boundary (Jones 2006). Despite extensive surveys, kipunji have not been recorded from other forests in either the Southern Highlands or the Udzungwa Mountains.

During the census, a total of 34 kipunji groups were identified in the Southern Highlands with an estimated total population of 1,042. Of these, 501 individuals in 16 groups were counted in Mt. Rungwe and 541 individuals from 18 groups in the Livingstone forest of Kitulo National Park. In Ndundulu, just four groups were identified with an estimated total of 75 animals. The total global population of the kipunji therefore, is estimated to be just 1,117 animals, living in some 38 groups (Davenport *et al.* 2008). During the same surveys, the Areas of Occupancy (AoO) for Mt. Rungwe, Livingstone Forest and Ndundulu were estimated to be 671 ha, 408 ha and 199 ha, respectively. The total for Rungwe-Kitulo therefore was 1,079 ha, and the total species' AoO was 1,278 ha based on data collected over three years (Davenport *et al.* 2008). Meanwhile the Extents of Occurrence (EoO) for kipunji for Mt. Rungwe, Livingstone and Ndundulu were 815 ha, 425 ha, and 528 ha, respectively. The total for Rungwe-Kitulo was 1,241 ha and the total species EoO was estimated to be 1,769 ha (Davenport *et al.* 2008).

A total population of just 1,117 animals is very small. As reported elsewhere, both the Mt. Rungwe and Livingstone forests are heavily degraded (Davenport 2005, 2006; Davenport and Jones 2005) and remote sensing analysis of forest cover has demonstrated that the extent of habitat connection between the various groups is extremely tenuous. Indeed the Mt. Rungwe-Kitulo portion of the population consists of a number of isolated sub-populations and this is compounded by the poor condition of the narrow Bujingijila Corridor that joins Mt. Rungwe and Livingstone (Davenport 2005). With the loss of this corridor, the Mt. Rungwe-Kitulo population will be further fragmented. Furthermore, and in addition to the continuing loss of habitat, this population continues to be hunted (Davenport 2005, 2006; Davenport *et al.* 2005).

The fragile status of the population in Ndundulu is particularly worrying and its causes remain unknown. However, given current thinking on primate population sizes, it may be that this population is no longer viable (Davenport *et al.* 2008). The recent census also revealed an interesting and statistically significant difference in mean group size between the Rungwe-Kitulo and the Ndundulu populations (Davenport *et al.* 2008). This may be due to the small total

population size in Ndundulu, or to fragmentation, reduced resource patches and food availability in Rungwe-Kitulo, as demonstrated in other primate species. Either way, the kipunji is more sparsely distributed than initially thought (Jones *et al.* 2005). The total EoO (species range) is just 17.69 km² giving grounds for much conservation concern, and being considerably less than the 100 km² required to fulfill the 'Critically Endangered' criterion of the IUCN Red List.

An estimated 541 individuals reside in Livingstone, a forest that has been incorporated into Kitulo National Park. This should significantly improve protection for the kipunji groups in this area, although the forest is severely degraded (Davenport 2006), and illegal activities, including logging and hunting of primates, are only now being brought under control. A new management plan for Kitulo National Park has recently been produced, in which the mandate for research and monitoring of the kipunji falls to the Wildlife Conservation Society (WCS). There are no immediate plans for habituation of the animal for tourism until appropriate and thorough research has been carried out on its potential impacts. However, a section of forest contiguous with Mt Rungwe, and containing groups of kipunji, is now being leased to, and managed by, WCS. The kipunji here are being studied and monitored full time by WCS staff as well as national and international students.

More than 51% of the total kipunji population lives in forests with comparatively little management. However, there are grounds for optimism. Ndundulu Forest Reserve was absorbed by the new Kilombero Nature Reserve in 2007 (Marshall *et al.* 2007) under the auspices of the Forestry and Beekeeping Division of the Ministry of Natural Resources and Tourism. Similarly, Mt. Rungwe, so long a neglected Catchment Forest Reserve, is now in the final stages of becoming a nature reserve as well. This will complement the adjacent national park and enable community involvement. A management plan is currently being written, and reserve rangers have recently been hired and trained. It will, however, be some time until illegal activities are brought under control, even with full resources at the authority's disposal.

On Mt. Rungwe, where forest clearance, hunting and fragmentation pose the most serious threats (Machaga *et al.* 2005), the reasons for the animal's discrete distribution are being studied. Moreover, research is being carried out on aspects of the kipunji's social and reproductive behaviour, feeding ecology, home range dynamics, predation and demography. Across Rungwe-Kitulo, the isolated sub-populations may already be subject to a loss of genetic variability due to low effective breeding populations. Some may no longer be viable and this is also under investigation.

Southern Ndundulu, meanwhile, is in excellent condition due chiefly to its remote location (Davenport and Jones 2005). However, the long-term viability of the 7% of the kipunji population must be considered uncertain, at best. It is possible that this population is simply dying out 'naturally', but research into the reasons for, and the viability of, the small Udzungwa population is ongoing. Whether any

tangible primate conservation measures could or should be applied in a largely undisturbed habitat is debatable. The focus of applied kipunji conservation work is currently the protection and restoration of the montane forest habitats of Mt. Rungwe, widespread environmental education, and support to both management authorities and local communities across the range.

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Cross River Gorilla

Gorilla gorilla diehli Matschie, 1904

Nigeria and Cameroon

(2000, 2002, 2004, 2006, 2008)

The Cross River gorilla (*Gorilla gorilla diehli*) is the most western and northern form of gorilla, and is restricted to the forested hills and mountains of the Cameroon-Nigeria border region at the headwaters of the Cross River. It is separated by about 300 km from the nearest population of western lowland gorillas (*Gorilla gorilla gorilla*), and by around 250 km from the gorilla population in the Ebo Forest of Cameroon. The most recent surveys suggest that between 200 and 300 Cross River gorillas remain. Groups of these gorillas concentrate their activities in 11 localities across a 12,000 km² range, though recent field surveys confirmed the presence of gorillas outside of their known localities suggesting a wider distribution within this range. This distribution is corroborated by genetic research, which has found evidence that many Cross River gorilla localities continue to maintain contact through the occasional dispersal of individuals.

There are many human settlements around the forests where the gorillas occur, including a number of enclaved villages within Okwangwo and Takamanda. The encroachment of farms, dry-season fires set to clear forest or improve pasture, and development activities, such as roads, continue to threaten the integrity of gorilla habitat. However, large tracts of lower elevation forest remain between the localities

where the gorillas are presently concentrated, and if these areas can be protected, the animals may be able to expand their range and population size. Genetic evidence suggests that the decline in the population of Cross River gorillas has been recent, and is probably associated with the introduction of hunting with firearms. After several years of awareness-raising by conservationists and researchers, hunting of Cross River gorillas for bushmeat has been reduced to a low level, but it is still a potential threat, as are wire-snare traps set for other animals. A conservation action plan to improve the survival prospects for the Cross River gorilla was published in 2007 (Oates *et al.* 2007), and many of the key recommendations contained in the plan have already been implemented.

Afi Mountain Wildlife Sanctuary (AMWS). AMWS is managed by the Cross River State Forestry Commission with support from a partnership of NGOs. Although levels of hunting have been reduced, the presence of more than 600 illegal farms within the sanctuary remains an unresolved problem. Widespread illegal logging in the contiguous Afi River Forest Reserve (ARFR) and the spread of farms from the intervening Buanchor enclave threaten to sever the habitat corridor linking Afi to the Mbe Mountains in the east. The sanctuary itself has been largely protected from logging due to its steep mountainous slopes, although the forest is frequently damaged by dry-season fires. Habituation of gorillas for the purposes of tourism is proposed, the potential costs and benefits of which are being debated. The sanctuary is also the proposed release site for the reintroduction of captive drills (*Mandrillus leucophaeus*) by the NGO Pandrillus.

The Mbe Mountains. The Mbe Mountains are a critically important corridor linking Afi Mountain to the west with Cross River National Park and the larger block of gorilla habitat to the east. Lacking any formal conservation status, traditional ownership of the Mbe forest is claimed by nine surrounding communities. In 2005, these communities formed the Conservation Association of the Mbe Mountains (Camm) to manage the area both for conservation and to provide benefits to local communities. Capacity building and

Key sites for the Cross River gorilla and the Ebo gorilla

Country/Site	Status	Altitude (m above sea level)	Area (km ²)	Gorilla range (km ²)	Estimated numbers
Nigeria					
Afi Mountain	Wildlife Sanctuary	130–1,300	c.100	40	25–30
Mbe Mountains	Proposed Community Wildlife Sanctuary	110–900	85	25	25–30
Okwangwo Division of Cross River National Park	National Park	110–1,700	640	65	25–50
Cameroon					
Takamanda	National Park and adjacent area of unclassified forest	80–1,700	676	80	45–59
Mone River	Forest Reserve	110–1,200	538	68	20–30
Mbulu	Unclassified forest	500–2,000	c.1,000	54	20–30
Kagwene	Gorilla Sanctuary	1,700–2,000	19	c.9	17–19
Bechati-Fossimondi-Besali	Unclassified forest	200–1,700	80–100	c.25	20–30
Ebo Forest	Proposed National Park	200–1,200	c.2,000	c.25	c.25

support for CAMM is currently provided by the local NGO Development in Nigeria, and an effective patrol system by a team of eco-guards has been established by the Wildlife Conservation Society (WCS). A protected area boundary has been provisionally demarcated, and the permanent demarcation and legal gazettement of this boundary is currently in progress. Although the mountains remain under threat from logging, agricultural encroachment and hunting, these threats have diminished somewhat since 2005.

Cross River National Park: Okwangwo Division. Cross River National Park (CRNP) is the most important site for Cross River gorillas in Nigeria and is contiguous with Takamanda in Cameroon. A number of new ranger posts have been constructed, and in 2008 all park rangers received basic training in anti-poaching techniques. Despite the efforts of park authorities, high levels of hunting occur in most areas of the Okwangwo Division (the northern sector of Cross River National Park where gorillas are found), driven by the lucrative bushmeat market. In addition, farmland belonging to three large village enclaves threaten to divide the park in two and thereby isolate the forest and gorillas of the 'Boshi Extension' in the extreme north from the rest of the Okwangwo Division. There have been recent efforts to promote transboundary collaboration between Okwangwo and Takamanda. Unfortunately, should the long-threatened privatization of national parks in Nigeria proceed as planned, the consequences for the remaining gorillas could be catastrophic.

Takamanda National Park. Originally established as a forest reserve in 1934, Takamanda was upgraded to a national park in November 2008. Takamanda's long history of use by local communities poses one of the biggest challenges to conservation in this area. The unsustainable harvesting of wildlife, certain non-timber forest products, and illegal timber extraction (from surrounding areas) must be brought under control. Many of these activities are driven by market forces in Nigeria, and a transboundary approach is essential to success. Despite these challenges, Takamanda and the adjacent Mawambi Hills located outside the southern boundary of the park provide refuge to a significant proportion of Cameroon's Cross River gorillas as well as scattered groups of drills (*Mandrillus leucophaeus*), chimpanzees (*Pan troglodytes ellioti*), Preuss's guenons (*Cercopithecus preussi*) and other large mammals. The area is also known to be of importance in terms of plant diversity, birds, reptiles and other taxa.

Mone River Forest Reserve. Mone was created as a Forest Reserve in the 1950s. Although there are no human settlements within the reserve, local people continue to harvest timber, wildlife, and other forest products. Due to a varied topography and rich vegetation, Mone still provides habitat for a number of large mammals, including Cross River gorilla in the northern half of the reserve (recent studies have confirmed that these gorillas are using an area larger than previously confirmed). In 2003, the government indicated that they might upgrade Mone to a Wildlife Sanctuary, and surveys have confirmed that this is urgently warranted. WCS in partnership with the UNEP/UNESCO Great Ape Survival

Programme (GrASP) is to lead a feasibility study into carbon-based marketing for the area.

Mbulu forest. The Mbulu forest is a large block of unclassified forest which cloaks a series of extremely rugged and remote valleys located between the Kagwene Gorilla Sanctuary and northern Takamanda. Human population pressure is relatively low, with human activities being largely restricted to farming in valley bottoms and accessing the forest via ridges or less steep slopes for hunting and the harvesting of other forest products. Because of the relatively low human pressure, Cross River gorillas and other important species have found refuge here, and the forests of Mbulu provide one of the best opportunities to maintain some form of habitat corridor connectivity between various Cross River gorilla sites in the area. WCS is reviewing the possibility of establishing new protected areas and corridors, while at the same time working with villages adjacent to Cross River gorilla sites to establish a community-based gorilla protection and monitoring system known as the 'gorilla guardian network.'

Kagwene Gorilla Sanctuary. Created in 2008, this sanctuary is the only protected area established specifically to conserve the Cross River gorilla. Although of limited size, Kagwene is home to a number of gorillas which are the subject of long-term research, monitoring and protection activities. Due to the daily presence of gorilla monitors, Kagwene is an important site for capacity-building related to gorilla monitoring and awareness-raising. In the near future, the Ministry of Forestry and Wildlife plans to post a warden and eco-guards to the Sanctuary, and will become increasingly involved in the financing of the site. Urgently required management measures include the demarcation of boundaries, recovery of farm-encroached forest, and integration of surrounding communities into management strategies. It is also important that forest corridor links with Mbulu are maintained.

Bechati-Fossimondi-Besali (BFB) Forest (now Leb-iale-m-Mone Forest Landscape). The BFB Forest is about 40 km south of Mone and is the southernmost location of the Cross River gorilla. Though of limited size (c.120 km²), the BFB Forest is home to a likely small, but as yet unknown number of gorillas. In 2007, the Environment and Rural Development Foundation (ERuDeF) expanded gorilla research and conservation activities to Ndumbin-Nkandu and the Bechati-Mone Forest Corridor. In 2009, research activities will include the Mak-Betchou Forest, Nkingkwa Hills and Mbanga/Mpongo-Ebensuk Forest. ERuDeF also initiated community-based management activities to secure a future for great apes in the BFB Forest, where the main threats are habitat fragmentation and forest encroachment by small farmers, and hunting.

Ebo Forest. About 250 km south of the Cross River population and 5 km north of the Sanaga River, the Ebo Forest in southwestern Cameroon is home to a small isolated population of gorillas the taxonomic affinities of which are still unclear. The forest, which covers almost 2,000 km² and is adjacent to a large FSC-certified logging concession at its northern perimeter, is characterized by extreme topography

and a diversity of habitats, and holds a unique assemblage of 11 diurnal primate species. Researchers from the Zoological Society of San Diego's Ebo Forest Research Project have been working in Ebo since 2005, and one of the three research stations is situated in the gorilla's range. Recent field research suggests that fewer than 25 individual gorillas survive in an area of about 25 km². The Ebo forest is also inhabited by important populations of other highly threatened species such as the drill (*Mandrillus leucophaeus leucophaeus*), Preuss's red colobus (*Procolobus preussi*) and the Gulf of Guinea chimpanzee (*Pan troglodytes ellioti*). The forest's primates are under extreme pressure from bushmeat hunting to supply the commercial trade, given the proximity of Ebo to the main urban centers in Cameroon. Over 1,000 km² of the Ebo Forest is currently being gazetted as a national park.

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Asia

Siau Island Tarsier

Tarsius tumpara Shekelle, Groves, Merker & Supriatna, 2008
Indonesia
(2006, 2008)

The Siau Island tarsier, *Tarsius tumpara*, is a newly described species that is Critically Endangered and faces an imminent threat of extinction. Shekelle and Salim (2009) used GIS data and field surveys to list specific threats. They include: a very small geographic range, of 125 km², and an even smaller area of occupancy, perhaps as little as 19.4 km²; a high density of humans (311 people per km²) that habitually hunt and eat tarsiers for snack food; and an extent of occurrence that is entirely volcanic in its geological composition, with Mount Karengetang, a massive and highly active volcano, dominating more than 50% of its geographic range. Furthermore, there are no protected areas within its range (Riley 2002; Shekelle *et al.* 2007; Shekelle and Salim 2009), and all captive breeding programs for tarsiers, including several by leading zoos and primate centers, have been dismal failures, leaving no *ex situ* conservation options for any tarsier species anywhere (Fitch-Snyder 2003).

The most reasonable interpretation of the scant data is that population size is very small, in the low thousands at best, and declining (Shekelle and Salim 2009). Despite the fact that Sangihe Island is renowned for its Critically Endangered avifauna (Whitten *et al.* 1987; Whitten 2006), Shekelle and Salim (2009) found that the conservation threat for *Tarsius tumpara*, on Siau Island, was greater, for every variable measured, than that faced by *T. sangirensis*, which nevertheless is Endangered (Shekelle and Salim 2009). Thus, in spite of the fact that *T. tumpara* was only recently described and remains almost unknown, sufficient evidence indicates that it teeters on the brink of extinction on an island where the entire endemic fauna and flora are at risk (Shekelle *et al.* 2007).

The taxonomic distinctiveness of Siau Island tarsiers was predicted by the Hybrid Biogeographic Hypothesis for Sulawesi (Shekelle and Leksono 2004). Sangihe and Siau Islands are part of a volcanic arc and are separated by approximately 60 km of deep ocean, greater than 1,000 m in depth; far greater than the 180 m depth normally used by biogeographers for the maximum extent of dry land exposed during glacial maxima. There is no feasible means for recurrent gene flow between these islands today, nor is there any historical indication of a land connection between them. Shekelle *et al.* (2008a) reported acoustic and morphological evidence that supported taxonomic separation of the Siau Island population, but a sister-taxon relationship between *T. tumpara* and *T. sangirensis* relative to other known species of tarsier. Shekelle *et al.* (2008b) reported genetic data for *T. sangirensis* along with numerous other tarsiers and comparative primate data. These data revealed that *T. sangirensis* is the sister-taxon of a clade consisting of all other Sulawesian tarsiers in their data set, with an average genetic distance between *T. sangirensis* and other Sulawesian tarsiers being approximately 80%; as great as that found between *Homo* and *Pan*, as measured at the same locus. They infer, therefore, that *T. sangirensis* split from other Sulawesi species several million years ago. Although tissue samples were collected for *T. tumpara*, genetic data are not available at this time owing to the extremely strict control of tarsier tissue for export from Indonesia in recent years, and the comparatively weak capacity for collecting such data within country (M. Shekelle pers. obs. and unpubl. data). Given the isolation between Sangihe and Siau Islands, however, it is reasonable to infer that the taxonomic uniqueness of *T. tumpara* is measured in hundreds of thousands, or even millions of years. Aside from the skull in Dresden, there is no further evidence in the literature of research on this species.

Shekelle's surveys found evidence of tarsiers in only two places, on the shores of a small fresh water pond at the extreme southern end of the island, and on a steep cliff face along the east coast road where it runs next to the ocean. Numerous other sites that looked promising based upon our experience with *T. sangirensis* turned up no evidence of tarsiers. Interviews with several locals indicated that tarsiers had formerly been common at these sites as recently as 10 years ago, but were now rare or non-existent. They also added that tarsiers, and other small endemic mammals such as the dwarf cuscus, were a popular snack food called "*tola-tola*", and that it had formerly been common to eat 5 to 10 animals at a single sitting after hunting them with air rifles. More recently, reports by a colleague (Noldi Kakauhe pers. comm.) indicated that tarsiers are present high on the flanks of Mt. Karengetang, near the edge of the caldera, by the village of Salili. Furthermore, as reports of *Tarsius tumpara* have spread and circled back to Siau Island, it has become apparent that some residents of Siau Island are sensitive to reports that some of the islanders eat tarsiers. Thus a careful line needs to be drawn between accurately reporting genuine threats to this species, and sensationalism that could damage relations between conservationists and island residents. Indeed, the specific name,

tumpara (the word for tarsier in the local dialect on Siau Island) was given as a means to honor local residents, in the hope that they would actively work to preserve their biological heritage (Shekelle *et al.* 2008a).

Myron Shekelle & Agus Salim

Javan Slow Loris

Nycticebus javanicus É. Geoffroy, 1812

Indonesia

(2008)

All Asian lorises are imperiled by the devastating loss of their habitat; indeed, this major threat resulted in Sri Lanka's Critically Endangered Horton Plains slender loris appearing rightfully in the last two incarnations of this list (Nekaris 2006; Nekaris and Perera 2007). An even greater immediate threat to Asian lorises, however, is their high demand in the rampant Asian pet and traditional medicine trades (Schulze and Groves 2004; Streicher 2004). Easy to catch due to their slow locomotion, numbers of lorises in animal markets far outstretch the ability of these slow-reproducing primates to recover their population numbers in the wild (Shepherd *et al.* 2004). Indeed, this threat raised international concern, resulting in the transfer of all members of the genus *Nycticebus* to CITES Appendix I in 2007 (Nekaris and Nijman 2007). Five species of slow loris are now recognized: *N. coucang* (greater), *N. pygmaeus* (pygmy), *N. bengalensis* (Bengal), *N. menagensis* (Bornean), and *N. javanicus* (Javan) (Roos 2003; Chen *et al.* 2007). All slow lorises suffer from trade throughout their range, but when combined with tremendous habitat loss, no other species has been harder hit than the Javan slow loris.

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

Nycticebus javanicus is found only on the Indonesian island of Java. Java has a long history of cultivation and deforestation that already started *c.* 1000 AD, but really took off in 1830 when the Dutch colonial government imposed the so-called 'cultuurstelsel'. To support this agro-economic system, farmers were forced to grow export crops on communal grounds, which were often forest (Whitten *et al.* 1996). By the end of the 19th century the natural forest was severely

fragmented, and at the beginning of the last century the remaining forest, especially in West and Central Java, showed a fragmentation pattern very similar to that seen today. Over the last few decades, the decrease in forest area has been slow. At present, less than 10% of the original forest remains, most of it covering the higher slopes of the central mountains.

GIS models have shown that historic forest loss and continued degradation mean that less than 20% of habitat suitable for *N. javanicus* remains. Species distribution modeling and a Gap Analysis have also revealed that only 17% of the potential distribution of *N. javanicus* is currently within the protected area network of Java. Furthermore, Thorn *et al.* (2008) have highlighted conservation priority areas for the increased protection of *N. javanicus*, based on GIS analysis and ecological niche modeling. These include recommendations for the extension of seven important protected areas across the island, as well as 11 priority survey sites where the current distribution and abundance of this enigmatic primate should be studied. More surveys are vital since the decreased number of Javan lorises in trade seems to correlate with exceedingly low numbers in the wild (Nekaris *et al.* 2008). Indeed, surveys by three research groups all showed animals to occur at 0.02 to 0.20 animals per km, when they could be found at all, meaning 5–10 km must be walked to see a single loris (Nekaris and Nijman 2008; Winarti 2008). Roads and human disturbance have been shown to correlate negatively with Javan slow loris abundance (Collins 2007; Winarti 2008).

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

In Java itself, lorises are often specifically targeted for the trade (Sanchez pers. obs.). Local villagers who find a loris take it to a distributor dealer who compiles a stock of lorises. These animals go to middlemen who then distribute them throughout the "bird" markets in the main towns in Java. The traders who ultimately sell the animals are aware that trading lorises is profitable, reaching a price in the market up to ten times or more the purchasing price at the stocker's level.

Once they arrive at a market, lorises face other threats. To avoid being bitten by the purportedly toxic lorises, traders habitually cut or pull out an animal's front teeth. Most of these lorises die due to dental abscess or pneumonia. Those that do survive are no longer able to eat their preferred food (gum) (Wiens *et al.* 2006), or to engage in the important behavior of social grooming with the toothcomb, meaning that any confiscated animals are unlikely to survive if released to the wild. Reintroduction itself is a threat to the Javan loris; three major trade hubs, markets in Jakarta, Bandar Lampung and Palembang, receive lorises from

throughout the region. The similar appearance of lorises to the untrained eye results in release of other loris species into Java, with potential for disastrous effects from hybridization or displacement by invasive species.

To combat the issue of trade, starting in 2002, a handful of foreign-aid assisted rescue centers were built up in Indonesia. These rescue centers became the haven for many different species of illegally traded wildlife confiscated by the Indonesian forest authorities, including hundreds of slow lorises. Up to 95–100% mortality of slow lorises has been reported by most rescue centers, due to untreated dental infections, improper care and malnutrition, as well as inappropriate releases. This problem is being combated with help from International Animal Rescue Indonesia (IARI), which set up the first facility specialized for the rescue and rehabilitation of lorises in Indonesia in 2006. Working closely with other NGOs, Indonesian Universities, and the Indonesian Ministry of Forestry, a Loris Rescue Unit is being set up to work on market investigations, rescue, rehabilitation and release of lorises, education and awareness, and supporting research work.

For a long time, slow lorises were thought to be common throughout Indonesia, and the presence of animals in trade was believed to be an indicator of their abundance. We are only beginning to unravel the complexity of their taxonomy and distribution, leading to an overall bleak picture. If trade cannot be halted, Critically Endangered will be a more apt listing for these evolutionarily distinct and beautiful primates. 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.

*K. A. I. Nekaris, K. Llano Sanchez,
J. S. Thorn, I. Winarti & V. Nijman*

Simakobu or Pig-Tailed Snub-Nose Langur

Simias concolor Miller, 1903

Indonesia

(2002, 2004, 2006, 2008)

The simakobu monkey (*Simias concolor*) again is serving as the flagship species for the four Mentawai Island primates. The other three species inhabiting the 7,000 km² archipelago located west of Sumatra are Kloss's gibbon (*Hylobates klossii*), the Mentawai Island leaf monkey (*Presbytis potenziani*), and the Mentawai macaque (*Macaca pagensis*). *Simias* is a monotypic genus with two subspecies. *Simias concolor concolor* Miller, 1903 inhabits Sipora, North Pagai, and South Pagai Islands and several small islets off of South Pagai. *Simias c. siberu* Chasen and Kloss, 1927 occurs only on Siberut Island.

Very little has been published on simakobu behavior and ecology. The first activity budget of habituated simakobus described the activities of two groups living in the Betumonga region of southwestern North Pagai. The data show that simakobus spend almost equal amounts of time resting

(46%) and feeding (44%), and less time moving (7%) (Paciulli and Holmes 2008). Wendy Erb is in the middle of a year-long study of male simakobu behavior, which should yield more complete data on basic activity patterns (pers. comm.).

New estimates of the amount of forest cover remaining on the Pagai Islands (about 826 km²) have been calculated using Google Earth Pro composite satellite imagery (Paciulli and Viola 2009). The forest cover coupled with primate density data (Paciulli 2004) indicate that there are approximately 3,347 simakobus, 1,049 Kloss's gibbons, 1,545 leaf monkeys, and 7,984 pig-tailed macaques on the Pagai Islands. 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).

The 190,500-ha Siberut National Park, a UNESCO Biosphere Reserve, covers 47% of Siberut Island and serves as the main reserve for the Mentawai primates. The large majority of the other remaining natural habitat lies outside officially protected areas. Most of these areas are subjected to human encroachment, product extraction, commercial logging, and conversion to cash crops and oil palm plantations (Whittaker 2006). Although hunting appears to be declining and opportunistic in many areas of the Pagais, where it still occurs it has devastating effects on *S. concolor*, the preferred game species (Mitchell and Tilson 1986; Fuentes 2002; Paciulli 2004). In addition, *S. concolor* seems to be particularly sensitive to logging, having 5 individuals/km² in unlogged Pagai forests to half that amount (2.5 individuals/km²) in Pagai forest patches logged 20 years earlier (Paciulli 2004). Drastic measures need to be taken to ensure that the Peleonan Forest on Siberut and areas on the Pagais are truly protected.

Lisa M. Paciulli

Delacour's Langur

Trachypithecus delacouri (Osgood, 1932)

Vietnam

(2000, 2002, 2004, 2006, 2008)

Delacour's langur, also known as the white-rumped black leaf monkey, is endemic to Vietnam, occurring in a very restricted area in the north of the country that comprises about 5,000 km² between 20°–21°N and 105°–106°E. The distribution is closely related to the limestone mountain ranges in the provinces Ninh Binh, Thanh Hoa, Hoa Binh, and Ha Nam. Currently there are 18 locations known where Delacour's langurs occur. They are isolated populations, and combined total at most only 400 to 450 km². We know of five localities where local people have reported that it has been extirpated. The northwestern border of the distribution is Mai Chau, between the Da River in the north and the Ma River in the south. The Da River seems to form the northern border of the species' range, but the exact southern boundary is unclear. There are a number of isolated limestone areas south of the Ma River, but the only location where they are known there is the limestone complex between Lang Chan and Ngoc Lan. This population is, however, now most probably extirpated. It seems that this species never occurred south of the Chu River.

During the decades following the discovery of Delacour's langur in 1930 there was only scanty information on its existence and distribution. The first sightings of live animals were reported in 1987 from Cuc Phuong National Park. The most important and for some subpopulations the only factor for the decline in numbers is poaching, which is not primarily for meat, but for bones, organs and tissues that are used in the preparation of traditional medicines. The 18 isolated wild populations of Delacour's langur have been confirmed over 10 years of surveys and monitoring by the Frankfurt Zoological Society. The total population counted in 1999/2000 was about 280 to 320 individuals. The recorded numbers of animals hunted over the 10 years (1990 to 1999) totaled 320, an annual loss of more than 30 individuals, but the real number is undoubtedly higher. Sixty percent of all existing Delacour's langurs occur in isolated populations with less than twenty animals. The loss of these subpopulations, and consequently 60% of the entire population, is foreseeable without management, strict regulations and law enforcement. Surveys in 2004 in two protected areas with important subpopulations—Cuc Phuong National Park and Pu Luong Nature Reserve—showed a decline in numbers of 20% in 5 years (2000 to 2004). Surveys were carried out in 2008 by the Frankfurt Zoological Society in search of possibilities to translocate small isolated populations under severe threat. The results of the surveys show a continuously dramatic decline. The population in Ngoc Son Nature Reserve is most probably extirpated, the population in Pu Luong Nature Reserve reduced by about 25%, and the population in Cuc Phuong National Park reduced to 8 to 11 individuals. It is to be expected that the population in unprotected areas which have yet to be surveyed will show a similar tendency. A reasonable estimate of the current population indicates no higher than 200 individuals. Surveys by the Frankfurt Zoological Society continue, and should provide background information about status of populations and logistics for translocation. The improvement of protection for most of the subpopulations is not a realistic option, and most subpopulations are already too small for recovery without active management.

Four areas where Delacour's langurs occur are protected: Cuc Phuong National Park, Pu Luong Nature Reserve, Hoa Lu Cultural and Historical Site, and Van Long Nature Reserve. Van Long Nature Reserve is believed to harbor the largest remaining population. Delacour's langurs there are well protected due to close cooperation between the provincial forest protection authorities and a local guard unit paid and trained by the Frankfurt Zoological Society. Since the establishment of the Nature Reserve in 2001, the population of Delacour's langurs has grown by about 35%, and currently numbers 80 to 90 individuals. Efforts to save this species are one focus of the Vietnam Primate Conservation Program of the Frankfurt Zoological Society and the Endangered Primate Rescue Center at Cuc Phuong National Park, established in 1993 primarily to safeguard the future of this and other endangered Vietnamese primates. The Endangered Primate Rescue Center is the only facility which keeps this species. The center started a breeding

program with five confiscated animals, and 15 individuals have been born since 1996. The aim is to reintroduce these langurs into well-protected areas to establish additional free ranging populations.

Tilo Nadler

Golden-headed Langur or Cat Ba Langur

Trachypithecus p. poliocephalus (Trouessart, 1911)

Vietnam

(2000, 2002, 2004, 2006, 2008)

The golden-headed langur, *Trachypithecus p. poliocephalus*, is probably the most endangered of the Asian colobines. This subspecies occurs only on the Island of Cat Ba in the Gulf of Tonkin, northeastern Vietnam. The Cat Ba Archipelago is in the world-famous Ha Long Bay, a spectacular karst formation that was invaded by the sea. The golden-headed langur inhabits tropical moist forest on limestone karst hills, and shares this habitat preference with the other six to seven taxa of the *T. francoisi* group. These so called karst langurs, including the Cat Ba langur and its closest relatives, the white-headed langur, *T. p. leucocephalus* Tan, 1955, in southern China, display strict behavioral adaptations to their karst habitat.

There are no systematic and reliable data available on the historic density of the langur population on Cat Ba Island. According to reports of indigenous people the entire island of Cat Ba (140 km²) and some smaller offshore islands were previously densely populated by langurs. Hunting has been the sole cause for the dramatic and rapid population decline from an estimated 2,400–2,700 in the 1960s to only 53 individuals by 2000. The langurs were poached mainly for trade in traditional medicines. Since the implementation of strict protection measures towards the end of 2000, the langur population on Cat Ba Island increased to current 60–70 individuals.

Although the growth of the population is encouraging, the overall status of the subspecies is most critical. As a result of habitat fragmentation, the remaining population is now divided into seven isolated sub-populations, probably only four of which include males, while the others are all-female groups and thus non-reproducing social units. The total reproductive output in this species is accordingly low. Since a peak in births in 2003, the reproductive output of the Cat Ba Langur has stagnated at 1–2 offspring per year.

Cat Ba Island and the surrounding area 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. The Cat Ba Archipelago (some 1,500–2,000 large and small islands, cliffs and rocks) was designated a UNESCO Man and the Biosphere Reserve in 2004.

Despite this, nature and wildlife protection on Cat Ba Island is deficient, though awareness as well as partnership and commitment with the local communities are slowly increasing. However, efforts to effectively conserve the langurs and their habitat face major obstacles due to the need to better address the local community's aspirations for development,

and due to the steadily increasing human population, besides persistent, severe deficiencies in law enforcement. As elsewhere in the region, poaching is driven by increasingly attractive commercial gains in satisfying the immense local and regional demand for wildlife and animal parts. The strictest protection regime possible is necessary for the survival of all the mammals and other species on Cat Ba that are, like the langurs, targeted by the Asian wildlife trade.

A conservation program for the golden-headed langur on Cat Ba was initiated in November 2000 by the Zoologische Gesellschaft für Arten- und Populationsschutz (ZGAP), München, and Allwetterzoo Münster, Germany. The aim is to provide for their protection, reduce population fragmentation, and contribute to the conservation of the biodiversity on Cat Ba Island in collaboration with Vietnamese authorities.

*Daniela Schrudde, Roswitha Stenke,
Phan Duy Thuc & Martina Raffel*

Western Purple-faced Langur

Trachypithecus (Semnopithecus) vetulus nestor Bennett, 1833
Sri Lanka
(2004, 2006, 2008)

One of the most serious problems facing Sri Lanka's western purple-faced langur (*T. v. nestor*) stems from the fact that it inhabits some of the most densely populated regions of the country. As a result, this endemic monkey's long-term survival is severely threatened by unplanned and haphazard urbanization. A recent survey involving nearly 1,900 km of travel through one-third of *T. v. nestor*'s historical range (Hill 1934) showed that nearly 81% of the areas surveyed consisted of deforested and human-dominated landscapes. Another analysis indicated that more than 90% of its entire range has been replaced by houses, home gardens, townships, temples, schools, plantations, commercial operations and other areas of human activity. Deforestation has fragmented and drastically depleted the preferred habitat and principal food sources of the highly arboreal and folivorous *T. v. nestor*.

Within the fragmented and human-dominated landscape, *T. v. nestor* subsists mainly on fruits from home gardens (Dela 2007; Rudran 2007). The nutritional consequences of feeding on a low diversity diet mainly of cultivated fruits are unclear, but likely to be detrimental over the long term, because *T. v. nestor* is adapted to obtain its nutrients and energy from leaves with the help of a highly specialized stomach containing symbiotic bacteria (Bauchop and Martucci 1968). Given these specializations, relying on a diet of fruits instead of leaves may undermine the functioning of this monkey's gut fauna and thereby compromise its ability to absorb nutrients. Furthermore, fruits tend to occur seasonally, which means that *T. v. nestor* may not be able to fully satisfy its energy requirements outside the fruiting season. When such detrimental effects have the potential to affect this langur through most of its range, its long-term survival becomes an issue of serious concern.

Besides depleting *T. v. nestor*'s primary food source and preferred habitat, deforestation and fragmentation also cause

other problems for this monkey's survival. For instance, when fragmentation forces it to move on the ground, for which it is ill-adapted, people will occasionally capture young individuals to raise them as house pets. While on the ground *T. v. nestor* also runs the risk of being killed by domestic dogs or speeding vehicles. Death by electrocution is another source of mortality when it climbs onto power lines and electricity cables (Parker *et al.* 2008). In some parts of its range *T. v. nestor* is occasionally shot and killed while feeding in home gardens (Dela 2004). Deforestation and fragmentation indirectly lead, therefore, to a host of human-induced fatalities, which reduce group sizes and undermine social organization.

The long-term effect of extensive deforestation resulting in local extinctions was also evident during the recent survey. The western purple-faced langur was seen or recorded as present only in 43% of the sites surveyed in the eastern half of its historical range (N=23), and 78% of the survey sites in the western half (N=27). The sites where it was seen or recorded as present were interspersed between areas where it was absent or rare, suggesting the occurrence of local extinctions.

Although facing a perilous future, certain facts revealed during the recent survey indicate that it is still possible to save this monkey from disappearing forever. The largest forests it now inhabits (about 21 km² in all) are found around two reservoirs (Kalatuwawa and Labugama) that supply water to 1.2 million inhabitants of Sri Lanka's capital, Colombo. Because of their importance to people and their size, these forests are the last and most secure strongholds for maintaining viable populations over the long term. The Forest Department responsible for these forests has indicated interest in replanting the pine plantations in them with native species that are exploited by *T. v. nestor*. Such an initiative would certainly increase the extent of *T. v. nestor*'s preferred habitat, but it would first require a study of this langur's dietary preferences in the wild, which have yet to be studied.

Another important fact that surfaced during the survey was that the Forest Department has plans to promote forest conservation among communities living around its forests, through environmental education and nature tourism programs. Such programs can help conserve *T. v. nestor*, but to be effective they must be translated into action almost immediately.

Most people living within this langur's range were found to be Buddhists, who have a strong aversion to killing animals. The Buddhist taboo against killing may explain why this monkey has survived for as long as it has in such a densely populated area despite its reputation as an agricultural pest and a nuisance causing damage to roofs of houses and other properties. Sporadic killing does occur, however, as conflict between humans and monkeys intensifies (Nahallage *et al.* 2008), and poverty plagues the lives of the local people. Despite this situation, our survey revealed at least two forested sites around Buddhist monasteries where the incumbents strictly enforced the principles

of their faith and protected *T. v. nestor* and other wildlife. Hence soliciting the support of the Buddhist clergy and using cultural traditions to protect wildlife is a real possibility in Sri Lanka.

The above mentioned facts indicate that opportunities still exist for conserving *T. v. nestor*, despite the survival problems of this endangered endemic. The survey led to the development of a comprehensive plan for conserving *T. v. nestor* that includes three initiatives; public education, personnel training, and research. Because of the urgent need for conservation action, some elements of these initiatives were launched immediately after the survey despite the paucity of funds.

The public education initiative was launched at two sites that were identified as important for the long-term conservation of *T. v. nestor*, and targeted rural communities, particularly school children and their parents, living close to them. One site was around the Labugama-Kalatuwawa reservoirs where a viable population of *T. v. nestor* could be maintained over the long term, and the other was an area where human-monkey conflict was particularly intense. The educational activities at both sites were conducted with the support and participation of local Buddhist temples and clergy, and culminated in a public exhibition of conservation-oriented children's paintings and essays, at which the country's Minister for Environment and his top bureaucrat awarded prizes to the most talented youngsters. These events were publicized via newspaper articles and radio talk-shows to inform a much larger audience throughout the island that efforts to help conserve *T. v. nestor* were supported by the government and influential officials of the country.

The training initiative was launched with a series of activities designed to help a group of six trainees learn about the biology and identification of Sri Lanka's primates, birds and butterflies. Similar workshops dealing with plants, land snails, reptiles, amphibians and invasive species have been scheduled for the future. The primary objective is to train local youth, particularly those living around the Kalatuwawa-Labugama reservoirs, to become well-informed naturalists, who could work independently as nature guides or with us to help conserve *T. v. nestor*.

The research initiative remains dormant for the moment due to a lack of funds, but proposals have been submitted to address this shortcoming. When funds become available, research on *T. v. nestor*'s ecology and behavior, particularly its dietary preferences in the wild, will begin, and the work on the public education and training initiatives will be expanded. The battle to win the hearts and minds of people and to help ensure the survival of *T. v. nestor* has only just begun. Much remains to be done, and success can be achieved if this battle is sustained until current trends of deforestation are reversed, and people become more aware of the value of their wildlife.

*Rasanayagam Rudran, Kanchana Weerakoon &
Ananda Wanasinghe*

Gray-shanked Douc Monkey

Pygathrix cinerea Nadler, 1997

Vietnam, Cambodia (?), Laos (?)

(2000, 2002, 2004, 2006, 2008)

The colobine monkeys of the genus *Pygathrix* are native to Indochina. Until only ten years ago, just two distinct taxa were recognized: the red-shanked douc, *Pygathrix nemaeus* (Linnaeus 1771), in the northern part of Central Vietnam and Central Laos; and the black-shanked douc, *P. nigripes* (Milne-Edwards, 1871) from South Vietnam and east Cambodia. The gray-shanked douc was first described as a subspecies of the red-shanked douc, but genetic studies have since demonstrated a divergence at species level (Roos and Nadler 2001). It occurs in Central Vietnam between 13°30' and 16°N, and has been recorded in five provinces: Quang Nam, Quang Ngai, Kon Tum, Gia Lai, and Binh Dinh. Currently, gray-shanked doucs are known only from Vietnam, but records exist close to the border with Laos, and there are photos of hunted animals from south-east Laos and far north-east Cambodia that suggest that the species occurs in small neighboring areas in both countries. Surveys and research on this recently discovered primate have been conducted by the Vietnam Primate Conservation Program of Frankfurt Zoological Society, and the Endangered Primate Rescue Center at Cuc Phuong National Park.

Gray-shanked douc populations are fragmented, and estimated to total 600–700 individuals. Their occurrence has been confirmed in eight protected areas: Song Thanh Nature Reserve, Ngoc Linh Nature Reserve, Ba To Cultural and Historical Site, An Toan Nature Reserve, Kon Cha Rang Nature Reserve, Kon Ka Kinh National Park, Mom Ray National Park and A Yun Pa Nature Reserve. Hunting—the principal threat to the species—is, however, still a problem inside these parks and reserves. Snares are the most commonly-used method since gun confiscation programmes were carried out in a number of the areas. Often hundreds of traps are installed in trees frequently used by the monkey groups, as well as on the ground where they are seen crossing between small forest patches. Trapped animals are often severely injured and mutilated. Forest loss within at least part of the species' range is attributable to the expansion of agriculture, illegal logging and firewood collection. Almost 10,000 ha of forest are selectively logging every year in the Central Highlands.

The Endangered Primate Rescue Center has received 37 confiscated gray-shanked douc monkeys since 1995, and has begun a breeding program to provide stock for reintroduction in protected forests. Based on information from villagers and forest protection authorities, less than one-quarter of the hunted animals are confiscated alive. Frankfurt Zoological Society is studying the species in the Central Highlands of Vietnam, specifically to provide recommendations for the establishment of special "Species Protection Areas", which will promote connectivity between the currently-isolated populations in the established parks and reserves.

Ha Thang Long & Tilo Nadler

Tonkin Snub-nosed Monkey*Rhinopithecus avunculus* Dollman, 1912

Vietnam

(2000, 2002, 2004, 2006, 2008)

The Tonkin snub-nosed monkey is one of four unusual, large, Asian colobine monkeys of the genus *Rhinopithecus*, all of which possess a characteristic turned-up nose. The three other species are endemic to China, while the Tonkin snub-nosed monkey is found only in northeastern Vietnam. This species was discovered in 1911, collected on perhaps no more than two occasions over the course of the subsequent 50 to 60 years, and consequently presumed to be extinct by a number of primatologists until it was rediscovered in 1989. Historically the species occurs only east of the Red River between about 21°09'–23°N. Due to widespread deforestation and intensive hunting in recent decades, its distribution has become severely restricted.

Currently, there are only five known locations with recent evidence where Tonkin snub-nosed monkeys occur, and these are completely isolated. In 1992, a population was found in Na Hang District, Tuyen Quang Province. As a result of the discovery, a nature reserve was established in 1994. The nature reserve comprises two separate areas: the Tat Ke and Ban Bung sectors. A study in 1993 observed 72 individuals and estimated 80 in the Tat Ke sector, and observed 23 individuals and estimated 50 in the Ban Bung sector. A later study, in 2004–2005, found far lower densities, and estimated only 17–22 individuals in the Tat Ke sector; no estimation of numbers was possible for the other subpopulation in Ban Bung sector. The main threat to the monkeys in Tat Ke Sector was hunting. This may result from a hydropower and flood prevention dam project in Na Hang. Construction began in 2002, and some 10,000 workers moved into the area for dam construction. This created a number of access roads and a demand for wildlife products and firewood. Conservation activities carried out by several organizations have been unsuccessful, and it has resulted in a reduction of this population.

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. A survey in 2006 provided no sightings and no reliable evidence of the survival of the population. Local reports indicate, however, a small group of 8–12 individuals still in the area. The current threats to the populations of the monkeys are hunting and habitat destruction. Conservation efforts should target reducing human activities inside the reserve.

A population of about 60 Tonkin snub-nosed monkeys was discovered in 2001 and a later study (2005–2006) confirmed about 90 animals in Khau Ca, close to Du Gia Nature Reserve, Ha Giang Province. This is the only population which is not immediately threatened. There, public awareness and community participatory activities are being linked to increased protection efforts under the supervision of Fauna and Flora International (FFI).

A new population of about 20 Tonkin snub-nosed monkeys was discovered in a small forest patch in Tung Vai Commune of Quan Ba District close to the border with China. This is the second population of Tonkin snub-nosed monkey discovered in Ha Giang Province. The newly discovered population at Tung Vai appears to be threatened by hunting and habitat loss due to timber exploitation, shifting cultivation and the collection of non-timber forest products for commercial purposes. The immediate measures are likely to be training and establishing patrol groups, awareness-raising, more survey work to locate other groups and assess the range of the monkeys, and assessment of the impact of cardamom production on the habitat.

The total population of the Tonkin snub-nosed monkey is believed to be less than 200 individuals.

Le Khac Quyet, Dong Thanh Hai & Tilo Nadler

Eastern Black Crested Gibbon*Nomascus nasutus* (Kunkel d'Herculais, 1884)

China, Vietnam

(2008)

The eastern black crested gibbon occurs in a very restricted area along the Sino-Vietnam border, comprising only about 48 km², around 22°55'N, 106°30'E, including the northern Phong Nam-Ngoc Khe forests (about 30 km²) of Trung Khanh District, Cao Bang Province, Vietnam, and an immediately adjacent area (about 18 km²) in Jingxi County in South China's Guangxi Zhuang Autonomous Region (La Quang Trung and Trinh Dinh Hoang 2004; Chan Bosco Pui Lok *et al.* 2008).

In the past, the species was believed to comprise two subspecies (*N. nasutus nasutus* and *N. n. hainanus*), the first occurring in Vietnam and the second on China's Hainan Island. Both have now been elevated to full species, based initially on differences in territorial calls and fur coloration (La Quang Trung and Trinh Dinh Hoang 2004), but supported by genetic data (Roos *et al.* 2007). At the 2006 Asian Primate Red List Workshop in Cambodia, both were recognized as distinct species (Geissmann 2007; Chan *et al.* 2008). The historical range of the eastern black-crested gibbon was east of the Red River in China and Vietnam. It was thought to have gone extinct over its historical range in mainland China (Tan 1985), but was rediscovered recently in Bangliang Limestone Forest in Jingxi County; the population was estimated at 19 individuals, living in three groups (Chan Bosco Pui Lok *et al.* 2008). In Vietnam, it was also feared extinct until scientists from Fauna and Flora International (FFI) rediscovered a population in the limestone forest of Phong Nam-Ngoc Khe Communes in the northernmost Trung Khanh District, Cao Bang Province, northeast Vietnam, along the border with Guangxi. The population was estimated to be 26 individuals in at least five groups, based on a survey conducted in August 2002 (Geissmann *et al.* 2002, 2003), and 37 individuals in 8 groups in a survey in September 2004 (Trinh Dinh Hoang 2004). **Based on simultaneous surveys in September 2007 on both sides of the border, the total**

population of the gibbon is around 110 individuals living in 18 groups (Le Trong Dat *et al.* 2008). Thus, the species should be listed as Critically Endangered.

Nomascus nasutus inhabits lower montane and limestone forests in a wet tropical monsoon climate at elevations of 500–900 m (Geissmann *et al.* 2000). The main threat to this species, given its restricted range, is habitat loss and disturbance. The habitat of *N. nasutus* is in danger of being cleared for cultivation, pasture for livestock, and firewood collection by local Vietnamese, as well as charcoal-production by local Vietnamese and Chinese. The species is also endangered from problems intrinsic to small population size such as inbreeding effects, poor mate-choice, and human or natural disaster (La Quang Trung and Trinh Dinh Hoang 2004).

Conservation efforts on this species have been initiated in China since its rediscovery two years ago. Work on a proposed nature reserve to protect the gibbons, including comprehensive surveys and official document preparation, is now in progress, and the reserve will soon be in place. As for its conservation in Vietnam, Fauna and Flora International (FFI), along with Cao Bang FPD, is spearheading the establishment of a Species Conservation Area and a joint forest protection system that involves communities, a ranger force, and border patrol. FFI is also partnering with the Cao Bang Rural Development Project to encourage sustainability and conservation education and research in the local communities of the region (La Quang Trung and Trinh Dinh Hoang 2004). There is an urgent need to integrate the conservation efforts of both countries if the species is to be saved.

Long Yongcheng & Tilo Nadler

Western Hoolock Gibbon

Hoolock hoolock (Harlan, 1831)

Bangladesh, India, Myanmar

(2009)

Western and eastern hoolock gibbons were formerly in the genus *Bunopithecus* as two subspecies. In 2005, Mootnick and Groves placed them in a new genus, *Hoolock* as two distinct species, the western being *Hoolock hoolock* and the eastern, *Hoolock leuconedys*. The western hoolock gibbon occurs in India, Bangladesh and Myanmar, and the eastern hoolock gibbon in India, Myanmar and China.

The range of western hoolock gibbon is strongly associated with contiguous canopy, broad-leaved, wet evergreen and semi-evergreen forests. Hoolock gibbons are important seed dispersers, their diet including mostly ripe fruits, with some flowers, leaves and shoots.

Western hoolock gibbons face numerous threats, and now may be dependent on human action for their survival. Threats include habitat loss due to human encroachment, forest clearance for tea, slash-and-burn cultivation, hunting as food and medicine, and capture for trade. Additional threats include decline in forest quality which affects fruiting trees, canopy cover and the viability of their home ranges. Isolated populations face additional threats arising from intrinsic effects of small populations. Some populations

surviving in a few remaining trees are harassed by locals and dogs while attempting to cross clearings between forest patches in search of food.

Habitat loss over the last 3–4 decades suggests that western hoolock gibbons have declined from more than 100,000 to less than 5,000 individuals (a decline of more than 90%). The contiguous forests have borne the brunt of persistent human impacts. Isolated forest fragments hold a few families of about 1–4 individuals; numbers insufficient for long-term survival. Apart from some border forests between India and Myanmar, the remaining habitat is fragmented, holding minimal populations. The extirpation of western hoolock gibbons from 18 locations between 2001 and 2005 has been documented; ten in India and eight in Bangladesh.

About 100 locations of western hoolock gibbons have been recorded in India. In 2005, 77 of those locations had less than 20 individuals, and 47 of these had less than 10. A Population Viability Analysis (PVA) predicted a 75% decline in the population in India and a 95% decline in the population in Bangladesh over the next two decades, based on the current effects of human impacts.

Earlier estimates of western hoolock gibbons in Bangladesh were about 200 in 22 separate locations. Anwar Islam and his team conducted site visits in additional areas since then, and now estimate a total of about 300 individuals comprising 82 groups in 37 sites. In northeastern Bangladesh there are 12 sites with 102 hoolocks. The rest are in 25 sites in the southeast. There may be populations numbering 50–100 individuals in remote areas of the southeast hill tracts, but this has not been confirmed because of inability to visit these sites due to insurgency. During the last 15 or so years, hoolock gibbons have been extirpated from many sites, including Chunati Wildlife Sanctuary in the southeast. The extent of degradation and fragmentation of hoolock gibbon forests in the country is severe and the available habitats are continuing to decline.

The southernmost population of the western hoolock gibbon in Myanmar has been surveyed by Geissmann *et al.* confirming the presence and identification of western hoolock gibbon (*Hoolock hoolock*) in southern Rakhine Yoma, Myanmar, albeit a very small number. Reports of several other surveys in southern Myanmar are pending (Geissmann *et al.* 2008).

There may be much yet to learn about the distribution of the two species of hoolock gibbons. J. Das *et al.* identified the eastern species from Lohit district of Arunachal Pradesh, India, for the first time in 2005. Also, in a study conducted in the early months of 2009, D. Chetry found a new population of *Hoolock leuconedys* of around 150 groups between the rivers Dibang and Lohit in Lower Dibang Valley District of Arunachal Pradesh, India.

Warren Brockelman has carried out surveys of the eastern hoolock, *Hoolock leuconedys* in two accessible protected areas east of the Chindwin River in Myanmar since 2005. Recent studies in Mahamyaing Wildlife Sanctuary, western Myanmar, using auditory sampling of groups, produced an

estimate of about 6,000 individuals and a mean density of more than 2 groups/km² in areas of suitable forest. Preliminary analysis of a survey by WCS–Myanmar and Wildlife Department personnel farther north in the Hukaung Valley (Kachin State) suggested that thousands of hoolocks survive there also. The Hukaung Valley Wildlife Sanctuary includes the headwaters of the Chindwin River and is contiguous with areas in India. The area of evergreen forest in the Hukaung Valley Reserve and contiguous PAs is so large (more than 20,000 km²) that the population there is likely to be in the tens of thousands. If so, this represents the largest population of hoolocks anywhere. Nevertheless, these PAs are not well protected and it is hoped that current interest in conservation in this multiple-use area will be sustained.

Eastern hoolock gibbons also occur in China. According to Fan Pengfei, a Chinese field biologist, the Chinese eastern hoolock gibbons survive only in Gaoligongshan Nature Reserve (GNR) in Baoshan, Tengchong, and Yingjiang. Based on field surveys, population size in GNR was estimated to be 20–21 groups. There are about 15 groups living outside Gaoligongshan Nature Reserve (based on interviews). The total population size is estimated to less than 150 individuals and is severely fragmented. The largest subpopulation in Yunnan has 8–10 groups; the second largest subpopulation has four groups, and in several sites there are only single groups. Twenty years ago researchers estimated the population size of hoolock gibbons to be less than 200. This was a low estimate due to failure of research to cover all distribution areas. The hoolock gibbon is threatened by poaching in some places and by habitat degradation and fragmentation outside GNR. There are no records of western hoolock gibbons in China to date.

There has been serious concern about the survival of hoolock gibbons for some decades. The species was listed on Schedule I, the highest schedule, on the Indian Wildlife (Protection) Act in 1972. It is categorized as Endangered on the IUCN Red List. The western hoolock gibbon was designated as one of the top 10 threatened gibbon taxa of the world in a Resolution taken in the gibbon symposium of the Congress of the International Primatological Society at Beijing in 2002.

Hoolock gibbons were assessed along with other South Asian primates at a Conservation Assessment and Management Plan workshop held in Coimbatore in 2002. Participants from northeastern India and Bangladesh assembled detailed locality tables which painted a bleak picture for western hoolock gibbons. Participants recommended that a Population and Habitat Viability Assessment (PHVA) Workshop should be conducted for the species. In 2005, a PHVA workshop was conducted for *Hoolock hoolock* in Dhaka, Bangladesh. Among other recommendations, workshop participants suggested that small, isolated, doomed individuals and groups in degraded areas should be translocated to more supportive habitat within their range.

The level of local knowledge required to conduct successful wild-to-wild translocations needed supplementation,

so a collaborative initiative between GOs and NGOs in India and Bangladesh for scoping and training in translocation was organized. Two workshops, held in September 2008 for all stakeholders from India and Bangladesh, and February 2009 for senior foresters or their representatives from India generated a great deal of interest as well as a new awareness of the subtleties of such an exercise. Tentative plans were made for each state at the workshop. Arunachal Pradesh has taken the initiative and engaged the Wildlife Trust of India to assist them with an exercise for several isolated groups in an agricultural field in the state. Other northeastern Indian states and Bangladesh are also considering conducting carefully planned and executed translocations. The CAMP, PHVA and translocation training workshops also generated considerably more public awareness activities on hoolock gibbon that are now taking place very regularly, which will be useful also to the translocations when they occur.

There are hundreds of western hoolock gibbons languishing as single individuals or in minute groups in the northeastern Indian states and in Bangladesh. Successfully translocating these to more viable locations in nearby larger areas with resident, established hoolock populations will not only enrich the gene pool and strengthen populations but also salvage animals and their genetic material that would not otherwise survive even a very few years. Such an exercise will also provide a platform with a remarkable profile for enhancing protection as well as for reclaiming and restoring forest patches to create more contiguous habitat for hoolocks. It should also create good will and interest by the public, whose cooperation is necessary for long-term success. However, such exercises should be undertaken with strict adherence to the IUCN/SSC Reintroduction Specialist Group (RSG) reintroduction guidelines. They should also be a “last resort”, after exploring all other means of conserving both habitats and species, working with locals in the current areas.

The population trends for the western hoolock gibbon observed over recent years in Bangladesh and northeastern India indicate a very rapid decline in numbers for which very little has been done in the way of mitigation. Immediate measures are required by governments, forest departments, local communities and NGOs to limit habitat destruction, initiate or improve habitat restoration and upgrade implementation of protective measures. Although there are indications of increased numbers in this report, it is only because more localities or areas are being visited and found to have hoolock gibbons sometimes in significant numbers. This should not, in any way, lead to complacency but to greater efforts to see that the threats which have plagued the hoolock gibbon in the past 3–4 decades are addressed and contained.

*Sally Walker, Sanjay Molur, Warren Y. Brockelman,
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Sumatran Orangutan

Pongo abelii Lesson, 1827

Indonesia (Sumatra)

(2000, 2002, 2004, 2006, 2008)

Sumatran (*Pongo abelii*) and Bornean (*P. pygmaeus* Linnaeus, 1760) orangutans are now considered to be two distinct species, comprising the genus *Pongo*. Three subspecies are recognized for *P. pygmaeus*, but the Sumatran orangutan is a single taxonomic unit. The long-term viability of the entire genus is in question, but the Sumatran orangutan faces the more immediate threat of extinction and is listed as Critically Endangered on the IUCN Red List of Threatened Species.

The species is endemic to Sumatra, Indonesia. Truly wild populations are restricted to the remaining lowland forests of the two most northerly provinces of the island, Nanggroe Aceh Darussalam (NAD) and North Sumatra. A small reintroduced population is also currently being established in Jambi Province, further to the south.

About 6,600 wild individuals remain (based largely on nest density surveys and 2002 satellite imagery). They survive in just 10 fragmented habitat units stretching from the central regions of NAD, south to the Batang Toru River in North Sumatra, with a notable gap in their distribution immediately west of Lake Toba. The southernmost populations in North Sumatra could be genetically and culturally distinct from their more northern relatives due to isolation. The largest populations occur within Nanggroe Aceh Darussalam, where until 2005 a separatist conflict made monitoring and conservation work problematic. Recent surveys appear to have confirmed the absence of orangutans in the northernmost forests of NAD such that almost all orangutans in Aceh can be found within what is known as the Leuser Ecosystem.

The Leuser Ecosystem is a 26,000 km² conservation area established by presidential decree that encompasses the smaller Gunung Leuser National Park (10,950 km²; itself part of the Sumatran Rainforest World Heritage Site) and the 1,025 km² Singkil Swamps Wildlife Reserve. About 5,800 orangutans are considered to remain in the Leuser Ecosystem. The Leuser Ecosystem, and the smaller National Park and Wildlife Reserve within it, forms the only conservation area where viable wild populations of the Sumatran orangutan, Sumatran tiger, Sumatran rhinoceros and Sumatran elephant, each of which is endangered in itself, still occur side by side. The National Park, however, mostly comprises high mountains, and as the orangutan is predominantly a lowland species, rarely being found above 1,000 m above sea level, the majority of orangutans are found within the larger Leuser Ecosystem but outside of the National Park itself. For example, the Ecosystem harbors c.88% of the remaining 6,600 Sumatran orangutans whilst only 30% are found within the National Park and 23% within the Singkil Swamps Wildlife Reserve.

Throughout its range, the primary threat to the Sumatran orangutan is habitat conversion and fragmentation. Logging, both legal and illegal, often leads to total conversion of forests for agriculture or oil palm plantations. Roads are also a constant threat, since they further fragment already

declining populations and also give access for additional logging and encroachment. Although precise rates of forest loss are difficult to determine, primary lowland forests in Sumatra have been devastated over the last 30 years. One study of forest cover concludes 301,420 ha, or 13% of the original 2,284,771 ha of forests, were lost in North Sumatra Province alone between 1990 and 2000 (Gaveau *et al.* 2007). A second analysis, more focused on orangutan habitat in Sumatra concluded that habitat supporting around 1,000 orangutans was being lost each year in the Leuser Ecosystem alone during the late 1990s (van Schaik *et al.* 2001). This was largely due to legal logging concessions and conversion of lowland forests to oil palm estates, but also to illegal logging and encroachment in some places.

Fortunately, the rate of habitat loss decreased markedly in many areas during the Aceh civil conflict, as even loggers did not consider it safe to work in the forests. In fact, Gaveau *et al.* (2007) found that satellite data indicated that the rate of loss was five times faster in Aceh between 1990 and 2000 (294 km² or 0.75% per year) than it was between 2000 and 2006 (58 km² or 0.15% per year). Orangutan populations have nevertheless plummeted in those regions that have still been affected by logging. Even small-scale selective logging can reduce local orangutan densities by as much as 60% in Sumatra (Rao and van Schaik 1997).

Encroachment and conversion, especially by settlers fleeing the conflict in NAD and migrants from Nias Island, also accelerated habitat loss in some parts. After the 2004 tsunami many people moved from coastal areas, and the subsequent increase in demand for timber still poses a significant threat. Several new roads (part of a project known as Ladia Galaska) have also begun further fragmenting remaining orangutan habitat. Proposed new roads are a particular concern in the Singkil Swamps Wildlife Reserve, especially as Sumatra's peat swamp forests support the highest densities of orangutans in the world. This is expected to become a major problem in coming years as illegal loggers and settlers gradually move in and open up new agricultural land. Throughout their range, orangutans are sometimes killed as pests at the forest edge as they raid agricultural crops (particularly highly prized fruits such as durian), and in parts of North Sumatra Province they are occasionally still hunted for food. A small yet still significant trade in young Sumatran orangutans as pets also persists.

Key conservation interventions rely heavily on a dramatic and rapid improvement in enforcement of wildlife and forest laws and far greater consideration for environmental issues in spatial planning decisions. Implementing patrols, improving law enforcement (especially the number and frequency of cases actually prosecuted), stopping illegal logging, halting legal logging and forest conversion to plantations, promoting forest restoration, halting road construction, addressing human-orangutan conflict, and providing connectivity in the landscape to allow for genetic exchange are all seen as prerequisites for the species' survival. There is some cause for optimism, however. The Indonesian government has developed a National Strategy and Action Plan for Orangutan

Conservation 2007–2017 (DitJen PHKA 2007) and the Government of NAD has also imposed a moratorium on all logging in the Province. Nevertheless, as with so many plans and laws, if not strictly followed and enforced, both could result in little or no change from business as usual. Indeed, if pre-civil conflict rates of habitat loss resume in NAD and the protected status of remaining habitat outside of the Leuser Ecosystem is not quickly enhanced, we could see a further 50% of Sumatran orangutans vanish within a decade. Effective long-term solutions to conserve northern Sumatra's remaining lowland primary forests are still urgently needed.

Ian Singleton, Jatna Supriatna & Serge A. Wich

Neotropical Region

Cotton-top Tamarin

Saguinus oedipus (Linnaeus, 1758)

Colombia
(2008)

Cotton-top tamarins are Critically Endangered and found only in northwestern Colombia. They have an extremely limited distribution, occurring in northwestern Colombia between the Río Atrato and the lower Río Cauca (west of the Río Cauca and the Isla de Mompos) and Río Magdalena, in the departments of Atlántico, Sucre, Córdoba, western Bolívar, northwestern Antioquia (from the Uraba region, west of the Río Cauca), and northeastern Chocó east of the Río Atrato, from sea level up to 1,500 m (Hernández-Camacho and Cooper 1976; Hershkovitz 1977; Mast *et al.* 1993). The southwestern boundary of the cotton-top's range has not been clearly identified. Mast *et al.* (1993) suggested that it may extend to Villa Arteaga on the Río Sucio (Hershkovitz 1977), which included reports of cotton-top tamarins in Los Katios National Park. Barbosa *et al.* (1988), however, were unable to find any evidence of cotton-top tamarins in this area or in Los Katios, where they saw only *Saguinus Geoffroyi*. Groups have been seen in the Islas del Rosario and Tayrona National Park in the Sierra Nevada de Santa Marta (Mast *et al.* 1993; A. Savage and L. H. Giraldo pers. obs.). However, these populations were founded by captive animals that were released into the area (Mast *et al.* 1993), and we believe to be outside the historic range of the species.

Colombia is among the top ten countries suffering deforestation, losing more than 4,000 km² annually (Myers 1989; Mast *et al.* 1993). There are just three protected areas in the historic range of the cotton-top tamarin—Parque Nacional Natural Paramillo (460,000 ha), Santuario de Flora y Fauna Los Colorados (1,000 ha) and Montes de María Reserve (7,460 ha). These protected areas have lost 42%, 71%, and 70% of their forests, respectively, since they were created (Miller *et al.* 2004). Cotton-tops can also be found in forest patches on private land, but there they lack the long-term protection of their natural resources. Land use in the region is dominated by large-scale agricultural production (cattle) and farming. Forest remnants can be found only where the land is

unfit for agriculture, and their long-term survival, buffering agricultural zones, is constantly threatened.

The extraction and exploitation of natural resources is constant in Colombia's Pacific coastal region. The Plan Pacífico (see Barnes 1993) entails that 160,000 ha (approximately 2.2% of the total forest area) are destroyed each year for wood and paper or to make way for agro-industrial production of African palm. There has been a considerable drop in mangrove coverage with the installation of commercial shrimp farms, and massive sedimentation and mercury contamination in rivers has been caused by deforestation and uncontrolled mining. Riverbanks have also been eroded, which has caused river beds to drop, threatening fish stocks and the ability of communities to transport goods (Barnes 1993).

Further threat lies in the imminent flooding of forests for hydroelectric projects. One of these, the Urra I dam, inundated more than 7,000 ha of primary and secondary forest in the Parque Nacional Natural Paramillo. The environmental impacts of the dam were seriously damaging for local communities and wildlife. The construction of Urra II was proposed in 2008, and if approved by the Colombian government, it will result in the destruction of a further 5,000 ha of forest in the park.

In the late 1960s and early 1970s, 20,000–40,000 cotton-top tamarins were exported to the United States for use in biomedical research (Clapp 1972; Hernández-Camacho and Cooper 1976). Today, cotton-top tamarins continue to be threatened by capture for the illegal pet trade, despite international laws condemning the activity. A recent population census was conducted in the historic distribution of the species that documented a dramatic decline in suitable habitat, and concluded that fewer than 6,000 cotton-top tamarins remain in the wild (Savage *et al.* in review a). Large expanses of forest (500 ha or more) that could support viable cotton-top tamarin populations do not now exist in the departments of Atlántico and Bolívar. What remains are numerous small, isolated forests with tiny remnant populations of cotton-tops. Dispersal opportunities for these animals are limited as the forest patches are surrounded by open land such as cattle pasture. Efforts to protect these forest patches, while creating corridors, are essential to ensure the survival of this Critically Endangered species.

To aid in the conservation of the cotton-top tamarin, we established Proyecto Tití (<www.proyectotiti.com>), a multi-disciplinary, *in situ* conservation program that combines field research, education initiatives and community development for the conservation of natural resources that is economically feasible for local communities in Colombia. The program works with national and international organizations to assist in the long-term preservation of the cotton-top tamarin and to develop local community advocates to promote conservation efforts in Colombia. Early studies (1988 at Colosó in the Montes de María reserve) revealed that there were many myths and misconceptions about the forest and the wildlife. More than 90% of the population we surveyed had no idea that cotton-top tamarins were endemic

to Colombia and not found in other countries (Savage *et al.* 1997). We developed classroom and field activities for elementary and secondary school children that were designed to create an awareness of the plight of the cotton-top tamarin and engage students in a variety of activities in the classroom and field, and in international exchanges that would promote the conservation of Colombia's natural resources (Savage 1993, 1997; Savage *et al.* 2000a, 2000b; Giraldo *et al.* 2003). Our education program continued to expand to include teacher-training programs, the establishment of a rural school dedicated to conservation and sustainable farming practices, and field training for Colombian university students. We developed a strong partnership with the Baranquilla Zoo, and we now reach urban audiences through a series of classroom workbooks (CARTITILLA) aimed at 5–7th grade school children (Guillen 2003). Urban communities were limited in their understanding of wildlife conservation issues and were the primary market for the illegal pet trade of cotton-top tamarins. The workbook focused on the cotton-top tamarin and its tropical ecosystem including knowledge-based activities, interactive games, role-playing scenarios, and inquiry-based questions that would lead students to a conservation-based discovery. It was used in 15 schools with more than 3,000 students. Our evaluations showed an 81% increase in the level of accuracy on correctly identifying a cotton-top tamarin, a 77% increase in understanding that cotton-top tamarins are found only in Colombia, and a 65% increase in the understanding of the pet trade as a threat to the survival of the species. Regional pride was instilled in these students so that they were more interested in exploring opportunities that would help to protect cotton-top tamarins for the future (Guillen 2003). Our extensive education program has created knowledgeable individuals that are concerned for the environment.

However, pressing economic issues created a disconnection between our efforts to educate communities to conserve natural resources and their ability to engage in activities that promoted wildlife conservation. In discussions with local villagers in Colombia we discovered the traditional Colombian “binde”, a small cooking stove that was made from a termite mound (Savage *et al.* 1997). Interviews with local villagers indicated that bindes required less firewood than cooking over an open fire. While accepted by local communities in Colombia, bindes were made from termite mounds and they would quickly crack and disintegrate with repeated use and were consequently little favored. Proyecto Titi designed a durable binde made of clay that was readily accepted by the communities and proved to significantly reduce the amount of firewood consumed. A family of five used approximately 15 logs a day to cook their food over an open fire. Using a binde, the number of logs consumed each day was reduced by two-thirds (Savage *et al.* 1997). Food cooked in a binde did not take significantly longer to cook than over open fire, and it retained its flavor better. Since bindes produce less smoke, women reported less eye and lung irritation than when cooking over an open fire (Savage

et al. 1997). Binde proved to be a successful tool in reducing the amount of trees harvested for firewood, besides improving the health of the villagers.

Efforts to manage waste are a challenge in local villages, and the situation is worsening, particularly in growing rural communities where disposal is generally by burning or by dumping in rivers or on the roadside. Enormous amounts of plastic appear in the forests; waste which animals tend to investigate or eat, resulting in disease transmission between humans and wildlife. A program was developed to turn the trash into a source of income. The goal was to create an artisan group that would make a product from the numerous plastic bags, so as to provide a stable income that, combined with effective conservation education messaging, would result in a commitment to protect the forests, and reduce the capture of cotton-top tamarins for the illegal pet trade. Proyecto Titi first engaged the village of Los Limites (population of 250) in protecting cotton-top tamarins and their habitat by helping it with the confection of tote bags crocheted with recycled plastic bags and called “eco-mochilas” (Savage *et al.* in review b). Fifteen women—heads of households and well-respected in their community—began the initiative, and were so successful it was necessary to provide business training as they became established entrepreneurs, developing products of a quality that sells in national and international markets. ASO-ARTESANAS was created in 2004 with 15 founding members and a five-person board of directors.

Proyecto Titi demonstrated a clear economic benefit to individuals that participate in community empowerment programs and produced tangible results that are contributing to the survival of the cotton-top tamarin in Colombia. To date, ASOARTESANAS has trained more than 600 women and recycled nearly 1.5 million plastic bags, and continues to reach out to communities and cities to assist in the collection of plastic bag litter, which has decreased in rural communities and is now rarely seen in the forest. This has had positive implications in reducing human and wildlife health concerns in the region, and has been positive for the cotton-top tamarin in the cessation of their trade as pets and in protecting their habitats through a substantial reduction in the number of trees harvested for firewood.

Anne Savage, Luis Soto, Iader Lamilla &
Rosamira Guillen

Variegated or Brown Spider Monkey

Ateles hybridus I. Geoffroy, 1829

Colombia, Venezuela
(2004, 2006, 2008)

There are two recognized subspecies of the variegated or brown spider monkey. *Ateles hybridus brunneus* Gray, 1870 is restricted to Colombia, occurring between the lower Ríos Cauca and Magdalena in the Departments of Bolívar, Antioquia and Caldas. *Ateles h. hybridus* occurs east from the right bank of the Río Magdalena extending into western Venezuela. Both subspecies are Critically Endangered due to habitat loss, hunting and the pet trade.

The large size, slow reproductive rate (single offspring at 3–4 year intervals) and generally low population densities of spider monkeys make them especially vulnerable to hunting. Historically, *A. hybridus* has suffered from habitat destruction, and only 0.67% of the current remaining range of *A. hybridus* is protected; most has been converted to farms for agriculture and cattle.

Ateles h. brunneus has a small geographic range in a region where forest loss, degradation and fragmentation is widespread. Currently, the remaining populations are surrounded by human populations, compounding the already high level of threat. Only 9% of their potential range remains as continuous forest. This subspecies has been identified in different areas of the Antioquia department, such as: Segovia, Remedios, Maceo, Yondó, Puerto Berrio, and Zaragoza. However, the habitat destruction and hunting pressure over this species has provoked possible local extinctions. Between 2007 and 2008, surveys in this Department yielded some possible areas where this species no longer exists (for example, Vereda El Brazil, Corregimiento La Sierra). Surveys have been conducted to determine the density of this subspecies in the municipality of Maceo. In 2006 one group of eight individuals were found in this area, and by 2008 just four individuals were spotted in the same area, after five months of surveys. A refuge remains, however, in the Serranía San Lucas in southern Bolívar, and in some parts of Nechí, identified as important areas for the establishment of national parks. A protected area is highly necessary for this subspecies that also would include two other threatened endemic primates, the white-footed tamarin, *Saguinus leucopus*, and the woolly monkey, *Lagothrix lugens*.

Ateles h. hybridus is extremely endangered due to habitat destruction in both Colombia and Venezuela. The lowland forest of the state of Zulia and the piedmont of the Perijá Mountains are heavily destroyed from expansionist cattle-ranching activities. Within the Perijá Mountains only 30% of the forest is relatively well preserved and protected. The rest is affected by rapid human expansion and land clearing, poor protection and increasing fragmentation, putting potential corridors at risk in most of its extent. Also in the Perijá Mountains, brown spider monkeys seem to be favorite game. In central Venezuela, some areas that had populations in 2001 were resurveyed in 2007 without successful sightings; most of the areas were already covered by secondary vegetation. The lowland forest from the eastern part of the Andean Mountains, San Camilo and Ticoporo, are under severe logging pressure.

Ateles hybridus can be found in at least six zoos in Colombia, presenting problems of surplus animals and consanguinity. This species is suffering also from the pet trade; about 20 confiscated individuals are currently in residence in four rescue centers and need to be relocated. There is an urgent need for surveys to establish areas with populations of this species and to propose conservation measures. An *ex situ* breeding program is also necessary to maintain healthy and viable captive populations.

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Peruvian Yellow-tailed Woolly Monkey

Oreonax flavicauda (Humboldt, 1812)

Peru

(2000, 2006, 2008)

The taxonomy of the yellow-tailed woolly monkey has been a matter of some discussion. First described as *Simia flavicauda* by Humboldt in 1812, it was again described by Thomas (1927a) as *Lagothrix (Oreonax) hendeii* a century later. Later in the same year, after receiving a new juvenile specimen, Thomas (1927b) elevated the subgenus *Oreonax* to full generic status. In his revision of the woolly monkeys, Fooden (1963) found that *S. flavicauda* and *O. hendeii* were actually the same species and very closely related to *Lagothrix*, and he thus named it *Lagothrix flavicauda*. Groves (2001) revised some available skulls and found it more closely related to *Ateles*, and consequently separated *flavicauda* from *Lagothrix*, and revived Thomas' old genus *Oreonax*. Most recently, Matthews and Rosenberger (2008a, 2008b) revised Groves' work and found evidence for a "misclassification because a heuristic measure of statistical support has been misconstrued as a biological and phylogenetic characteristic", and therefore argued against the validity of *Oreonax* as a genus. A more comprehensive reassessment of the systematics of *Lagothrix* is still needed, using a wider set of characters and samples, both in morphology and molecular genetics.

The Peruvian yellow-tailed woolly monkey is endemic to Peru, and is found only in a small area in the Tropical Andes. *Oreonax flavicauda* is known to persist only in primary premontane, montane and cloud forest between 1,500 to 2,700 m above sea level (Leo Luna 1982; Butchart *et al.* 1995; DeLuycker 2007; Shanee *et al.* 2008). Historically, the distribution of the species may have included the regions of Amazonas, San Martín, Huánuco, Loreto and La Libertad, as predicted by the species distribution modeled by Pacheco *et al.* (2007). Now the species is restricted to scattered forests in only two regions—Amazonas and San Martín (Heymann and DeLuycker 2007; Shanee *et al.* 2008). There are no current estimates of remaining population numbers. Indiscriminate clear-cutting of primary cloud forest is the principal threat to this species, and its habitat has been largely deforested, resulting in a strongly fragmented landscape.

We estimated the extent of the historical distribution area of *O. flavicauda*, based on a model without taking into account current deforested areas and human settlements, to be 41,446 km². In 1981, it was estimated that the potential forested habitat was at least 11,240 km² and it was predicted that at least 1,600 km² would be deforested for agriculture by 1991 (Leo Luna 1984). With a modeled distribution using known localities and suitable habitat, we estimate the current potential distribution of *O. flavicauda* to be 7,690 km², a number that is rapidly diminishing due to a high rate of human immigration to the area, combined with unregulated land use. In addition, much or most of this forest is now highly fragmented or isolated from other tracts of forest. *Oreonax flavicauda* has likely declined drastically in numbers due to a major reduction in area of occupancy and a decrease in the quality of their habitat.

Very little is known about the ecology and behavior of the yellow-tailed woolly monkey. Results from studies in the early 1980s indicated that the sizes of its multi-male/multi-female groups range from 5 to 18 individuals. *Oreonax flavicauda* eats a variety of fruits, flowers, leaves, lichens, leaf bases of bromeliads, epiphyte roots and bulbs, and possibly insects (Leo Luna 1982; DeLuycker 2007). Surveys in the Amazonas region found groups ranging from 7 to 10 individuals (Cornejo *et al.* 2007), but DeLuycker (2007) reported an unusually large group (17–20 individuals) in an area relatively close to agricultural plots. The species appears to be very sensitive to habitat disturbance (Leo Luna 1987; DeLuycker 2007). Where the forest is disturbed by logging, *O. flavicauda* decreases its use of the area (Leo Luna 1984), often retreating further into high-altitude forests far away from human settlement, where it is able to use larger tracts of forest. In 1981, it was estimated that *O. flavicauda* occurred in low densities, from 0.25 to 1 group per km² (Leo Luna 1987). Recently, a survey conducted in a forest fragment provided an estimate of 1–2 groups per km² (Cornejo 2007). Based on the difficulty of locating groups of *O. flavicauda* during an intensive three-month survey, DeLuycker (2007) suspected this species to have large home ranges (as do other atelins), but Cornejo (2008) estimated the home range of a single group as only 69 ha.

The species is known to be present in the Río Abiseo National Park (PNRA) (2,745 km²), the Alto Mayo Protected Forest (BPAM) (1,820 km²), and the Reserved Zone Cordillera de Colán (ZRCC) (641 km²), all of which were established with assistance from the Asociación Peruana para la Conservación de la Naturaleza (APECO). Between 1996 and 2001, more than 6,000 ha of primary forest were cleared inside the BPAM (Peru, INRENA 2008). The forest of the BPAM is now considerably fragmented, a result of lack of enforcement and a substantial human population living in the protected forest itself. The “Reserved Zone” Cordillera de Colán (ZRCC) is finally being categorized as a National Sanctuary and a Community Reserve of Awajun Natives, after many years of being without a formal categorization and a management plan. BPAM and ZRCC also suffer from illegal selective logging—ZRCC has two operative mining concessions near its borders, and both areas have the constant threat of human unregulated migration. *Oreonax flavicauda* has been extirpated from all but the most distant and isolated forests on the eastern side of the Río Alto Mayo. Illegal hunting still occurs within and outside protected areas, and if monkeys are encountered, they are likely to be shot, because of their size, conspicuousness, and trustful behavior toward humans. The species’ velvety, thick, long fur, its skin and skull, and yellow genital hair-tuft are sought after as trophy items, and make this species a target for hunters even when they do not hunt it for subsistence. Infants taken when their mothers are shot are sold in markets as pets. PNRA is the only governmental protected area that, because of its inaccessibility, is actually protecting the yellow-tailed woolly monkey. Unfortunately, PNRA is only protecting 852 km² of suitable habitat for the species (M. Leo Luna unpubl. data).

There is very little information on the biology and natural history of this species, resulting mainly from the difficulties imposed by the mountainous and precipitous terrain where it lives. A complete, range-wide survey of its cloud forest habitat is urgently needed to develop plans to protect the remaining populations of *Oreonax flavicauda*. These surveys should also include population genetic studies, to examine genetic variability and the viability of existing populations.

Currently, a number of institutions are investing efforts and resources in northeastern Peru’s cloud forests. Some community-based conservation projects are underway (Ucumari, Apenheul, Neotropical Primate Conservation [Shanee *et al.* 2007, 2008] and the Museo de Historia Natural–UNMSM in Amazonas), protected area policies and management plans are being enforced (APECO and Deutsche Gesellschaft für Technische Zusammenarbeit–GTZ), private reserves are being established (Asociación Ecosistemas Andinos, Sociedad Peruana de Derecho Ambiental), and conservation education campaigns are being held (Yunkawasi). While these conservation efforts have already produced some positive results, they are not enough. The regions of Amazonas and San Martín have the highest rates of deforestation of Peru (Reategui and Martínez 2007)—the product of very deep social conflicts in the area, with illegal logging and illegal land traffic being the main problems.

Urgent conservation initiatives necessary for the yellow-tailed woolly monkey’s survival should continue and include: increased protection within designated parks, reserves, and protected forests, which currently lack enforcement; the establishment of a contiguous area of protected forest, to create a biological corridor; control of illegal logging; purchase of land; the provision of alternative economic models for local communities living along buffer zones, in order to prevent further migration into the primary cloud forests; and the implementation of a strong conservation education plan.

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Madagascar

Greater Bamboo Lemur

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Sclater's Black Lemur, Blue-Eyed Black Lemur

Eulemur flavifrons

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Africa

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Neotropical Region

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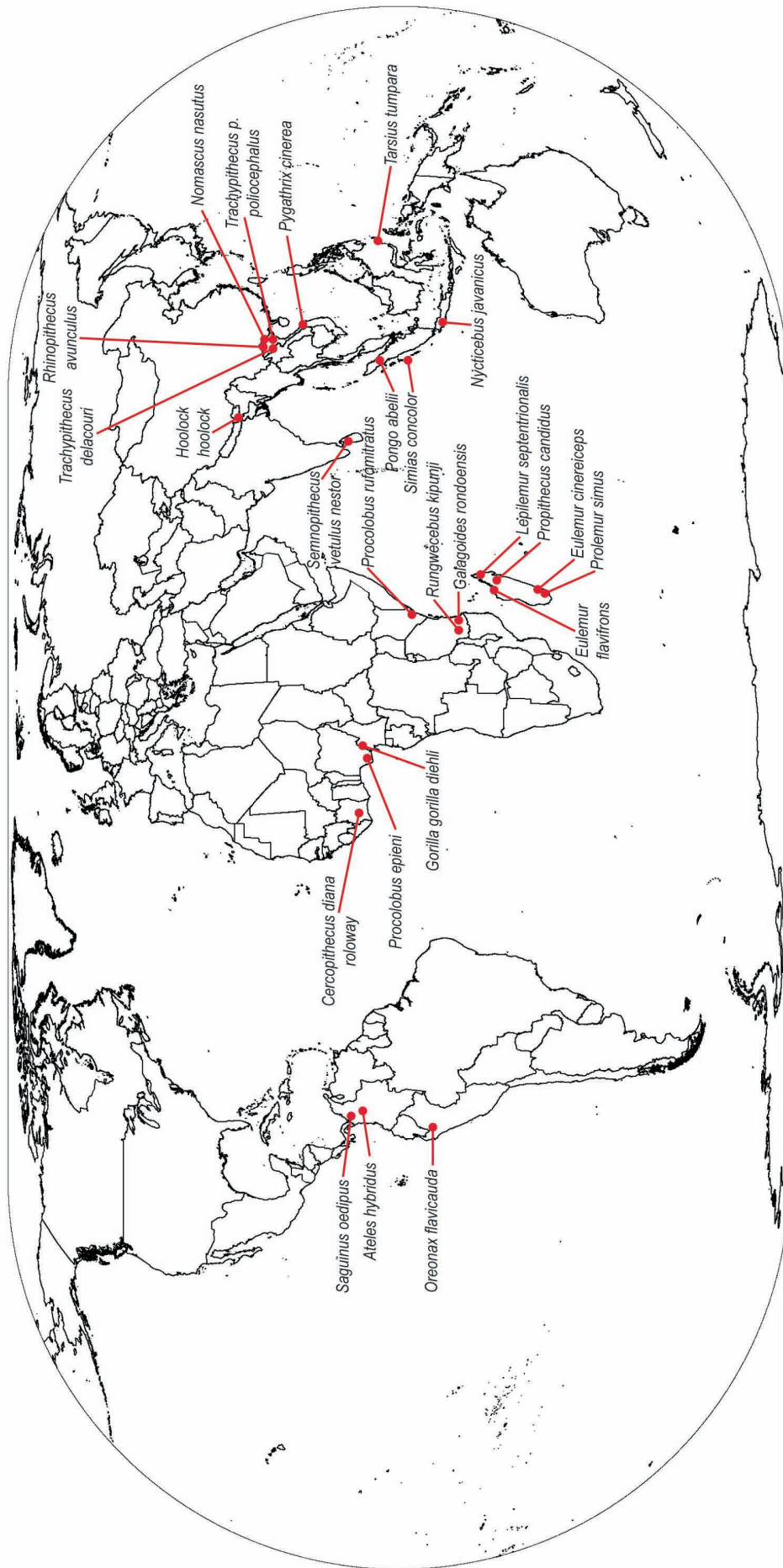
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The World's 25 Most Endangered Primates 2008–2010



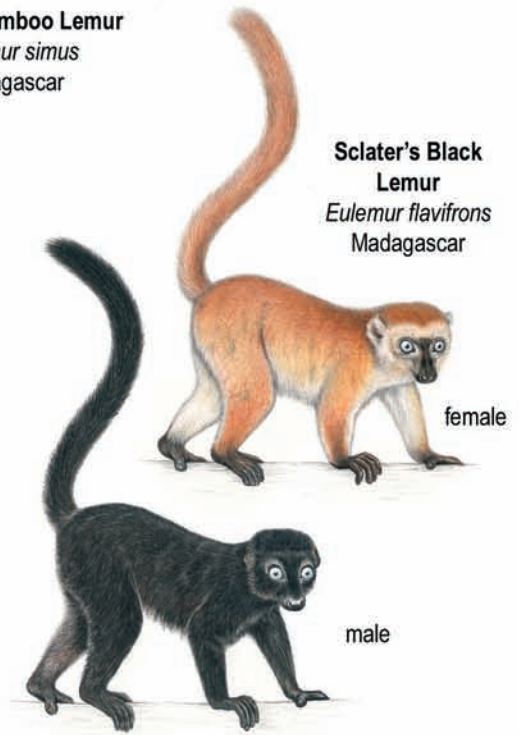
**White-collared
Brown Lemur**
Eulemur cinereiceps
Madagascar



Greater Bamboo Lemur
Prolemur simus
Madagascar



**Sclater's Black
Lemur**
Eulemur flavifrons
Madagascar



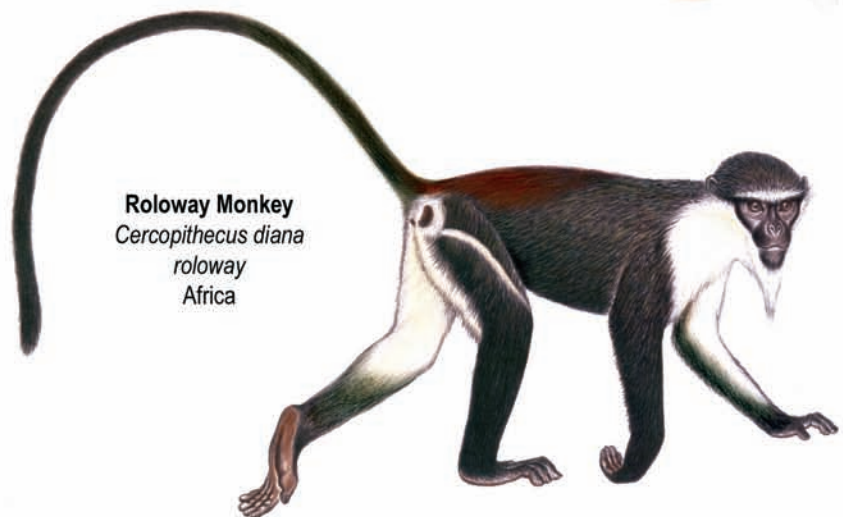
**Northern
Sportive Lemur**
*Lepilemur
septrionalis*
Madagascar



Rondo Dwarf Galago
Galagoides rondoensis
Africa

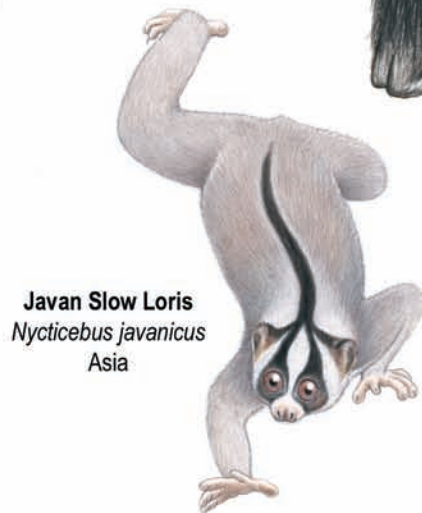
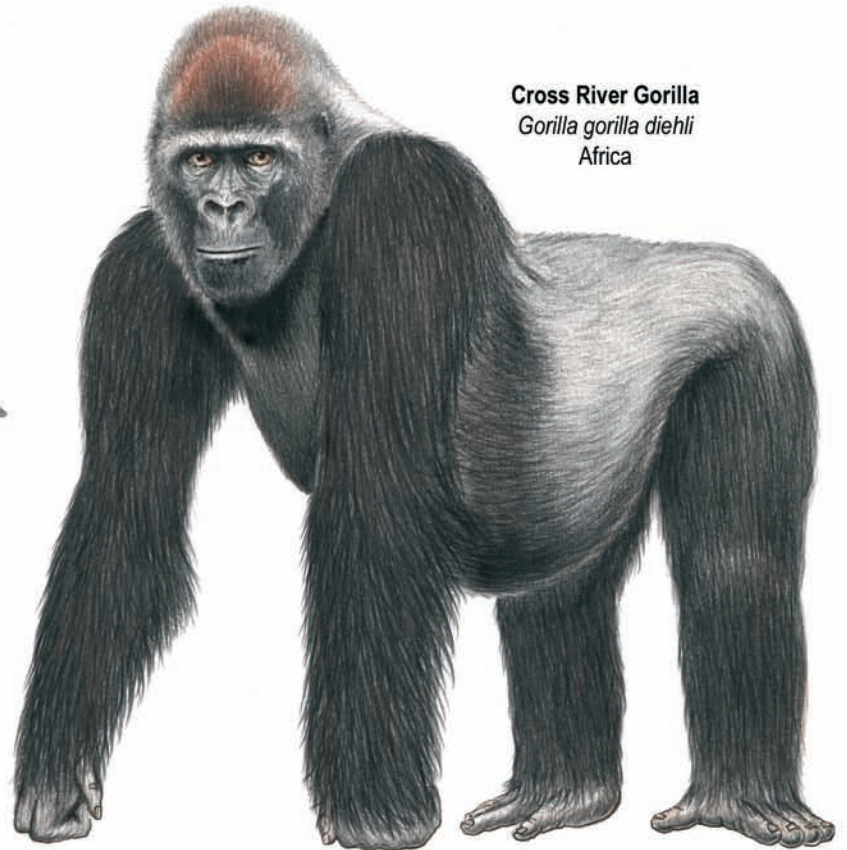
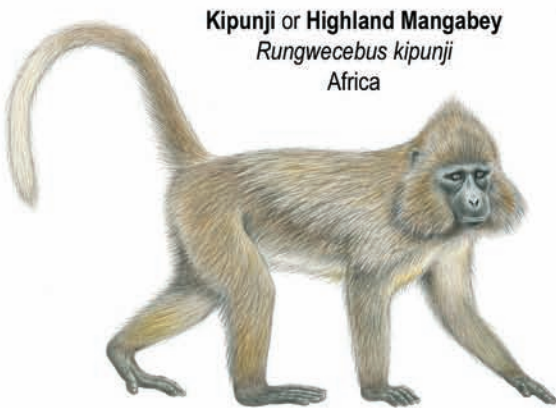
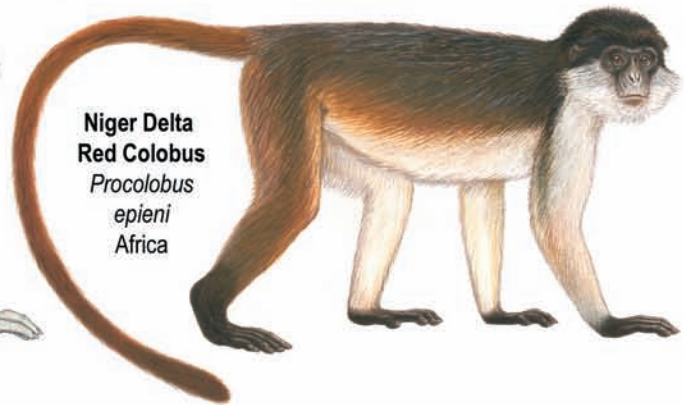
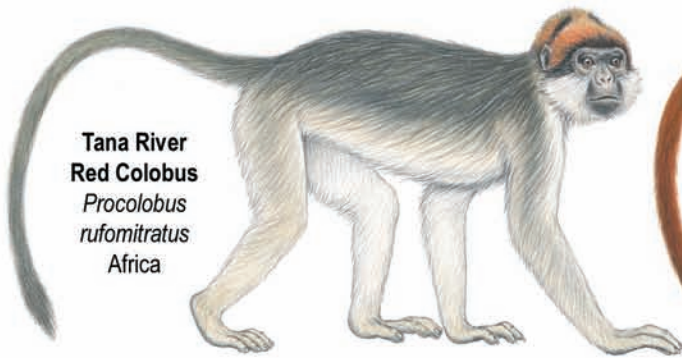


Roloway Monkey
*Cercopithecus diana
roloway*
Africa

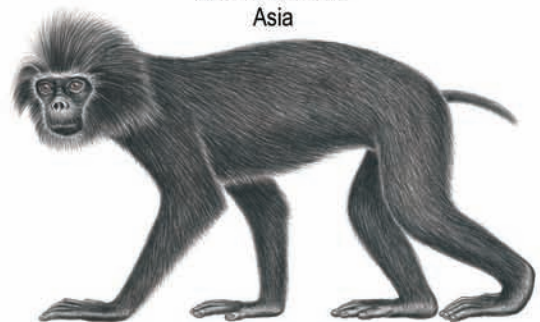


Silky Sifaka
Propithecus candidus
Madagascar





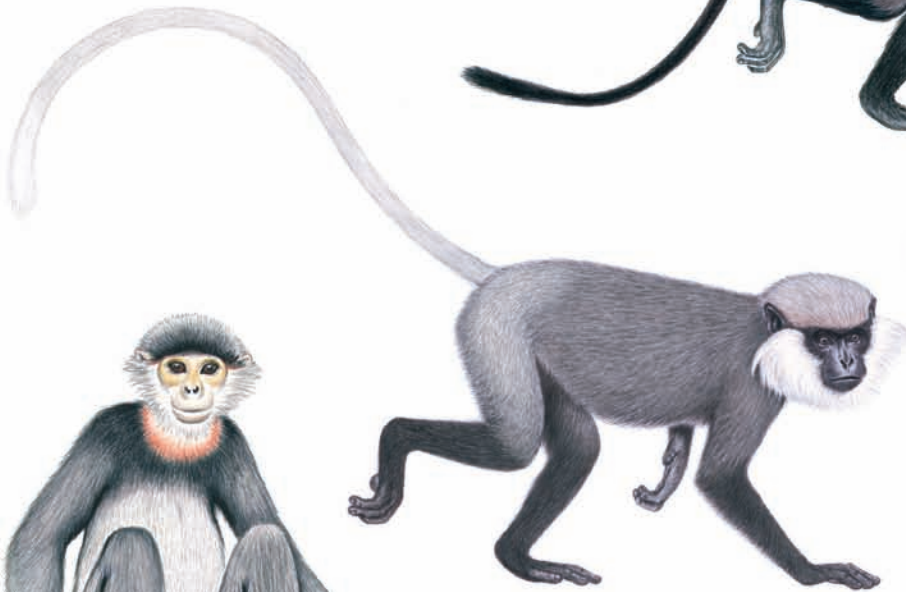
Pagai Pig-tailed Snub-nosed Monkey or Simakobu
Simias concolor
Asia





Delacour's Langur
Trachypithecus delacouri
Asia

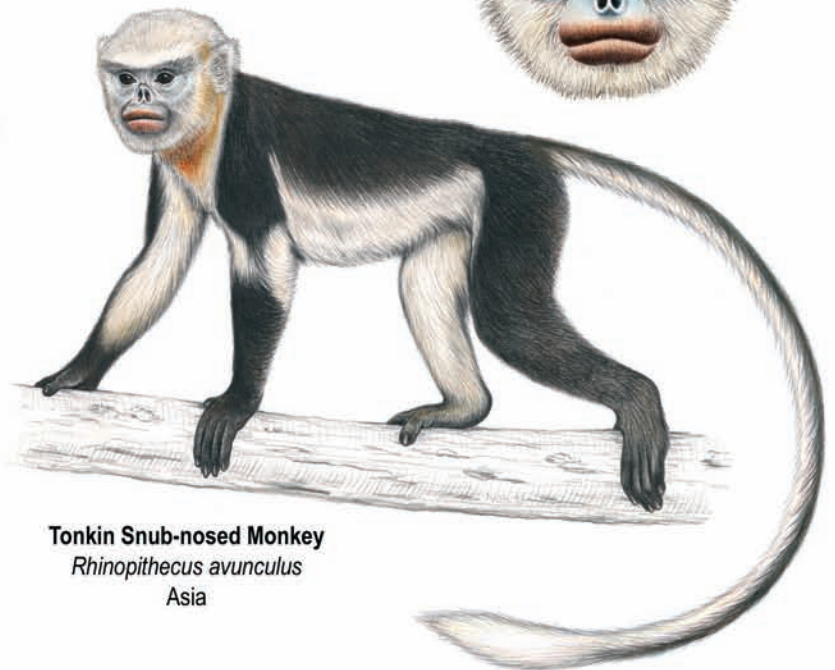
Golden-headed Langur
or
Cat Ba Langur
Trachypithecus poliocephalus poliocephalus
Asia



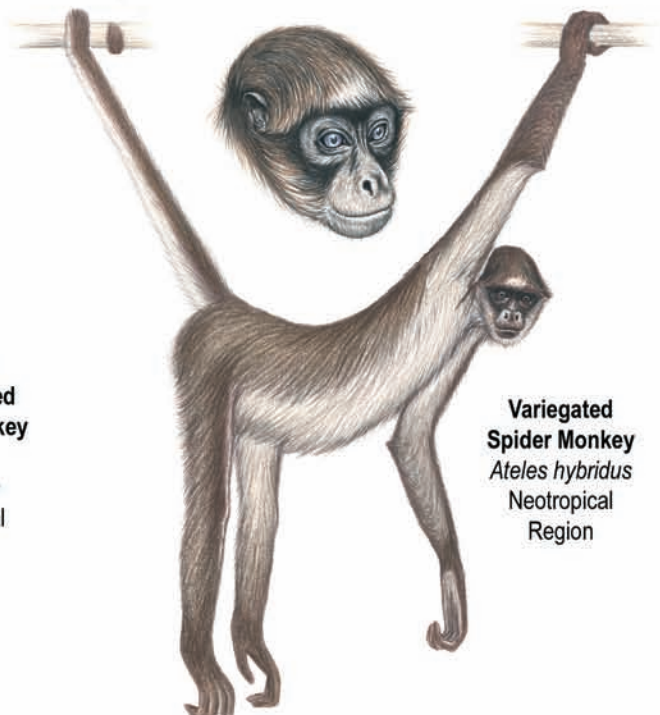
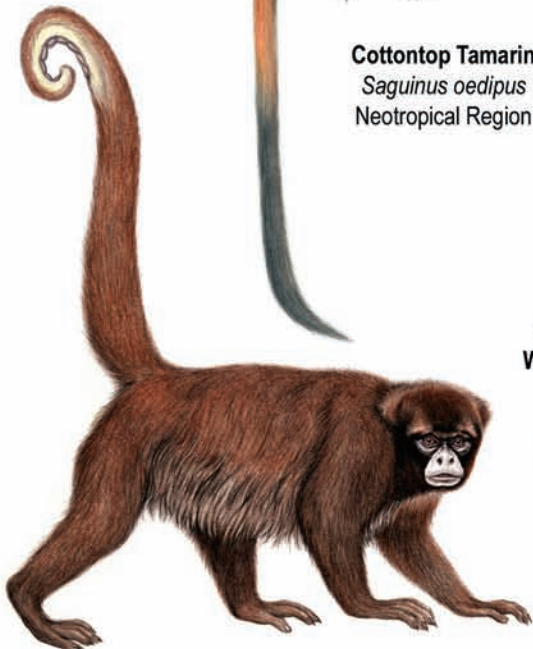
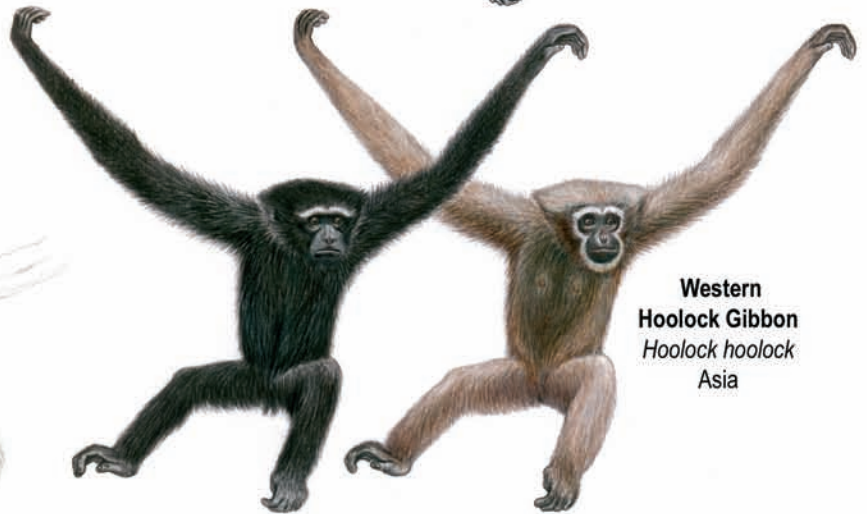
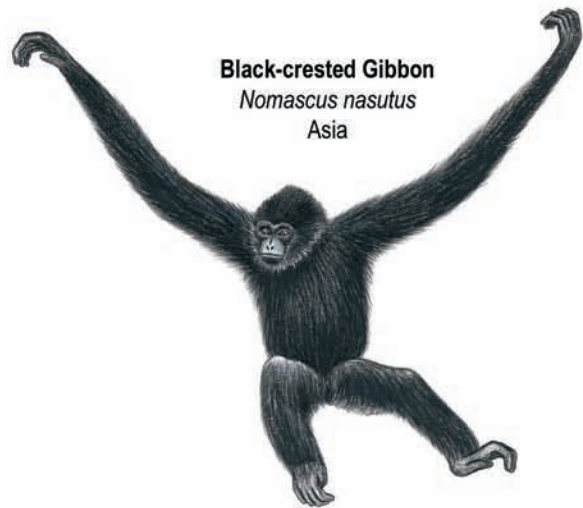
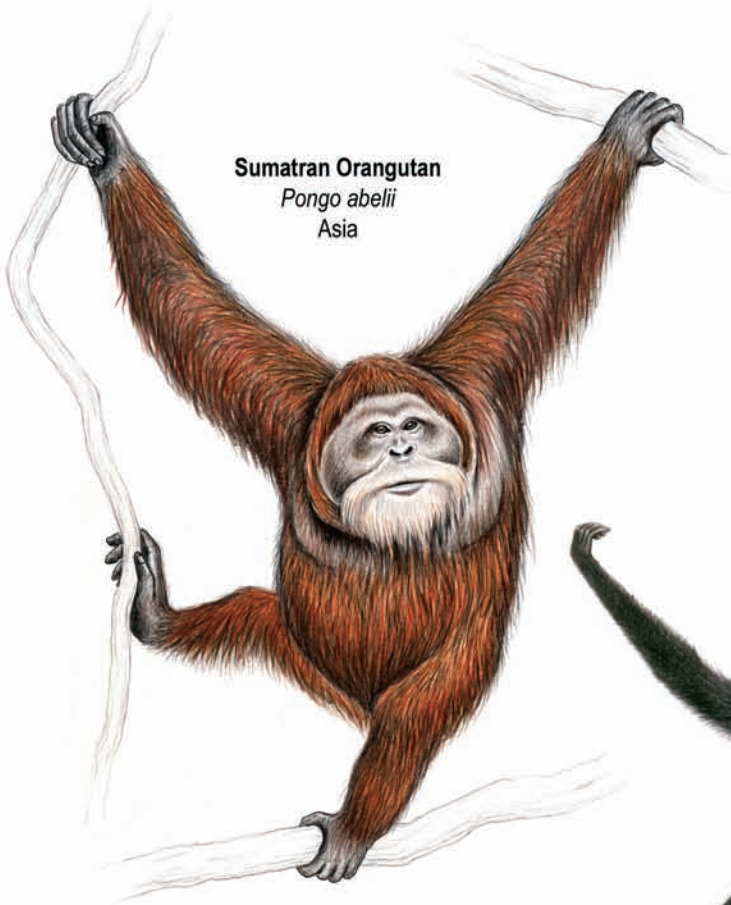
Western Purple-faced Langur
Semnopithecus vetulus nestor
Asia



Gray-shanked Douc
Pygathrix cinerea
Asia



Tonkin Snub-nosed Monkey
Rhinopithecus avunculus
Asia



The Taxonomy and Conservation Status of *Saimiri sciureus albigena*: A Squirrel Monkey Endemic to Colombia

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Abstract: The Colombian squirrel monkey (*Saimiri sciureus albigena*), endemic to central Colombia, is classified as Near Threatened on the 2008 IUCN Red List of Threatened Species. Its geographic distribution is restricted to a small area of the Colombian Llanos, where there are major human impacts, involving the loss, fragmentation and degradation of its forests due particularly to agricultural conversion, and this species' persecution for the pet trade. Here we review its status, and argue that it is threatened and that it be considered as Vulnerable (VU) on future iterations of the IUCN Red List. We suggest the possibility, based on comparative studies of 1140 base pairs of the cytochrome b mitochondrial gene of 38 *Saimiri* specimens of nine different taxa, that the Colombian taxa could be considered subspecies of *Saimiri cassiquiarensis* (not *S. sciureus* as is currently believed), all showing similar haplotypes that are different from *Saimiri sciureus sciureus* of the eastern Amazon. An alternative arrangement could place the three Colombian taxa north of the Río Amazonas—*albigena*, *macrodon*, and *cassiquiarensis*—as full species.

Key Words: Colombian squirrel monkey *Saimiri sciureus albigena*, threatened status, genetics, Colombian Llanos, Colombia.

Introduction

Three squirrel monkeys occur in Colombia, all considered subspecies of the common squirrel monkey, *Saimiri sciureus* (see Groves 2001; Defler 2004). According to Hernández-Camacho and Cooper (1976) and Defler (2004), the Ecuadorian squirrel monkey, *S. s. macrodon* Elliot, 1907, occurs in the basins of the ríos Putumayo and Caquetá, south of the Río Apaporis (Fig. 1). Humboldt's squirrel monkey, *S. s. cassiquiarensis* (Lesson, 1840), occurs to the north of the Apaporis, populating the basins of the ríos Vaupés, Guainia and Inirida, northward as far as the southern (right) margin of the Río Vichada and west to the Andes, between the upper Apaporis and upper Río Guayabero. The Colombian squirrel monkey, *S. s. albigena* Pusch, 1942, has the northernmost distribution in Colombia, in the Llanos Orientales, eastern Andean piedmont and upper Río Magdalena (Defler 2004). The highest altitude recorded for the occurrence of this squirrel monkey is 1,500 m in Huila (Hernández-Camacho and Cooper 1976). The eastern limits in Arauca and Casanare are poorly defined (Defler 2004) (Fig. 1), but influenced by increasing grasslands, decreasing rainfall, and reduced extent of gallery forest as the limiting factors. *Saimiri s. albigena* occupies gallery forest with low canopy and sclerophyllous and hillside forest, palm

forest (associations of *Mauritia flexuosa*) and, extending to the south, the species occurs in the seasonally flooded and *terra firma* rain forests of the Amazon basin (Hernández-Camacho and Cooper 1976). *Saimiri s. cassiquiarensis* and *S. s. macrodon* were categorized as of Least Concern (LC) on the 2008 IUCN Red list of Threatened Species but *S. s. albigena* was considered to be Near Threatened (NT) because a sizeable part of its range has been heavily deforested, and population loss was estimated to be at least 20% in the past 25 years (based on a generation length of 8 years) due to continuing habitat loss and exploitation for pets (IUCN 2008). The IUCN Red List appraisal concluded that it almost qualified as Vulnerable (VU) under the criterion A2cd, that is, “an observed, estimated, inferred or suspected population size reduction of $\geq 30\%$ over the last 10 years or three generations, whichever is the longer, where the reduction or its causes may not have ceased or may not be understood or may not be reversible (A2), based on a decline in area of occupancy, extent of occurrence and/or quality of habitat (c) and actual or potential levels of exploitation (d)” (IUCN 2001). Here we review the status of *S. s. albigena* and its categorization on the IUCN Red List, specifically examining the latter proposition that it might in fact be better regarded as Vulnerable (VU) rather than Near Threatened. We also discuss the results of a molecular genetic study

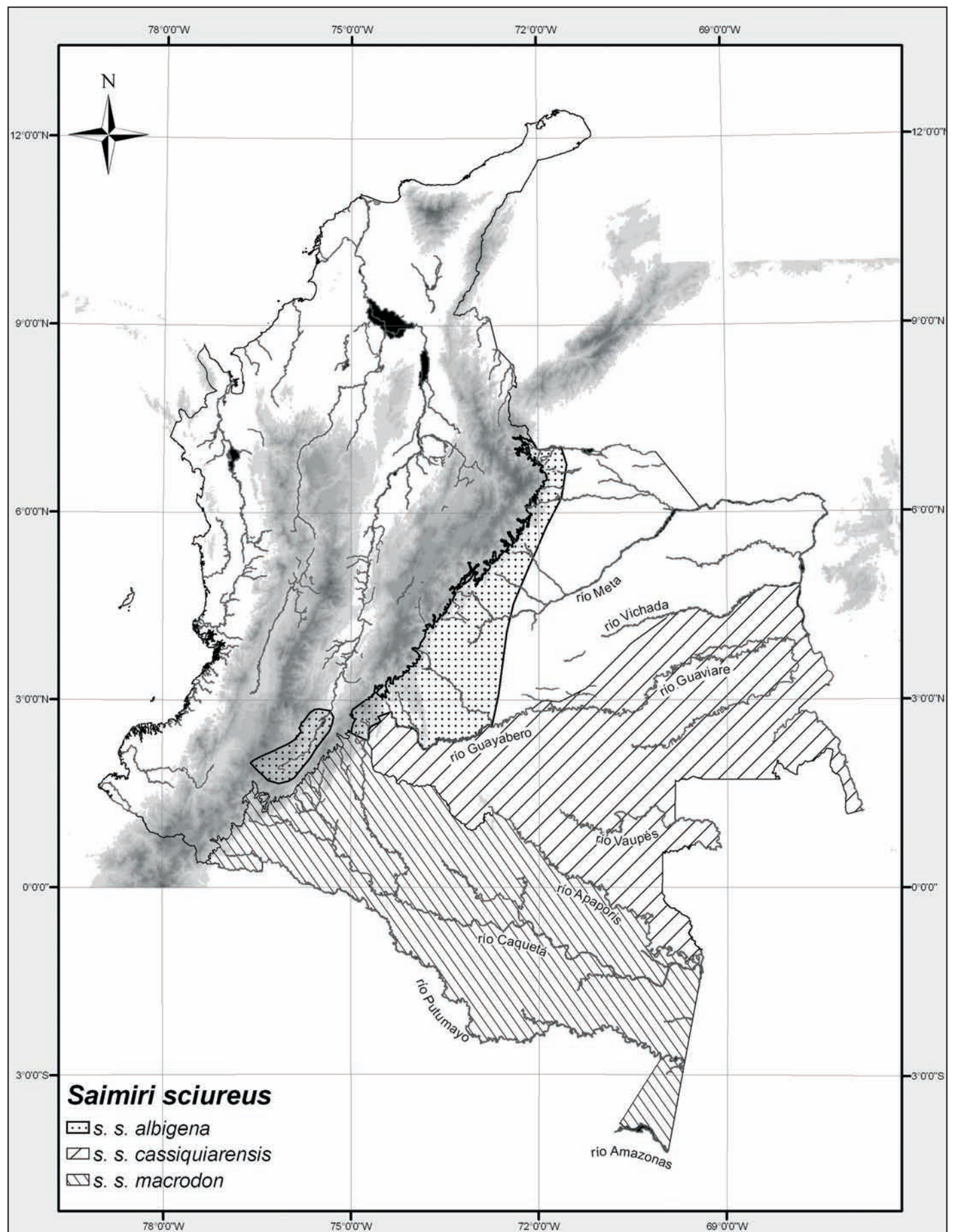


Figure 1. Confirmed distributions of *Saimiri* in Colombia (Hernández-Camacho and Cooper 1976; map from Defler 2004).

(mitochondrial cytochrome b gene) of nine different taxa of *Saimiri* and discuss their implications for the taxonomy of the three currently recognized Colombian forms.

Fragmentation and Habitat Loss

The area of the geographic range that has been confirmed for this endemic subspecies of *Saimiri* is 100,000 km². It comprises a mixture of pasture and gallery forest fragments of different sizes and differing extents of isolation, along with some continuous forest areas in the Department of Meta (Fig. 1). Taking into account the actual habitat available in the subspecies' range, there are about 60,000 km² of gallery forest fragments and continuous forest remaining: about half of the original distribution. The principal threats to this habitat are colonization, African oil palm plantations, pasture for livestock, and deforestation for other sundry reasons. During the last three years increasing governmental incentives for oil palm plantations have resulted in the loss of about 400 ha in a small area around San Martín de los Llanos, in Meta, and deforestation due to oil palm plantation still continues. Deforestation is occurring throughout the range of *Saimiri sciureus albigena*; the area that has been converted to oil palm plantations in the eastern part of the country (mainly in Meta Department) congruent to the distribution of *S. s. albigena* was 121,464 ha in 2008 with a suggested potential there of 1,933,382 ha of converted forest and savanna, all resulting in the destruction of nearby gallery forests (Colombia, Ministerio de Agricultura de Desarrollo Rural, Gobernación del

Meta 2007). This increase in forest conversion to oil palm has compromised fragments of gallery forest that undoubtedly harbored *S. s. albigena* groups and will continue to do so at an increased rate.

The use of fences as part of their home ranges and as corridors between forest patches is one of the reasons why these squirrel monkeys (which typically have large home ranges) have been able to persist in fragmented areas (Carretero-Pinzón *et al.* in prep.). In larger forests, squirrel monkeys can have home ranges of 240 ha or more (Terborgh 1983; Mitchell 1990; Carretero-Pinzón 2000), while in fragments *S. s. albigena* usually have home ranges of around 100 ha (Carretero-Pinzón 2008). In fragments assessed since 2004, *Saimiri* groups were found only in small and large fragments connected by fence rows, but they were absent from even the large fragments that were completely isolated (Fig. 2). Since we began our field research in 2004, one of the *S. s. albigena* groups we were monitoring disappeared from one of the fragments for no apparent reason in 2005 (Carretero-Pinzón pers. obs.). Their absence in larger fragments is evidence of localized extinctions due to deforestation and isolation. Data from environmental impact assessments carried out in the departments of Arauca, Meta, Boyacá and Casanare during 2008 have shown that for seven sites surveyed only one in Meta and one in Boyacá provided evidence that squirrel monkeys were still present. The impact assessments were made on behalf of oil companies, which mean that the forest of these two sites will probably eventually disappear (J. L. Barrera pers. comm.).

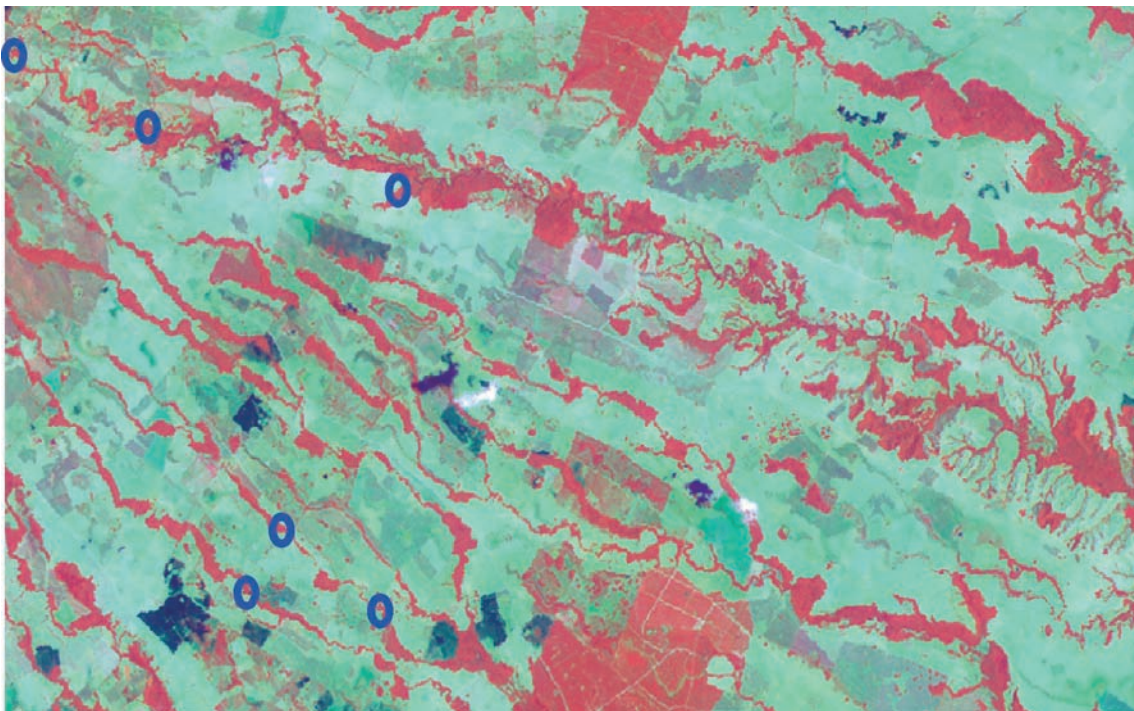


Figure 2. Six groups of *Saimiri sciureus albigena* (blue circles) located in 4,000 ha near San Martín, Department of Meta, Colombian Llanos (Red are forest fragments and oil palm plantations, brown and black parts are burnt areas, and the green marks pastures and open areas. (LANDSAT 0758 ETM EarthSat, Image, 2001).

Illegal Trade

Very large numbers of squirrel monkey were exported to the US and Europe during the 1960s and 1970's, largely from Iquitos (Peruvian Amazon), Letícia (Colombian Amazon), and Barranquilla on the Caribbean coast of northern Colombia (Cooper 1968; Cooper and Hernández-Camacho 1975; Mack and Eudey 1984). Although exports stopped in 1974, the illegal national trade continues, and *S. s. albigena* is one of the most common taxa found in rescue and rehabilitation centers (pers. obs.). They are common as pets in rural areas where often people kill many individuals of the group just to catch one.

Census Numbers

Data for a small portion of their distributional range show that in an area of 4,000 ha there are just 500 ha of gallery forest fragments sheltering only 70 individuals in two subpopulations of 30 and 40. The subpopulations were isolated from each other (three groups in each). We have observed behavioral differences in these isolated populations compared to populations in continuous forest. It is likely that the subpopulation of 30 individuals will go extinct if their particular fragments are not connected in the near future; no group size increase has been observed (from mid-2005 and January 2007) and we do not know the mortality rates in these groups (Carretero-Pinzón unpubl. data). It is possible that there are healthy large populations in the Tinigua National Natural Park (201,875 ha) and the Serranía de la Macarena Natural National Park (630,000 ha), but this needs to be confirmed. The population in Tinigua was secure until 2002, but then guerilla activity made it impossible to continue monitoring them (X. Carretero pers. obs.) and more recent information is lacking. There is much social unrest in the region where these parks are located, associated with guerilla activity and illegal crops such as coca. Another population is in a private reserve, Las Unamas Natural Reserve at San Martín, Meta department (Enciso 2006) that would appear to be well protected, and has a relatively large area of continuous forest (c. 1,300 ha). The first author began an assessment of this population in late 2008. These private reserves may be only safe, however, as long as their present owners live.

The Conservation Status of *Saimiri sciureus albigena*

Based on the information we have about the status of *Saimiri sciureus albigena* we consider that it should be categorized as Vulnerable (VU) following the IUCN Red List categories and criteria (IUCN 2001). The criteria for this are A2 (a, b) and B1 (a, b (ii, iii and iv)) as follows: 'A' is a reduction in population size based on (2) an observed and estimated population size reduction of 30% or more in the last 10 years that may not be reversible, based on (a) direct observation (during five years of work in the zone with this species by the first author) and (b) a diminishment of an index of abundance. The

evaluation is also based on (B) (geographic range), since we estimate the extent of occurrence to be less than 20,000 km² (1) since much of the range above the Río Guaviare consists only of forest fragments and both the findings that the range is severely fragmented (a) into remnant gallery forests that have not as yet been converted to agricultural use, and a directly observed continuing decline (b), in the area of occupancy (ii), area, extent and quality of habitat (iii) and in the number of locations or subpopulations (iv). The principal pressures driving the negative changes in the populations of this taxon are mainly agricultural activity and, especially, the rapid conversion of the land to oil palm plantations, and there seems to be high attrition from animals trying to disperse across roads and over open ground, judging from the frequent dead animals we have observed.

The Taxonomy of *Saimiri sciureus*: Molecular Genetics Data

In a series of recent molecular population genetic and phylogeographic studies, Lavergne *et al.* (in press) and M. Ruiz-García (unpubl. data) analyzed 1,140 base pairs of the cytochrome b mitochondrial gene of 38 *Saimiri* specimens representing nine taxa: *S. sciureus sciureus* (Linnaeus, 1758), *S. s. collinsi* Osgood, 1916 (from Marajó Island; recognized by Cruz Lima [1945], Cabrera [1957] and Hill [1960], but considered a synonym of *S. s. sciureus* by Hershkovitz [1984] and Groves [2001, 2005]), *S. s. macrodon*, *S. s. albigena*, *S. s. cassiquiarensis*, *S. ustus* (I. Geoffroy, 1844), *S. oerstedii* (Reinhardt, 1872), *S. boliviensis boliviensis* (d'Orbigny, 1834) and *S. b. peruviansis* Hershkovitz, 1984 (Fig. 3). *S. ustus* was quite clearly separated, confirming its classification as a distinct species by Elliot (1913), Cruz Lima (1945), Hill (1960), Hershkovitz (1984) and Groves' (2001, 2005). The results showed that *S. s. albigena* (two specimens from the Meta Department in Colombian Llanos) had two haplotypes for the cytochrome b gene that were not shared with other *Saimiri* taxa. These *albigena* haplotypes showed the lowest mean genetic divergence with *S. s. cassiquiarensis* (of 0.53, SD = 0.20), but no haplotypes were shared between the two neighboring squirrel monkeys *S. s. cassiquiarensis* and *S. s. macrodon*. A median joining haplotype network (Fig. 3) clearly showed that *albigena* represents an independent branch related to a group composed otherwise of *S. s. cassiquiarensis* and *S. s. macrodon* (and, more distantly, with *S. ustus*). The genetic differentiation in this *Saimiri* group from *S. sciureus sciureus* of French Guiana was conspicuously higher: *S. s. sciureus* differs by 45 nucleotide substitutions (ns) from *S. ustus* and 48 ns for *S. s. albigena*. In contrast, the number of substitutions within the above mentioned group, including *S. ustus*, was conspicuously lower. *S. ustus* presented 26 ns compared to *S. s. albigena*.

The Median Joining network (Fig. 3) was clearly useful for estimating divergence times between the haplotypes identified. The main *S. s. sciureus* haplotype diverged from the *S. s. albigena* haplotypes from 1 to 2.3 million years ago

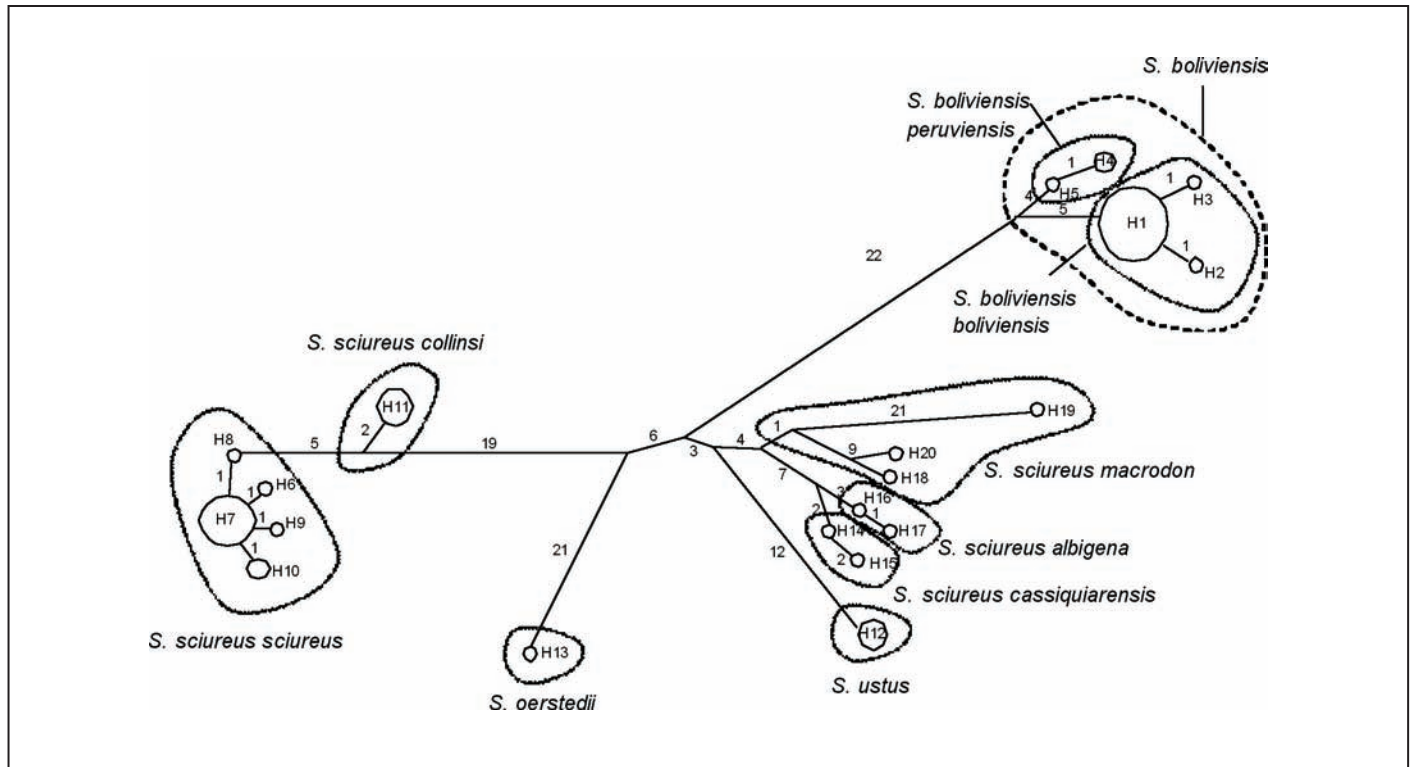


Figure 3. Median joining haplotype network for *Saimiri* taxa based on 1140 base pairs of the mitochondrial cytochrome b gene.

(mya), and *S. s. albigena* diverged from *S. oerstedii*, 2.7 to 6 mya. *Saimiri oerstedii* is closer, genetically, to *S. s. sciureus*, than is *S. s. albigena* (Fig. 3). The divergence between *S. s. macrodon* and *S. s. albigena* dates back to 1.1–1.9 mya, and the split of *S. s. albigena* from *S. s. cassiquiarensis* was more recent (0.28–0.63 mya), during the Quaternary (Pleistocene and Holocene, 1.6 mya to recent). The split between *S. sciureus sciureus* and *S. s. albigena* was in the beginning of the Pleistocene or in the last phase of the Pliocene during a period of heavy glacial advances and retreats with consequent dry periods, while that between *S. s. albigena* and *S. s. cassiquiarensis* could have corresponded to some period of the second and third Pleistocene glaciations. Pleistocene forest fragmentation could be responsible for the separation of the different haplotypes in *S. s. albigena* (Haffer 1997; Whitmore and Prance 1987). The separation of *albigena* from *cassiquiarensis* could be related to the presence of certain rivers or forest refuges related to cyclical climatic changes. For example, the Río Apaporis separates the distribution of *macrodon* and *cassiquiarensis*. Likewise, it is probable that *albigena* and *cassiquiarensis* were separated by expansion of the Eastern Llanos.

If we assume that *S. ustus*, *S. oerstedii* and *S. boliviensis* are distinct species, different from *S. sciureus* (see Hershkovitz 1984), and strictly follow the results obtained with the cytochrome b mitochondrial gene, then *S. s. sciureus* and *S. s. collinsi* of the eastern Amazon could be grouped. If *S. sciureus macrodon*, *S. s. albigena* and *S. s. cassiquiarensis* of the central and northwestern Amazon (Brazil, Colombia,

Ecuador, Peru and Venezuela) were to be considered a separate group but sufficiently closely related as to be considered subspecies, then *cassiquiarensis* Lesson, 1840 would be the oldest name of the three taxa, and hence the nominotypical species. However, following the Phylogenetic Species Concept (see Groves 2001, 2004), it would seem most appropriate that they be considered distinct but closely related species: *S. macrodon*, *S. albigena*, and *S. cassiquiarensis*. Further research using other molecular markers could confirm or modify the phylogeny of the squirrel monkeys that we have identified here, and would reinforce the need to modify the taxonomic arrangements as proposed by Hershkovitz (1984) and Groves (2001).

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Conservation Priorities for the Peruvian Yellow-Tailed Woolly Monkey (*Oreonax flavicauda*): A GIS Risk Assessment and Gap Analysis

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Abstract: An inductive GIS (Geographical Information System) process was used to create a realistic Habitat Suitability Model (HSM) for the current distributional range of the Peruvian yellow-tailed woolly monkey (*Oreonax flavicauda*) to aid current conservation initiatives and to help set future conservation priorities for the species. In combination with this, we produced an ecological risk assessment model of the study region to assist in site selection for priority areas for conservation actions, which included the expansion of the existing protected area system and the creation of new reserves in areas forming natural biological corridors in the northeastern Peruvian departments of Amazonas and San Martín. This study incorporates information regarding the threat of hunting and other anthropogenic pressures on the species into the site selection process. *Oreonax flavicauda*, currently on the *IUCN Red List of Threatened Species* as Critically Endangered, was once thought to be extinct. Since its rediscovery in 1974, however, there has been little research on this species due to its small population size, restricted distribution, and the difficulty of access to its mountainous habitat. A gap analysis showed that the current protected area network was inadequate to conserve the yellow-tailed woolly monkey's current suitable habitat. This finding underlines the urgency of upgrading the protected area network as well as implementing other conservation initiatives. The selection of sites suitable for the creation of new protected areas was based on habitat composition, altitudinal and geographical limits, and proximity to human influences, using an inductive process of extracting information from locations where *O. flavicauda* is known to occur, from existing demographic information on human populations, and by qualitative judgments. We recommend urgent action to protect this species before population numbers decrease further.

Key Words: Yellow-tailed woolly monkey, *Oreonax flavicauda*, deforestation, arcGIS, cloud forest, tropical Andes

Introduction

The yellow-tailed woolly monkey (*Oreonax flavicauda*) is endemic to the Peruvian Andes, and the largest of Peru's primates (Leo Luna 1987). It is also one of the most threatened; listed as Endangered on Appendix I of CITES (2005) and as Critically Endangered on the *IUCN Red List of Threatened Species* (Cornejo *et al.* 2008). It has also been on the list of the World's 25 Most Endangered Primates since 2006 (DeLuycker and Heymann 2007; Cornejo *et al.* 2009). Even so, comparatively little investigation or conservation work has been carried out on this species and very little is known about its status. There are no accurate population estimates, but Nowak (1999) wrote that there were only 250 individuals surviving in the wild. Although this is probably an underestimate, the current population will not be much larger than this,

and is surely decreasing, making more effective conservation measures critical to the species' survival.

Oreonax flavicauda, a flagship species for the Tropical Andes Biodiversity Hotspot (Myers *et al.* 2000) has a very limited geographical range (Leo Luna 1987; Shanee *et al.* 2007, 2008). It can be found only in a small area of primary montane and cloud forest in the Peruvian departments of Amazonas and San Martín (Butchart *et al.* 1995) between the altitudes of 1,500 m and 2,700 m above sea level (Leo Luna 1982; DeLuycker 2007). Early locality records have also shown the occurrence of this species in small areas of the neighboring departments of Huanuco, Loreto, La Libertad and Cajamarca (Mittermeier *et al.* 1975; Graves and O'Neil 1980; Leo Luna 1980, 1982, 1989; Parker and Barkley 1981; DeLuycker 2007).

The threats that determined the status of *O. flavicauda* as Critically Endangered include hunting and deforestation.

Despite being prohibited under Peruvian law, subsistence and trophy hunting still occur throughout the species' range. Hunting is made easier by its conspicuous nature and large size. At least 20 monkeys were hunted in the areas surrounding the Bosque de Protección de Alto Mayo and the Zona Reservada Cordillera de Colán over 18 months in 2007–2008 (Shanee *et al.* in prep.). The rate of deforestation in Amazonas and San Martín is among the highest in Peru (Peru, INEI 2008), fuelled by the need for agricultural land, coffee cultivation and small- and large-scale timber extraction (DeLuycker 2007; Shaneec *et al.* 2007; EDGE 2008). The widespread deforestation that has occurred in this region has, in many areas, forced *O. flavicauda* into isolated forest fragments (Shanee *et al.* 2007). Although the area currently occupied by *O. flavicauda* is unknown, in 1981, Leo Luna (1982) estimated its potential forested habitat to be at least 11,240 km², but predicted that this would decrease by 1,600 km² by 1991. Based on these figures and projected rates of deforestation, DeLuycker and Heymann (2007) estimated that by 2006 potential forested habitat would have been reduced to 7,240 km². These (conservative) estimates represent a loss of approximately 35% of the total potential habitat for this species in just over two decades.

Until the 1950's this species was well protected due to the inaccessibility of its habitat, characterized by high mountain ridges and steep river valleys. Since then, however, new roads have been built throughout its range. The roads have brought with them immigrants from the Peruvian coast and the high mountain sierras, and have resulted in increased deforestation and the fragmentation of much of its habitat. Woolly monkeys have very low reproductive rates, with long interbirth intervals, and population densities are always low. These factors contribute to its vulnerability to anthropogenic pressures (Leo Luna 1987).

Here we create the first realistic Habitat Suitability Model (HSM) for the yellow-tailed woolly monkey, using inductive GIS modeling methodology to predict its current geographic distribution in Amazonas and San Martín. We also carried out a GAP analysis of the current protected area (PA) network in the region to assess the extent to which its habitat is protected, to determine zones of potential threat, and to predict which areas would be optimum for the creation of new protected areas and corridors to bridge priority areas.

Methods

There is limited information on the habitat preferences and current distribution of *O. flavicauda*. It is known to occur throughout the departments of Amazonas and San Martín on the eastern slopes of the Andean Cordillera in northeastern Peru between 3° and 9° south and 75° and 79° west. These departments border Ecuador to the north, and the departments of Loreto, Huanuco, La Libertad and Cajamarca (to the east, south, southwest and west, respectively). The topography of the departments of Amazonas and San Martín range from high mountain sierras to lowland rainforest. Combined these departments have a human population of close to a million

(Peru, INEI 2008). There are a number of protected areas, including the Zona Reservada Cordillera de Colán in Amazonas and the Parque Nacional Río Abiseo and Bosque de Protección de Alto Mayo in San Martín. Yellow-tailed woolly monkeys inhabit forests at altitudes of 1,500–2,700 m above sea level. Their range is limited in the north by the Río Marañón valley and to the northwest by the Río Utcubamba valley.

Data collection

Data collected for this study included field observations, ecological niche data, land use maps, and digital maps including the Digital Elevation Model (DEM) of Peru (<<http://www.srtm.usgs.gov>>). We searched the literature, printed and online material, to include all sightings and distribution data for the species. We included sightings resulting from our ongoing surveys in Peru (S. Shaneec pers. obs.; Hans Dignum pers. comm.), and where possible we contacted people involved in previous research in order to record any additional unpublished localities. We checked online museum collections but the only extra record we found was discounted because the location given was too vague.

We used ArcView 9.2 (ESRI 2008) for analysis and modeling. Land use maps were obtained from the Instituto de Investigaciones de la Amazonía Peruana (IIAP) (Peru, IIAP 2007, 2008) and additional reserve data from Amazonicos Para la Amazonia (Miraliz Egoavil pers. comm.). To make data sets more manageable, land-use files were reclassified to nine land cover classes, as follows: Palm Forest, Scrub, Marsh, Deforested areas, Forest, Fig (*Ficus* spp.) dominated forest, Population Centers, Mixed Areas, and waterbodies). The forest class was then subdivided into a further six classes as follows: Mixed communities, Andean forests in high mountains with medium trees associated with thickets, Andean forests with medium and large trees in high mountains, Sub-Andean oriental, Lower montane forest with medium trees, and Sub-Andean of steep mountains with large trees. DEM90 from the Shuttle Radar Topography Mission in raster format was also reclassified to a set of 12 altitudinal classes from 0 to >3,000 m above sea level.

Distribution and Habitat Suitability Modeling (HSM)

Locality data collected were converted into decimal degrees and assigned the WGS84 coordinate system. Using similar methods to Ramp *et al.* (2005), a kernel density transformation was applied to *O. flavicauda* point locality data. In some cases several localities were aggregated to a single point to avoid over representation due to more detailed field work at any given point. A total of 43 localities were used to create 27 points (Fig. 1). This was used to determine species "hotspots", where there were the highest densities of *O. flavicauda* sightings.

The final HSM was created using three methods; Environmental Envelope Modeling, Inductive Decision-tree Methods, and Habitat Suitability Preference Indices. Habitat preference indices were created for each of the separate land cover, forest, and altitudinal classes found within the predicted distribution

of *O. flavicauda* following a similar method used by Aspinall (1993). Each of the variables in these three were weighted and allocated a Marginal, Good or Very Good habitat class. Once suitability classes had been assigned to the seven land cover classes and 12 altitudinal categories, Boolean maps were created for each of the different classes (19 in total).

Ideally, habitat models require field validation, but in this case validation techniques were limited to statistical sign and literature validation. Chi-square tests were used to test forest and altitudinal preferences shown by actual localities against equal numbers of random points. A final test was made by overlaying the HSM against known deforested and urban areas.

GAP analysis and ecological risk assessment

ArcGIS 9.2 (ESRI 2008) was used to produce a Gap analysis of the current protected area network within the study region to examine how much of the species habitat is

currently protected and to show gaps in the existing network. Datasets used to complete the gap analysis included the following: weighted HSM, protected area network data layer, urban area data layer, road network dataset, and populated areas data layer. Overlaying the protected area (PA) network dataset (Peru, IIAP 2007, 2008) on top of the weighted HSM highlighted areas that needed attention. Approximate area values were calculated by multiplying the area of each raster cell by the total number of cells for each suitability category (i.e., Marginal, Good, Very Good). This was used to find out the area of each suitability class that fell within each of the existing PAs to estimate efficiency at protecting the species current habitat based upon the weighted HSM.

A risk analysis was carried out to assess areas of *O. flavicauda* habitat facing the highest threats due to proximity to human development (Peyton *et al.* 1998). Urban and populated areas and road network data were used to assess threat levels. Each area outside a PA was classified to one of four threat levels based on proximity away from human development (>8 km No Risk, >4 km and <8 km Low Risk, >0.5 km and <4 km Medium Risk, <0.5 km High Risk). The new risk layers, together with the weighted HSM layer, were then converted to vector layers and given unique fields depending on degree of threat. Non-suitable habitat was then removed from the layer and a new layer created to show only areas of suitable habitat at all levels. This was then used to create a layer showing only areas of No or Low Risk, the PA network was then overlaid on top of the risk assessment layer to highlight gaps in the network and priority areas of high suitability for the creation of new reserves and corridors. These areas were those that showed No or Low Risk combined with Good or Very Good habitat.

Results

Habitat Suitability Modeling (HSM)

Habitat preference from suitability indices calculated based on field observations and literature searches determined that forested (excluding palm dominated forest) areas from 1,600 to 1,800 m above sea level, and 2,200 to 2,400 m above sea level constituted “Good habitat”, and forested areas between 1,800 and 2,200 m above sea level constituted “Very Good habitat”. Based on this model there is a total of 6,302 km² of habitat available to *O. flavicauda*, of this only 2,024 km² is rated “Very Good”. The majority of remaining habitat is found in the northern area of the species’ range and along the southwestern border of the Department of San Martín.

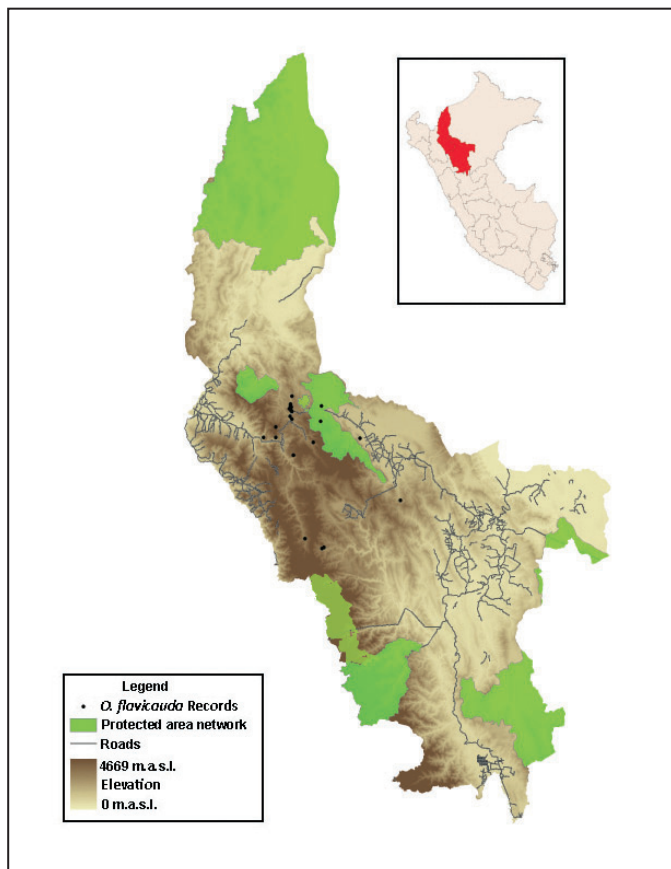


Figure 1. *Oreonax flavicauda* locality records used in this study.

Table 1. Suitable habitat within the protected area (PA) network and outside the PA network.

Habitat suitability	Total Area (km ²)	Within PA network (km ²)	Within PA network (%)	Outside PA network (km ²)	Outside PA network (%)
Good	4278	1128	26	3150	74
Very Good	2024	739	37	1285	63
Total	6302	1867	30	4435	70

Gap analysis

Only 30% of remaining suitable habitat in Amazonas and San Martín is found within the PA network, leaving 70% unprotected. Thirty-seven percent of remaining habitat rated as “Very Good” is found within the PA network, leaving 63% unprotected (Table 1).

Gap analysis of existing protected areas within the species’ range showed the largest areas of unprotected habitat to be along the southwestern border of San Martín and in the northern area of the species’ range, between the Zona Reservada Cordillera de Colán and the Bosque de Protección de Alto Mayo (Fig. 2). It was also found that there was a significant difference in deforestation within protected areas (703.8 km²) compared to deforestation outside those areas (24,276 km²).

Based upon remaining suitable habitat in areas of minimal risk, one new protected area, one wildlife corridor, and four extension zones to existing protected areas are suggested (Fig 3). Together these would protect an extra 2,806 km² of *O. flavicauda* habitat (leaving only 1,620 km² or 26% unprotected).

Discussion

No range-wide studies are available for the distribution of *O. flavicauda*. This is mainly due to the rarity of the species, but also to the fact it was thought extinct until its rediscovery

in 1974 (Mittermeier *et al.* 1975). Two studies (Leo Luna 1980; Shanee *et al.* 2008) have evaluated a number of sites, but neither covered the entire range of the species. Other occurrence records do exist, but these studies are spread over the last three decades and the older studies are less relevant due to the widespread deforestation since the 1970s. Proper analysis of the actual current distribution of this species is needed to help conserve *O. flavicauda* and its habitat.

The two main threats facing tropical wildlife are hunting and land conversion. Hunting still occurs in Amazonas and San Martín, particularly in native communities but there is considerable opportunistic hunting by immigrant communities (Shanee pers. obs.). Until the 1950’s this species was fairly well protected due to the inaccessibility of its habitat. Since the construction of new roads, which began in the 1970’s, immigration has resulted in the human population increasing to around 1 million inhabitants. The proliferation of road building and mining concessions in recent years is increasing deforestation, and further contributes to the risks this species faces. This widespread deforestation throughout the species’ range has caused fragmentation of the forest, which forces species into smaller areas where competition for resources is more pronounced, access for hunters is facilitated, and connectivity between individual populations reduced, increasing the risk of genetic degeneration from inbreeding.

This study was limited by the paucity of current data from departments other than Amazonas and San Martín but,

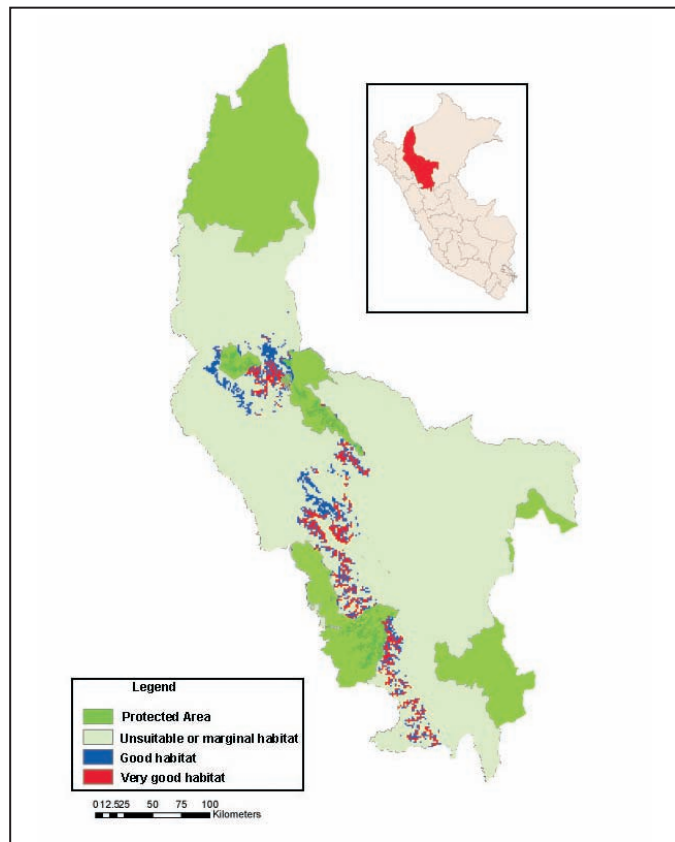


Figure 2. Protected areas network showing gaps in habitat protection for *Oreonax flavicauda*.

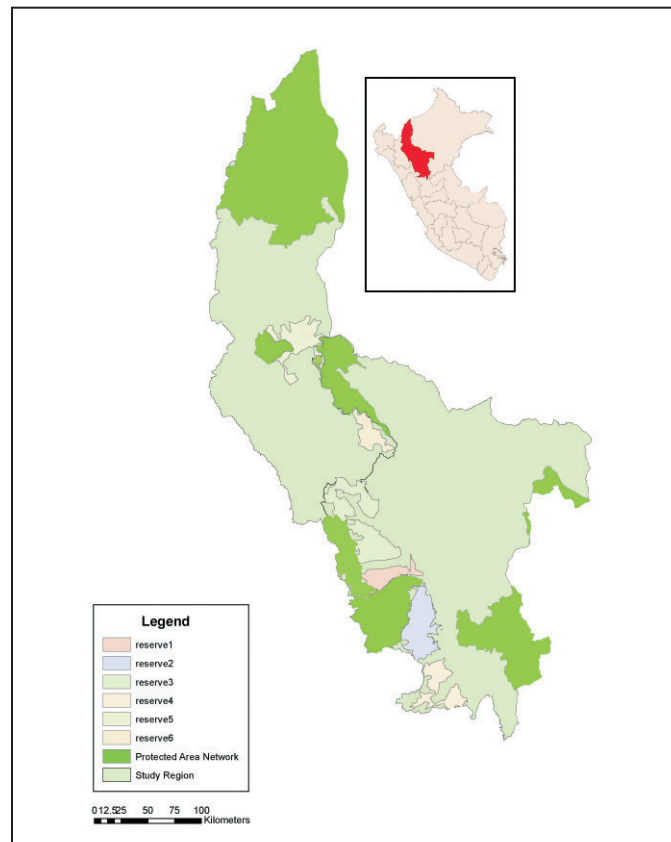


Figure 3. Areas proposed for protected areas for the conservation of *Oreonax flavicauda*.

even so, we feel the results presented here to be valid, as the vast majority of the species' range is within the area covered by our analysis. Gap analysis is a useful tool for wildlife conservation, and has been used effectively for many species (for example, Mariano *et al.* 2006; Smith *et al.* 2003; Tognelli *et al.* 2008) to assess protection levels and park efficiency. This study shows that areas currently protected are fairly efficient in protecting the forest within their boundaries. However, it is clear that deforestation does still occur within them, if at a slower rate than outside their boundaries. This is confirmed by other sources, for example it is estimated that as much as 4,528 ha of forest had been destroyed in the Bosque de Protección de Alto Mayo by the year 2000 (Peru, INRENA 2008), and that between 4,399 (Peru, INRENA 2008) and approximately 5,000 families currently live there (ParksWatch, Peru 2007). This has been attributed to the fact that until recently there were only three park guards employed to protect the 182,000 ha of the park. It is hoped that with the release of the new management plan (Peru, INRENA 2008) the situation will improve. Approximately 70% of *O. flavicauda*'s potential habitat is not currently protected, showing that there are opportunities for expanding the existing PA Network to protect the species. Recommendations concerning this have been published in previous studies (Butchart *et al.* 1995; DeLuycker 2007; Shanee *et al.* 2007).

It was estimated that by 2005 potential forested habitat left in the region would be reduced to 7,240 km² (DeLuycker and Heymann 2007). This study estimated that actual forest loss is higher than expected, and that only 6,302 km² of habitat are still available. For over three decades urgent recommendations have been made for the establishment of new protected areas for this species. This has resulted in the creation of three government reserves, Parque Nacional Río Abiseo (274,500 ha), Bosque de Protección de Alto Mayo (182,000 ha) and the Zona Reservada Cordillera de Colán (64,115 ha), two conservation concessions, Alto Huyabamba (143,928 ha) and Abra Patricia-Alto Nieva (10,000 ha), as well as the several small private conservation areas. We believe that the current PA network in the area is still insufficient because immigration rates in the region are still among the highest in Peru. There are also plans for new private and public reserves, including; a bi-regional conservation area (approximately 300,000 ha) as well as the Huicungo Municipal Conservation Area (92,827 ha) and Breo Conservation Concession (113,722 ha) in the south of San Martín.

The National Protected Area System (*Sistema Nacional de Areas Nacionales Protegidas*) of Peru includes a number of protected area categories, each affording a different level of protection. In the case of *O. flavicauda*, the Parque Nacional Río Abiseo affords the most protection as it is of indirect use only (strict protection). Bosques de Protección, such as Alto Mayo, are a category of so-called "direct use." Limited exploitation of the forest is permitted as long as it does not affect vegetation cover or water courses. Bosques de Protección are often viewed as the lowest form of protection (S. Shanee pers. obs.). A Zona Reservada, such as that of the Cordillera de

Colán, is a transitional stage, for areas where the government is as yet unable to determine the appropriate protected status. It is probable that the Zona Reservada Cordillera de Colán will become a Santuario Nacional (Cesar Barta pers. comm.), in which only indirect use is permitted. These are usually created as refuges for the conservation of an individual species, such as *O. flavicauda* (Peru, INRENA 2009). Conservation concessions and private conservation areas are of indirect use, but the status and level of protection afforded depends on the individual management plans.

The creation of reserve three (Fig. 3) in the District of Huicungo, San Martín, would protect the largest area of the Good and Very Good habitat that we identified. The Good and Very Good habitat would also be partially covered by the three proposed reserves and concessions mentioned above (covering an additional 1,632 km² of *O. flavicauda* habitat). Likewise, the creation of a protected corridor between the Bosque de Protección de Alto Mayo and the Zona Reservada Cordillera de Colán in the Province of Bongara, Amazonas, would not only protect a large area of Good and Very Good habitat, but would also ensure future genetic flow between populations in the two protected areas. There are currently two NGOs working in this area; Asociación Ecosistemas Andinas (ECOAN) and Neotropical Primate Conservation (NPC). ECOAN have recently been granted a 10,000-ha conservation concession for lands bordering the Bosque de Protección de Alto Mayo and are involved in developing ecotourism. Neotropical Primate Conservation and the Natural History Museum of San Marcos University are working with the communities of Yambrasbamba and Corosha towards the development of a series of private conservation areas (*Area de Conservación Privada*) to close the gap between protected areas. Both NGOs also promote reforestation in the area, and NPC has education and community assistance programs running in conjunction with conservation work.

There is also a reserve being planned in the cross border region of La Laguna de los Condores in the southeast of Amazonas and west of San Martín. These and other conservation actions will not only protect *O. flavicauda* but also many other endemic and threatened species. Further field studies to properly evaluate other areas highlighted by this study and to develop a detailed map of the species range are set to begin in 2009. We conclude by raising the issue of the necessity and urgency of increasing the size and efficiency of the current PA network in Amazonas and San Martín in order to ensure the survival of the yellow-tailed woolly monkey.

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A Comment on the Status of “*Colobus polykomos dollmani*” in Côte d’Ivoire

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Abstract: The report by Sery *et al.* (*Primate Conservation* 21: 55–61, 2006) that *Colobus polykomos dollmani* had been observed in southern Côte d’Ivoire during surveys in 2003–2004 is questioned. Not only is this taxon of the black-and-white colobus monkeys of uncertain validity (being regarded by some authors as a set of hybrids between *Colobus polykomos* and *C. vellerosus*), but Sery *et al.* also do not provide sufficient evidence on the features of the monkeys they observed for these to be reliably assigned to a taxonomic entity. Whether or not *C. p. dollmani* is a valid taxon, future research in the region between the Sassandra and Bandama rivers should be encouraged so as to clarify the affinities of any remaining black-and-white colobus populations and to formulate concrete plans for the conservation of any evolutionarily distinctive populations located.

Key words: Black-and-white colobus, taxonomy, geographic distribution, *Colobus polykomos dollmani*, *Colobus vellerosus*, Côte d’Ivoire

In an important article in *Primate Conservation*, Sery *et al.* (2006) reported on the results of primate surveys they conducted in 2003 and 2004 in three forest reserves (Dassiéko, Bolo and Niégré) and one national park (Marahoué) located between the Sassandra and Bandama rivers in Côte d’Ivoire. The authors listed the potto, seven forest monkey species, the olive baboon, the green monkey and the chimpanzee as each being present in at least one of the four survey sites, based on the verbal reports of local people (particularly hunters), or the presence of bushmeat in nearby villages. The only species which they noted as having seen or heard themselves in the field were: Lowe’s monkey (*Cercopithecus campbelli lowei*), the spot-nosed monkey (*C. petaurista petaurista*), the white-naped mangabey (*Cercocebus atys lunulatus*), the olive baboon (*Papio anubis*), the olive colobus monkey (*Procolobus verus*), and Dollman’s black-and-white colobus monkey (*Colobus polykomos dollmani*).

Sery *et al.* (2006) stated that the main aim of their survey was to gather information about the status of *Colobus polykomos dollmani* between the Sassandra and Bandama. They noted that they observed one group of this monkey in the Bolo Forest Reserve (5°07’–5°26’N, 5°47’–6°03’E). They also noted that local people reported that they had seen this colobus in Dassiéko Forest Reserve in 2003, that it was present

(but rare) in Niégré Forest Reserve, and that in Marahoué National Park it had last been seen in 2002 (see Figure 1 for the location of these sites). Sery *et al.* said that hunting and habitat destruction are the main threats to the survival of this subspecies and other taxa, and argued that, without immediate and vigorous action, *C. p. dollmani* will probably be extinct in the near future.

We share the concerns of Sery *et al.* (2006, 2008) for the survival of the primates and other wildlife in the forests of Côte d’Ivoire, and agree that better protection of remaining forests and wildlife should be a very high priority. We disagree with them, however, in parts of their evaluation of the status of *C. p. dollmani*. Not only is this taxon of somewhat doubtful validity (as Sery *et al.* [2006] themselves acknowledge), but we think that Sery *et al.* may be in error in portraying the range of this population as occupying the entire moist forest zone between the Bandama and Sassandra rivers. In particular, we are not yet convinced about its existence in the vicinity of the Bolo, Dassiéko and Niégré reserves on the eastern side of the lower Sassandra River, in the absence of a careful description of the black-and-white colobus monkeys observed there.

Colobus polykomos dollmani was named in 1927 by Ernst Schwarz based on two specimens collected by Willoughby P.

Lowe close to the Bandama River, west of Bouaké, in 1922 (Schwarz 1927); these specimens are now in the Natural History Museum, London. Prior to Schwarz's description, only two kinds of black-and-white colobus monkey were generally considered to be present in the Upper Guinea forests of West Africa (see, for example, Elliot 1912). Schwarz (1927, 1929) refers to these two forms as *C. polykomos polykomos* from the western forests ("from French Guinea through Sierra Leone and Liberia to the western part of Ivory Coast", i.e., to the west of the Sassandra-Nzo rivers) and *C. polykomos vellerosus* from the eastern forests ("the Gold Coast and Togoland", i.e., to the east of the Bandama River).

Lowe (1932) described travelling by train to Bouaké from Abidjan, then going cross-country to Beoumi (shown in Fig. 1). From Beoumi he went a short distance due west to the Bandama. Today, Beoumi is on the shore of a lake created by a hydroelectric dam constructed downstream at Kossou in 1973, so Lowe's collecting locality is almost certainly now under water. We know of five other specimens in the London museum, collected in 1953–1954 by Angus Booth from between 25 and 40 km west of Daloa (6°53'N, 6°27'W), and one specimen in the Tervuren museum, also from west of Daloa.

One of us (JFO) has examined the specimens attributed to *dollmani* and found them to be somewhat variable in their pattern. The color pattern on the skin of the male type specimen (ZD.1923.2.3.4) is intermediate between *polykomos* and *vellerosus*. The ruff of hair around the face is slightly more silvery and wispy than the pure white and denser ruff of *vellerosus*, there are a few scattered white hairs on the shoulders (compared to abundant silvery shoulder hairs in *polykomos*),

and the thighs are black—lacking the silvery-white patch of hair that is typical of *vellerosus*. The other specimens vary in the number of white hairs on the shoulders, but all lack a white thigh patch. Typical examples of *C. polykomos* and *C. vellerosus* are shown in Figure 2.

During his collecting trips in the early 1950s, Angus Booth observed what he referred to as *C. p. dollmani* on the western edge of the Bouaflé Forest Reserve, near Daloa (Booth 1954). Part of the Bouaflé reserve was later incorporated into the Marahoué National Park, created in 1968 (I. Herbinger pers. comm.). In this area, Booth reported collecting one specimen of *C. p. vellerosus* and one of *C. p. dollmani*. Booth noted that the "dollmani" specimen had a faint white thigh stripe (in that respect resembling *vellerosus*), and that the "vellerosus" specimen had a white patch on either knee. He commented: "Both these specimens are indicative of genetic instability which can with certainty be ascribed to hybridisation" (Booth 1954). These two specimens (ZD.1956.347 and ZD.1956.356) are also in the Natural History Museum, London, and are listed in the *Catalogue of Primates in the British Museum (Natural History)* as "*Colobus polykomos vellerosus?*" with a note that one (ZD.1956.347) is a hybrid between *vellerosus* and *dollmani* (Napier 1985). Groves *et al.* (1993) suggested that, in the absence of a skull, ZD.1956.347 might be a juvenile that has not yet developed a full thigh stripe. We have examined this specimen and are convinced that it is a small adult female; Booth's own collecting label identifies it as an adult and its recorded weight of 6.2 kg is within the range of adult female body weights for West African black-and-white colobus (Delson *et al.* 2000).

In his later, classic paper on the zoogeography of West African primates, Booth (1958) argued that *dollmani* is closer to *polykomos* than to *vellerosus* in appearance and that it may be "either a semi-stabilised hybrid swarm, or, in view of the great ecological plasticity of the group, be descended from populations which evolved independently in a relic patch of Woodland or Riparian Forest during the postulated dry period" (Booth 1958). Booth is here referring to the last dry climatic phase of the Pleistocene.

Based on a study of male loud calls, combined with other information, Oates and Trocco (1983) elevated *polykomos* and *vellerosus* to the status of separate species. Given the variability in coat pattern in the museum specimens attributed to *dollmani* and the relatively small area over which the specimens had originated, Oates and Trocco treated these specimens as representing a hybrid zone between *Colobus polykomos* and *C. vellerosus*, rather than as a distinct taxon. Subsequently, Groves *et al.* (1993) carried out a very careful analysis of pelage and cranial features of the specimens catalogued as *dollmani* in the Natural History Museum, London, comparing them to specimens of *polykomos* and *vellerosus* in that museum and to two additional skulls of *polykomos* in Karlsruhe. They concluded that the hypothesis that the population known as *Colobus polykomos dollmani* "are hybrids and not a true subspecies, is strongly supported" and they provided an evolutionary model for how the hybridization could have

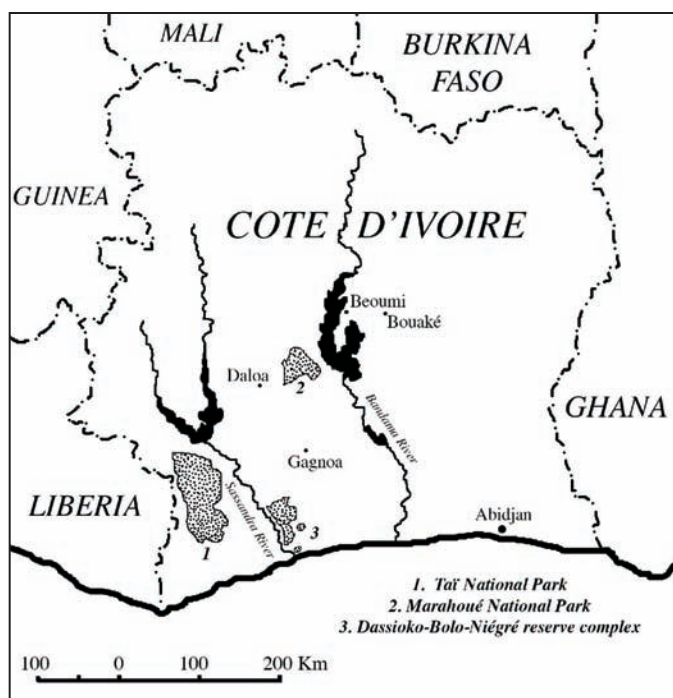
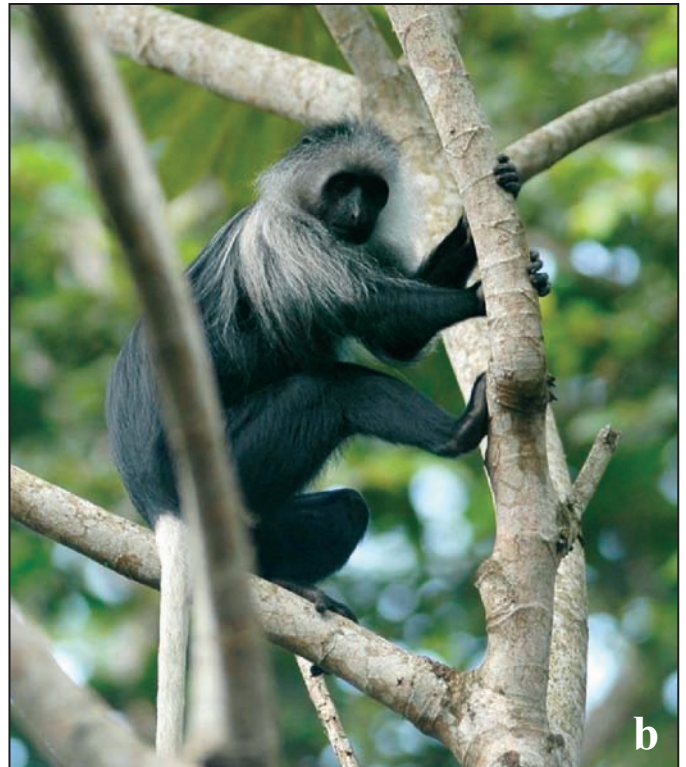


Figure 1. Map of Côte d'Ivoire showing localities mentioned in text. Drawn by Stephen D. Nash.

occurred. They recommended “further surveys of this genetically and evolutionarily significant area, followed by a strenuous conservation program for any populations that remain.”

The survey reported by Sery *et al.* (2006) is a good follow-up to the recommendations of Groves *et al.* (1993), but it leaves unresolved the status of the “dollmani” population (whose possible hybrid nature is acknowledged by Sery *et al.*). Between the Sassandra and Bandama rivers, Sery *et al.* were only able to get evidence of the continued survival of black-and-white colobus monkeys in the Bolo-Dassiéko-Niégré forest complex, quite close to the coast, and about 120 km south of Daloa and Marahoué (the region where the specimens attributed to *dollmani* originated). Booth (1954) has a map showing symbols for the presence of *vellerosus* to the west of the lower Bandama River, including one in the approximate vicinity of Gagnoa (6°08'N, 5°57'W). Gagnoa is about 80 km from the northern (5°26'N) and the eastern (5°47'W) limits of the Bolo-Dassiéko-Niégré forest complex (see Fig. 1). Based on Booth's map, it might be expected that the form of black-and-white colobus monkey inhabiting this forest complex is *Colobus vellerosus* rather than *dollmani*. Apart from obtaining local reports of their presence, Sery *et al.* (2006) only observed black-and-white colobus monkeys in the western block of the Bolo Forest Reserve; these monkeys are said to have had white bands on their thighs (G. B. Sery in correspondence with D. Zinner pers. comm.), which is a key feature of *vellerosus* (Fig. 2c) consistently lacking in *dollmani*.

In the absence of other conclusive evidence we are not convinced that colobus monkeys with affinities to *dollmani* occur in the Bolo-Dassiéko-Niégré forest complex south of their previously known range. Sery *et al.* (2006) paint a very bleak picture of the state of Marahoué National Park in terms of habitat destruction and hunting; it may well be that no black-and-white colobus still survive there, in the heart of the past known range of *dollmani*. It appears that further surveys are urgently needed, both to ascertain if any black-and-white colobus monkeys occur anywhere within the previously known range of *dollmani* (i.e., in or to the immediate west of the Marahoué National Park) and to clarify the affinities of any other black-and-white colobus surviving between the Sassandra and Bandama rivers, including the Bolo-Dassiéko-Niégré forest complex. If further studies can locate a population whose pelage suggests close affinity with *dollmani*, we agree with Groves *et al.* (1993) and with Sery *et al.* (2006) that strenuous efforts should be made to protect this population because of its evolutionary significance. However, if further surveys should show that the remaining black-and-white colobus monkey population in the Bolo-Dassiéko-Niégré complex are members of *Colobus vellerosus*, and no colobus can be located in or around Marahoué, then the population that has been called *dollmani* may now unfortunately be already extinct.



Figures 2a and 2b. *Colobus polykomos* from the Taï Forest, Côte d'Ivoire. Photographs by W. Scott McGraw.



Figure 2c. *Colobus vellerosus* from the Boabeng-Fiema Monkey Sanctuary, Ghana. Photograph by Julie Teichroeb.

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Chimpanzees are Close to Extinction in Southwest Nigeria

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Abstract: A survey to assess the distribution and status of chimpanzees in the southwest region of Nigeria was conducted in order to identify areas where effective conservation action could be taken. Seventeen sites in five states were surveyed. Information was gathered through directed searches, using hunters as guides, and through interviews with local hunters, community members and government personnel. Findings suggest that chimpanzees survive at only about half the survey sites, and that the viability of these remaining populations is in doubt. According to interviewees, chimpanzees were present at all the survey sites 10 to 15 years ago. These findings suggest that in southwest Nigeria as a whole, their population size and distribution has sharply declined over the last decade. Recommendations include the formal and effective protection of some of these sites.

Key words: Chimpanzees, habitat loss, logging, hunting, protected areas, conservation

Introduction

Recent genetic evidence on the evolution of chimpanzee populations consistently divide West African chimpanzees into two separate groups, one in the westernmost Upper Guinea forest (*P. t. verus* Schwarz, 1934) and the second further east in Nigeria and western Cameroon, referred to as *P. t. vellerosus* (Gray, 1862) by Gonder *et al.* (2006), although Oates *et al.* (2009) have shown that the correct name is *P. t. ellioti* (Matschie, 1914). The distributional limits of the Upper Guinea and the Nigeria-Cameroon chimpanzees remain poorly understood; mtDNA of hair samples taken from chimpanzees in southwest Nigeria grouped with one subspecies according to some tree building models, but with the other when using other models (Gonder *et al.* 2006). Resolving the evolutionary relationship of southwest Nigerian chimpanzees is important in planning a strategy that can ensure the conservation of representative diversity in the living great apes, but resolving this issue is made difficult because chimpanzees in this region are rare, and samples of their hair and feces are hard to find.

Nigeria is one of fifteen countries scoring highest for primate species richness, nine of which are in Africa (Cowlshaw and Dunbar 2000, in Chapman *et al.* 2006). However, Nigeria also has the largest human population in Africa, approximately half of which occurs in the southern moist forest zone,

within the historic range of chimpanzees (Oates *et al.* 2003). Much of the remaining forest habitat is contained within state forest reserves, established during the colonial period to ensure timber supplies and safeguard watersheds. Like many developing nations, Nigeria depends on its natural resources to generate much needed revenue. Combined with a relatively high level of economic development, due in large part to oil revenues, this has led to continuing high rates of uncontrolled forest conversion and natural resource exploitation. In southwest Nigeria, aided by the flatness of the terrain, this has led to highly fragmented chimpanzee habitat.

Dramatic declines in chimpanzee populations have already been documented for *P. t. verus* in Ivory Coast (Campbell *et al.* 2008) and for *P. t. troglodytes* in Gabon (Walsh *et al.* 2003). Limited surveys of southwest Nigeria have been conducted in Ondo State (Agbelusi 1994), at Omo Forest Reserve, Ogun State (Perrson and Warner 2003), at Ise Forest Reserve in Ekiti State (Ogunjemite *et al.* 2005) and on the eastern edge of the Niger delta (Bocian 1999).

The Conservation Action Plan for West African Chimpanzees (Kormos *et al.* 2003) identified the forests of southwest Nigeria as of highest priority for a survey to assess chimpanzee status. Only one small national park in southwest Nigeria, Okomu National Park in Edo State, affords protection to its wildlife, but no wide-ranging primate survey of this region has been conducted since 1982 (Anadu and Oates 1982).

The 1982 survey concentrated on the white-throated monkey *Cercopithecus erythrogaster* and the forests of Bendel State (now Edo and Delta States). A better knowledge of the abundance and distribution of chimpanzees and other species, the threats to their survival, and the challenges facing conservation policy makers, is vital for the establishment of a realistic and effective conservation strategy. This paper therefore reports on a six-month survey that commenced at the start of 2006 in southwest Nigeria, the main goal of which was to obtain information on key areas where chimpanzees survive in viable numbers, in order to plan for more effective conservation of these areas.

Methods

The survey covered 17 sites (14 forest reserves, one game reserve, one national park, and one privately-owned, former forest reserve) in five states (see Table 1 and Fig. 1). The natural vegetation of most of these sites is lowland moist forest and swamp forest, although the one game reserve surveyed further north fell within the transitional zone between high forest and true Guinea savannah. Termed 'derived' savannah, this habitat was mostly composed of dry savannah woodland as a consequence of anthropogenic changes.

Information was collected through directed searches of each site and, where possible, interviews of local hunters,

timber fellers, community members and government personnel. Broad area-reconnaissance surveys of each site were implemented, involving walking slowly and quietly through the forest along existing tracks, such as logging roads and hunting paths. A more systematic methodology could not be implemented because of time constraints and because very little was known about most of the sites, for example, the state and degree of habitat fragmentation of each. Up-to-date maps were not available from the forestry departments. Maps that dated back at least 30 years that show the original logging compartments were available for only a few sites. The amount of time spent at each site varied, but in most cases was limited to just a few days (Table 1). This survey can, therefore, only be considered a rapid assessment. It depended on finding reliable guides, and hunters, who remained suspicious of us, were reticent to act as guides, resulting in the inadequate sampling of some of the sites. Because I was unable to dispel suspicions as to the project's true motives, discussions remained informal (that is, the author did not write down the answers given by guides in front of them or present them with a systematic questionnaire). Hunters were primarily used to direct the author to areas where they had observed chimpanzees previously. The few available maps were also used to focus research on areas where natural forest remained.

I was usually able to confirm the presence or probable absence of chimpanzees (there were only a few exceptions),

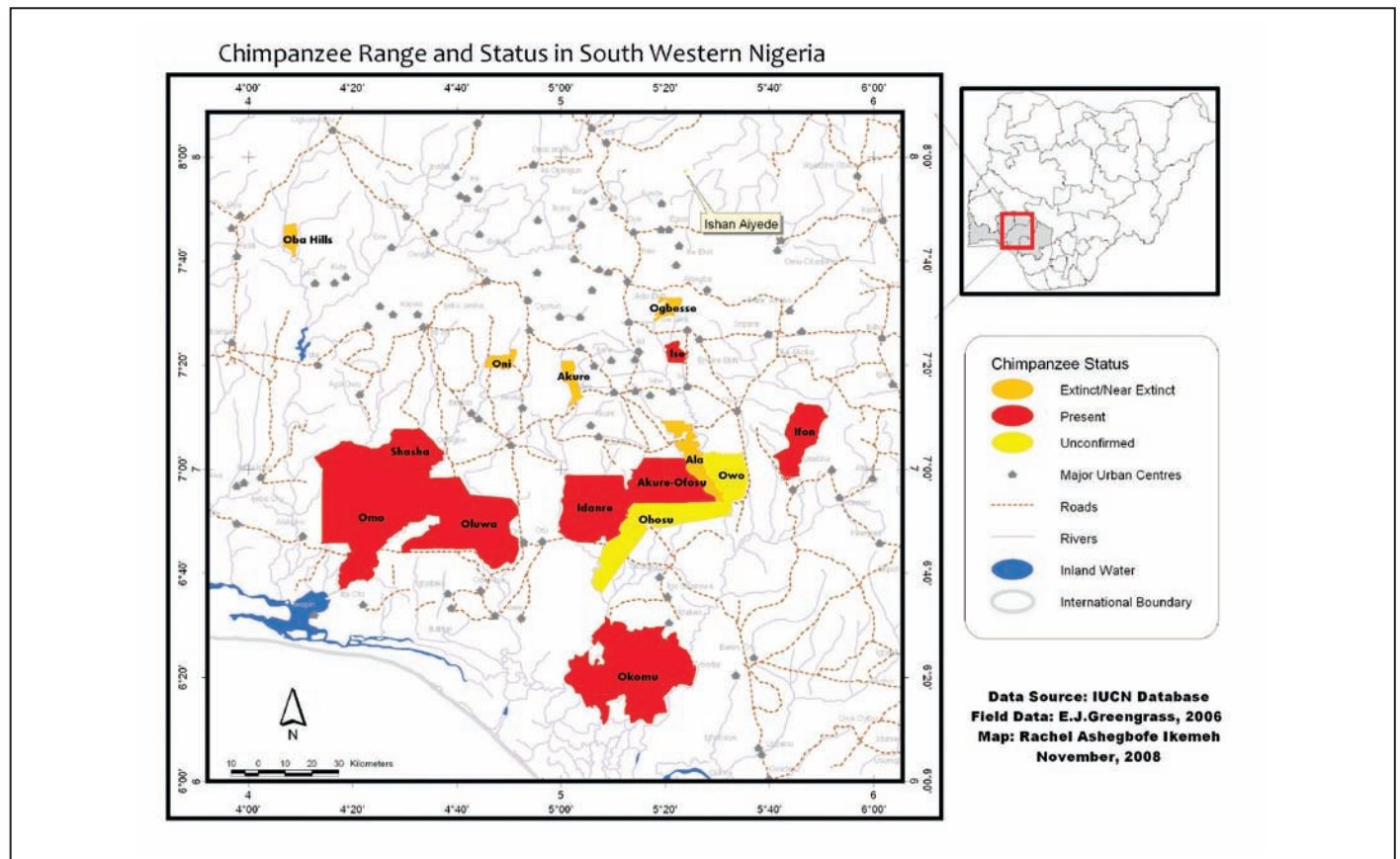


Figure 1. Chimpanzee distribution and status in southwest Nigeria. The former Ologbo Forest Reserve, located approximately 60–80 km southeast of Okomu, is not shown on this map. Note that even where chimpanzees are reported as present, their distribution is patchy due to the fragmentation of their habitat.

and where present, was able to obtain some information on their distribution and status. Hunters were the best guides, providing the most accurate information, and answering questions related to chimpanzee presence reliably. Observations of habitat quality and the evidence of chimpanzee presence that I collected complemented their information. Abundance was more difficult to assess though, and questions related to abundance were not answered reliably, although habitat quality provided indirect clues to population status.

Confirmation of chimpanzee presence was obtained directly, from sightings, vocalizations and drumming, or indirectly from finding dung piles, discarded food wadges and nests. A Garmin GPS 12 XL was used to record the locations

of chimpanzee presence, but it did not work well under the dense forest canopy and as a consequence very few recordings were made. The chances of gathering direct evidence were maximized by camping inside the forest whenever possible, as chimpanzees are most active and most vocal in the early morning and late evening.

Results

There was considerable variation in the quality of the habitat at each site. There is no primary forest left in southwest Nigeria. Larger sites still had blocks of actively exploited secondary forest at their centers, but two of the smaller sites had

Table 1. Inferred status of chimpanzees at different survey sites in southwest Nigeria.

Site	Legal status	Size (ha)	No. of days in the field	Directed search with hunter	No. of nest groups encountered	Other signs	Inferred status
Ondo State							
Idanre	Forest Reserve	54,100	6	Yes	2 (one group of 2 nests, one group of 1 nest)	Vocalization: pant-hoot	Present
Akure-Ofosu	Forest Reserve	40,100	2	No	0	None	Probably present based on hunters' reports and remaining forest cover
Ala	Forest Reserve	19,900	3	Yes	0	None	Extinct / near extinct
Owo	Forest Reserve	24,100	1	No	0	None	Unconfirmed, insufficient information gathered
Ifon	Game Reserve	28,200	2	Yes	0	Food remains	Present
Oluwa	Forest Reserve	82,800	4	Yes	Unknown (5 nests of various ages observed along river)	Sighting of adolescent male; vocalization: food grunt	Present
Akure	Forest Reserve	7,000	2	No	0	None	Extinct / near extinct
Osun State							
Shasha	Forest Reserve	9,000	1	No	0	None	Probably present based on hunters' reports
Oba Hills	Forest Reserve	6,800	5	Yes	0	Vocalization of annoyance	Extinct / near extinct
Oni	Forest Reserve	5,600	1	No	0	None	Extinct
Ekiti State							
Ogbesse	Forest Reserve	7,500	1	No	0	None	Extinct
Ise	Forest Reserve	6,200	6	Yes	2 (5 nests each)	None	Present
Ishan-Aiyede	Forest Reserve	c.7,000	1	No	0	None	Unconfirmed, not enough information gathered
Edo State							
Okomu	National Park	21,200	11	National park guide	2 (one group 2 nests, other group 1 ground nest)	Food remains; feces	Present
Ohosu	Forest Reserve	47,100	1	No	0	None	Unconfirmed: not enough information gathered
Ologbo	Privately owned former Forest Reserve	4,000	3	Yes	1 (1 nest)	None	Present
Ogun State							
Omo	Forest Reserve	130,500	7	Yes	2 (each group 1 nest each)	Vocalization: scream; drumming; feces	Present

been completely converted to farmland, with no natural forest remaining. While eight of the 14 forest reserves, in theory, make up two contiguous forest blocks (the Idanre, Akure-Ofosu, Ala, Owo, Ohosu complex and the Omo, Oluwa, Shasha complex), in reality the remaining natural forest within these reserves is becoming increasingly fragmented and disturbed as a result of unregulated and unsustainable timber extraction practices. Factors affecting the quality of the habitat at each site included their size and their distance from major towns and roads. Southern Nigeria has a well developed road network with some forest reserves traversed by major roads. Illegal squatter farming camps invariably follow logging activity. While these camps may have originated as hunting camps in the recent past, they have now evolved into permanent farming settlements. Hunting activity is prevalent at all the sites but it is the combined influence of all these activities that threaten remaining wildlife populations.



Figure 2. Illegal timber fellers at the former Ologbo Forest Reserve (4,000 ha), Edo State, southwest Nigeria. Today a private reserve. Photograph © Elizabeth J. Greengrass. July 2007.



Figure 3. 'Derived' savannah that has replaced much of the rain forest in southern Nigeria today. These grasslands are the direct result of unsustainable logging and farming practices over an extended period of time. Photograph © Elizabeth J. Greengrass. 2008.

Chimpanzees were confirmed at seven sites based on the direct and indirect evidence collected by the author. Chimpanzees were probably present at two more sites, based on the reports of hunters and the quality of the habitat (see Table 1 and Fig. 1). Chimpanzees were extinct at two sites that had been completely converted to farmland and where no natural forest remained. At three more sites chimpanzees were either extinct or very close to extinction. Forest conversion was such that very little natural habitat remained, and could not have supported more than a few individuals. At Akure Forest Reserve, for example, the only natural forest remaining is the Queen Elizabeth Plot (2–5 km² in size). Despite the proximity of humans, many of whom hunt inside the plot, people reported seeing a chimpanzee on the forest edge on a number of occasions. Chimpanzee-like vocalizations were also heard in one of only two forested gullies remaining at the Oba Hills Forest Reserve. The reserve has been almost entirely converted to plantations and farms, and the persistence of a few isolated individuals may be due only to the species' natural longevity.

By nature, chimpanzees are extremely vocal and come together to socialize in large parties on a regular basis. The pant-hoot, the chimpanzee's long distance call, helps to keep individuals that are not in close proximity in contact with each other. Despite this, however, evidence of the presence of chimpanzee presence and evidence of social cohesion among individuals was rare during this survey. Only at Idanre Forest Reserve were chimpanzees heard to pant-hoot. The largest nest site observed (at Ise Forest Reserve) was composed of only five nests, suggesting that larger social groups were rare or absent. Drumming was heard only once at Omo Forest Reserve (in response to a female's scream when she observed researchers). The paucity of such evidence can be explained by the fact that human pressure and disturbance were severe at six of the seven sites where the presence of chimpanzees was confirmed.

Because this was the first time that most of these sites had been surveyed for chimpanzees, the rate of decline and significance could not be quantified. However, according to interviews with hunters and local communities, chimpanzees survived at all the survey sites 10–15 years ago. This indicates that chimpanzees have disappeared from about half the survey sites within a very short period of time. In southwest Nigeria as a whole their population size and distribution has sharply decreased within the last decade.

Discussion

The rapid decline in the chimpanzee population that this survey identified coincided with a boom in timber production during the 1990s. At this time, new, more efficient sawmills were established, encouraging the timber industry to grow and local economies to become even more reliant on the trade. Most, if not all, timber operations are illegal, unregulated and unsustainable. At one of the survey sites (Ise Forest Reserve) Ogunjemite *et al.* (2005) found that the rate of exploitation was over five times higher than the rate of forest regeneration,

and logging pressure was intense and increasing. However, the timber industry is highly organized and illicitly supported at all levels of government, and where corruption is institutionalized there is little incentive to correct this. Legitimate companies practicing long-established working plans that were in the majority 30 years ago are now all but absent from



Figure 4. The broken forest canopy, partly the result of past logging, on the periphery of the Okomu National Park (21,200 ha), Edo State, southwest Nigeria. The forest improves as you travel further into the park and is probably the best example of mature secondary forest in southwest Nigeria. Photograph © Elizabeth J. Greengrass. 2008.



Figure 5. The canopy at the former Ologbo Forest Reserve (4,000 ha), Edo State, southwest Nigeria. The forest in this reserve, now privately owned, is badly damaged, and one of the poorest examples of mature secondary forest in the region. Photograph © Elizabeth J. Greengrass. 2007.

the region. The findings of this survey that natural resources in southwestern Nigeria are being over-exploited at an unsustainable rate are not new and have been reported by Agbelusi (1994), Anadu (1987), Anadu and Oates (1982), and as far back as the 1960s (Petrides 1965 in Anadu 1987). Clearly, many wildlife species, chimpanzees included, have been threatened by widespread habitat loss for a very long time.

The contribution of hunting to the decline of the chimpanzee population in recent times is less clear, but almost certainly hunting pressure peaked with timber production because threats to primate populations are correlated (Chapman *et al.* 2006). Nowadays, however, hunting appears to have lessened in intensity as a direct consequence of over-exploitation in the recent past. In many parts of southwest Nigeria, hunters do not hunt on a full-time basis. Hunting camps in forest reserves are becoming rare, or have evolved into agricultural settlements. While in the early 1980s, a large proportion of the Nigerian population from all income groups ate bushmeat regularly (reported as 50% by Chapman and Peres [2001] or 80% by Anadu [1987] from the same source), nowadays the consumption of bushmeat is quite rare, at least in the southwest. Some bushmeat is still sold at markets along major roads but the species offered are typically limited to those that persist even under heavy hunting or that also thrive in degraded habitats, such as Maxwell's duiker (*Cephalophus maxwelli*) and the grasscutter or cane rat (*Thryonomys swinderianus*).

While the survey demonstrates the ability of chimpanzees to persist under conditions of high human pressure and disturbance, the viability of these remaining populations is in doubt. Timber trees can contribute disproportionately to the diets of some primate species (Chapman and Peres 2001), indicating that logging has a severe impact by reducing food availability. While chimpanzees do have flexible diets, their energy requirements as large primates with large home ranges predisposes them to a reliance on high energy fruits. Diet quality and feeding efficiency may determine certain aspects of sociability and female reproductive success (Greengrass



Figure 6. Forest cutting and burning by subsistence farmers at the former Ologbo Forest Reserve (4,000 ha), Edo State, southwest Nigeria. Photograph © Elizabeth J. Greengrass. 2008.



Figure 7. A chimpanzee knuckle print preserved in wet sand along a logging road at the former Ologbo Forest Reserve, Edo State, southwest Nigeria. This picture was taken in May 2008, after active law enforcement had been implemented and loggers were forced to leave. Photograph © Elizabeth J. Greengrass.

2005), and a recent paper showing that core area quality is associated with variance in reproductive success (Emery-Thompson *et al.* 2006) suggests that, by lowering the quality of their habitat, logging pressure may affect the reproductive success of female chimpanzees. A reduction in food availability associated with logging may decrease the animals' condition and increase their vulnerability to disease or parasites (Milton 1996 in Chapman *et al.* 2000) and also increase infant and juvenile mortality. Despite their behavioral adaptability, chimpanzee communities have defined territories that limit the individuals' abilities to avoid widespread human-induced disturbance.

The formal and active protection of some sites through a combination of law enforcement and revenue-generating activities is strongly recommended. While there is scope for the development of a domestic tourism industry, carbon credit schemes may be a more feasible means by which stakeholders in Nigeria can benefit from protecting their natural resources. Since this survey was conducted, further survey work has been carried out in the Omo, Oluwa, Shasha complex (Oates *et al.* 2008) and plans are presently underway for longer-term protection of that area. Since 2007, as part of the Round table on Sustainable Palm Oil (RSPO) initiative, the former Ologbo Forest Reserve has also been protected by a private oil palm company. While the project initially met with a great deal of success in stopping illegal logging, problems related to corrupt logging cartels working in the area that have a large influence at both the local and state level, and the company's failure to recognize the importance of employing a specialist to manage the protected area, will weaken the project in the long-term.

This is the third catastrophic decline in a chimpanzee population documented within the last decade. The severe declines documented for *P. t. troglodytes* (see Walsh *et al.* 2003) and *P. t. verus* (see Campbell *et al.* 2008) in Gabon

and Côte d'Ivoire, respectively, occurred in countries that were believed to be the last strongholds for these subspecies. These surveys record a global decline in ape populations over the last two decades that suggest that conservation efforts are not advancing as much as they should. My results suggest that chimpanzees in southwest Nigeria are now on the brink of extinction. Unless effective action is taken, they will be extinct within the next few years in most of the sites where they were found to remain during this survey.

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Chimpanzee Tourism in Relation to the Viewing Regulations at the Mahale Mountains National Park, Tanzania

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Abstract: We investigated the current situation of chimpanzee tourism at the Mahale Mountains National Park by direct observations of tourists, tour guides, and trackers. The median number of people in a tourist group was seven, which is within the number allowed by park regulations; however, the actual number was sometimes as large as 11 because groups occasionally included several trackers and guides in addition to six tourists. On 23.1% of 121 observation days, the number of groups observing chimpanzees was greater than that permitted by park regulations. This resulted in as many as 39 people visiting one group of chimpanzees in a day. Problematic behaviors on the part of tourists included moving too close to chimpanzees, observing chimpanzees without authorized park guides, and simultaneous observation by two or more groups of tourists, among others. Based on these analyses, we argue that better control of tourist groups is needed in order to reduce the potential negative impacts of tourism on chimpanzee health and behavior.

Key words: Chimpanzee, *Pan troglodytes schweinfurthii*, ecotourism, Mahale Mountains National Park, number of tourists, behavior of tourists

Introduction

All great apes, including chimpanzees (*Pan troglodytes*), are either endangered or critically endangered (Beck *et al.* 2001; Caldecott and Miles 2005). Because of their evolutionary proximity to humans and the behavioral similarities between humans and apes, people all over the world are fascinated by great apes and wish to see them in the wild (Lonsdorf 2007). In many parts of Africa, the result has been the development of tourism, the main goal of which is to observe habituated chimpanzees or gorillas in the wild. This form of tourism is often regarded favorably in conservation terms as ecotourism that provides alternative income for local people who would otherwise use forests for agriculture, housing materials, firewood, and medicinal plants and sometimes hunt wild animals, including great apes, for meat. Such tourism can serve as an important source of funds for the conservation budget. For example, in Uganda, tourism has become the principal internal source of foreign exchange, and chimpanzee and gorilla tourism are responsible for 52% of the tourism revenue (Wrangham 2008).

Although there are many definitions of ecotourism (Fennell 2001), it usually includes the notion of sustainability,

which means there should be no or minimal negative impact on the target species or the natural environment (for example, Ceballos-Lascrain 1996). However, in many places, ecotourism has become an elusive term (Mugisha 2008) or merely a catchphrase (Nishida and Nakamura 2008) because, in reality, more weight is sometimes placed on increasing revenue than on minimizing human impact on nature or improving the well-being of local people. For example, Krüger (2005) analyzed 251 case studies of ecotourism and concluded that more effective control of tourist numbers and management of their distribution are needed for ecotourism to serve the purpose of promoting conservation.

Perhaps the most serious negative impact of tourism on great apes is the potential for the transmission of diseases from humans (for example, Boesch 2008). In Mahale and Gombe, diseases, especially infectious diseases, are the primary cause of chimpanzee deaths (Nishida *et al.* 2003; Williams *et al.* 2008), and outbreaks have killed as many as 12 chimpanzees at one time (for example, Hanamura *et al.* 2008). Although the cause of death could not be precisely determined, a virus that was formerly known only in humans has been detected

in samples collected from chimpanzees at the time of disease outbreaks (Kaur *et al.* 2008; Köndgen *et al.* 2008). If fatal diseases are transmitted from humans to chimpanzees, then presumably transmission risk becomes higher when the number of people who visit a group of chimpanzees becomes larger, in addition to other factors such as the duration of the visit and the proximity of approach.

Another negative impact of tourism on chimpanzees is that the presence of humans may cause chimpanzees to alter their natural behaviors. Reports have documented the behavioral changes that occur as great apes become habituated to humans (for example, Blom *et al.* 2004; Johns 1996). However, even for well-habituated chimpanzees, the presence of too many visitors may have negative effects, such as physiological and psychological stress (Wallis and Lee 1999).

Usually, each protected area has regulations or at least general rules that are designed to control the number of tourists and their behaviors. Such regulations specify the number of groups permitted per day and the number of people permitted in a group. However, the existence of regulations is not the same as their implementation, and a realistic appraisal is needed of the practices and behavior of tourists, guides and park staff involved in ape tourism.

The only quantitative research addressing the details of great ape tourism was conducted on mountain gorillas of the Bwindi Impenetrable National Park in Uganda, reported by Sandbrook and Semple (2006). The authors interviewed tourists after their return from viewing the gorillas and asked them how close they had been to the gorillas. They found that the distance between the tourists and the gorillas was significantly less than that permitted by park regulations, providing a warning that the existence of regulations does not necessarily reduce potential problematic behaviors.

History of tourism at Mahale

The habituation of chimpanzees at Mahale for research purposes began in 1965 (Nishida 1968), and the research has continued since then. Initially, two groups (or communities) of chimpanzees were habituated for research, but one of them became extinct in the 1980s (Nishida *et al.* 1985). Only one habituated group of chimpanzees is, therefore, currently accessible at Mahale for both research and tourism purposes.

In 1987, small-scale chimpanzee tourism was introduced to Mahale; up to six visitors at a time were accepted into a tent camp and allowed to spend up to one hour observing the chimpanzees. In the mid-1990s, Tanzania National Parks (TANAPA) began to accept tourists in its own tourist camp. In the early 2000s, two more companies joined the tourist business at Mahale, increasing the number of tourists in the park from only 200 per year in the early 1990s to more than 1,000 in 2005 (Nishida and Nakamura 2008). Usually each tourist stays three to four days (sometimes a week), thus the total man-days is much larger.

As competition developed among commercial tourism companies at Mahale, TANAPA and the Frankfurt Zoological Society (FZS) requested a preliminary assessment of the

tourism there (Walpole 2004). However, the assessment was insufficient because it was conducted during a relatively low tourist season and lasted for only one week. Furthermore, because the chimpanzees were in the higher part of their range at the time, the assessor did not observe any chimpanzees during the week, and was consequently unable to observe how the tourists were behaving when observing chimpanzees.

Other reports from Mahale include annual statistics on the number of tourists visiting (Nishida and Mwinuka 2005) and reports of researchers' impressions of an overabundance of tourists (Nakamura and Nishida 2004; Nishida and Nakamura 2008), although these impressions were not accompanied by empirical data. Thus, the aim of this study was to gather quantitative data on chimpanzee tourism at Mahale by direct observation of tourist groups that were visiting the chimpanzees and to assess how closely the procedures and behaviors of the tourists, guides and trackers complied with the park regulations.

Regulations of tourism at Mahale

According to the park regulations (Tanzania National Parks 2006), maximum viewing-group size is seven, including six tourists and one TANAPA guide. In addition to these seven, one tracker and one tour-camp-provided guide may accompany the tracking excursion, but once the chimpanzees are sighted, only the six tourists and the TANAPA guide may approach the chimpanzees. The tracker and tour-camp-provided guide must remain 200 m away from the chimpanzees all the time. The figure is similar to gorilla tourism where limitation is set at six in Uganda and eight in Rwanda and Democratic Republic of Congo (Homsy 1999).

The maximum number of groups per day permitted to observe the chimpanzees is three but they have to be timed so that two or more groups do not visit the chimpanzees at the same time. As such, the park regulations allow 18 tourists and 21 people in total to visit the chimpanzees per day. This figure is different from gorilla tourism where only one visit per day per group is permitted (Homsy 1999). In addition to these tourists and guides, three researchers and three research assistants are permitted to follow the M group chimpanzees.

There are regulations that stipulate the distance of the observers (10 m) from the chimpanzees, observation duration (one hour), restriction of flash lights, eating in the presence of or near chimpanzees, defecating in the forest, leaving belongings unattended. The regulations do not stipulate a minimum or maximum chimpanzee party size for visitation.

Methods

The study was conducted from July 2006 to October 2008 at the Mahale Mountains National Park, Western Tanzania. For more details about the site see Nishida (1990). While observing chimpanzees of the M group, the first author (MN) and a research assistant opportunistically recorded the number of tourist groups that visited the chimpanzees and the number of people within each group, sorted into categories

(i.e., tourists, official park guides, private camp trackers, and guides employed by tourist companies). Such data collection was possible because we personally recognized each park guide, camp tracker, and camp guide. We also recorded the tourists' behaviors, their distance from chimpanzees (estimated visually), and any apparent health conditions (such as coughing) among the tourists that might be potential threats or disturbances to the chimpanzees. Both observers were well accustomed to the normal behaviors of chimpanzees, allowing us to monitor each chimpanzee's location (as visitors sometimes unintentionally disturb chimpanzees in the nearby bush) and the chimpanzees' responses to tourists. Due to the observation conditions, we were not able to record all of these data for each tourist visit. In calculating medians, unconfirmed data were excluded from the analysis. We did not encounter all tourist groups because sometimes observers and tourists followed different parties of chimpanzees. Thus, the number of groups per day and the number of people in a group may be slightly underestimated. In addition to these data, information on tourists recorded in camp diaries was used to supplement our data (for example, when other researchers observed different tourist groups or when they witnessed some violation of the regulation by tourist groups).

Results

General pattern of chimpanzee tourism at Mahale

In Mahale, three tourist camps are run by tour companies in addition to the TANAPA camp, Banda, which also receives tourists. When a camp has guests, it usually sends out one to three trackers in the early morning to locate chimpanzees. If multiple trackers are sent, they often try to locate different parties in order to offer the tourists a better option (usually, a larger party with an alpha male or with infants is preferred by tourists over, for example, a lone, low-ranking male). When the trackers locate the chimpanzees, they communicate with their camp using transceivers. Because the chimpanzees are usually on the move, the trackers continue following the chimpanzee party, sometimes for more than four hours, until the tourists arrive. As there are three tourist camps, it is often the case that three to five camp trackers simultaneously follow a party of chimpanzees, in addition to any researchers and research assistants.

Tourists are led by a camp guide (and/or a camp manager) and accompanied by an official park guide. This double-guide system was introduced because initially most of the park guides did not speak English. The camp guides, who are usually not local people but are from large cities in Tanzania, have relatively high levels of education and speak English well. After they come to Mahale, they are trained to give a general presentation about chimpanzees. Although the park guides now also speak English and some are locally hired, they do very little guide work with tourists. As a matter of fact, some of them do not have enough knowledge of chimpanzee identities and their behavior; instead, camp guides fulfill that role. A park guide accompanies the tourist group because

regulations require it, and the tourists pay a fee for the guide. Although we did not collect data on the nationality of tourists systematically, according to the identity expressed in greetings with tourists, the majority of them seemed to be from North America and Europe, some from Asia, and we met virtually no African tourists except for those from South Africa.

A tour group usually stays with the chimpanzees for a period of one hour, the time permitted by regulations, and the time is measured by the park guide. Even after the tourists arrive at a site, the camp trackers do not retreat but wait nearby, because if the chimpanzees move to an area where they cannot be observed, the trackers must follow them to locate an alternative place where the chimpanzees will be visible to the tourists.

Number of people in a tourist group

Table 1 shows the number of people observed in tourist groups. The median number of tourists (excluding guides and trackers) was five (range 1–7, $N = 233$ groups). The median total number in a group (including guides and trackers) was seven (range 1–11, $N = 218$ groups). On average, the number of people in a group was within the limit set by park regulations, but the total number per group was sometimes larger, as there were often several trackers and two guides in addition to the six tourists.

Table 1. Number of people in a tourist group visiting chimpanzees. Park regulations permit 6 tourists and 1 guide per group.

Number of people in a group	Tourists only (number of groups)	Including guides and trackers (number of groups)
1	8	2
2	39	5
3	19	10
4	39	18
5	41	21
6	84	22
7	3	44
8		43
9		40
10		10
11		3
Total*	233	218

* Total numbers differ because we could sometimes count only tourists and could not count trackers who sometimes keep distance from the tourists.

Number of tourist group visits per day

For the 121 days on which we recorded our observations, one tourist group visited on 38 days (31.4%), two on 38 days (31.4%), three on 17 days (14.1%), four on 16 days (13.2%), five on 10 days (8.3%), and six on 2 days (1.7%; Table 2). Of the days on which there were more than three tourist groups, 96.6% were concentrated between June and October (corresponding to the dry season), with only one exception in February.

Table 2. Number of tourist groups per day. N = 121 observation days.

Number of groups	Observed days	Percentage to the total
1	38	31.4%
2	38	31.4%
3	17	14.1%
4	16	13.2%
5	10	8.3%
6	2	1.6%
Total	121	

Total number of people visiting the M group per day

A median of eight (range 1–27) tourists visited per day (N = 107 days). The median total number (including trackers and guides) of daily visitors per day was 13 (range 2–39, N = 102 days).

Problematic behaviors of tourists, camp trackers, and guides

Table 3 summarizes some of the problematic behaviors of tourists. The minimum distance permitted between tourists and chimpanzees at Mahale is 10 m. Fifteen cases were observed when tourists violated this limit. In ten other cases, the groups were accompanied only by camp guides but not by authorized park guides. It is a matter of concern that in nine cases, tourists or trackers were observed coughing while they were observing the chimpanzees.

Table 3. Problematic behaviors of tourists, guides, and camp trackers.

Behaviors	Number of cases observed
Too close to chimpanzees	15
Tourists without park guides	10
Coughing	9
Disturbing chimpanzees	9
Two or more tourist groups in one place	7
Going into bush	5
Disturbing researchers	4
Using flashlights	4
Staying longer than permitted	3
Not wearing face masks	2
Unattended belongings	1
Defecating in the forest	1

Discussion

The number of tourists in a group usually fell within the six-tourist limit specified by park regulations (Tanzania National Parks 2006). Also, most groups adhered to the observation time of one hour, with only a few exceptions (see Table 3). This suggests that the tourist camps were trying to abide by the regulations regarding group size and observation time. In contrast, the number of visiting tourist groups exceeded the limit of three groups per day on 23.1% of the observed days. This figure may be even larger, given that we probably did not encounter all tourist groups. This happens

because each camp has the capacity to accommodate more than one group of six tourists per day, and if more than one tourist group want to visit Mahale, these camps comply with the requests. Even in such cases, the total number of groups in a day does not necessarily exceed three because another camp may not have guests on that day. However, the problem often occurs during the high tourist season when all camps tend to have two or more groups of tourists, and camps do not communicate with each other about the number of tourist groups each is planning to take to visit the chimpanzees on a given day. MN asked the park guides several times about this situation. The guides responded that the TANAPA authority accepted the situation and had no plans to attempt to control it until they introduce a new booking system. Although at present a median of 13 people per day may seem tolerable, as it is within the limits of the regulations, daily numbers fluctuate. The fact that 39 people could visit a group of chimpanzees on one day should not be disregarded. If the number of tourists continues to increase and additional tourists fill the days that are currently vacant, it is possible that 30 to 40 people will be visiting every day the single habituated chimpanzee group of about 60 individuals. If 39 people keep visiting every day, that would make 14,235 people per year.

The large number of tourist groups also poses a problem when two or more tourist groups meet unintentionally. This often happens when two groups of tourists arrive from different directions on an observation trail, with chimpanzees between the two groups. Such encounters may have negative impacts on chimpanzee behaviors, as some shy chimpanzees, especially some females, are stressed by being surrounded by humans. In addition, it has been reported that visitor satisfaction declines when the number of people simultaneously observing the chimpanzees exceeds six to eight (Johns 1996). The meeting of two or more tourist groups should, therefore, be avoided for this reason as well by perhaps controlling and differentiating the timing of visit to the chimpanzees.

In 15 cases, tourists were observed approaching closer than the permitted 10-m distance from the chimpanzees. This figure may seem relatively small compared to the almost daily violations of the distance rule observed in Bwindi gorilla tourism (Sandbrook and Semple 2006). The low incidence of distance violations at Mahale may be partly due to the presence of researchers at the observation location. Usually when tourists arrive, researchers are already observing the chimpanzees. Because we repeatedly ask camp guides to stay at a distance from the chimpanzees, they may not dare to go closer in our presence. In many of the 15 observed cases of distance violation, tourists and guides did not seem to notice the presence of researchers. This suggests that, in the absence of researchers, tourists might approach closer to chimpanzees even more often. The main reason that people must keep a distance from chimpanzees is to avoid interspecies transmission of diseases, but distance is also crucial for the safety of tourists. It should be required to inform tourists prior to their visits to chimpanzees why the distance rule has to be adhered to. Likewise it is necessary to prohibit tipping guides for getting closer to the apes.

Researchers previously proposed a rule that everyone who visits the chimpanzees should wear a surgical mask (Hanamura *et al.* 2006), and the rule is now accepted and followed by all camps. However, another regulation specifies that anyone who is not in good health must not visit the chimpanzees. Tourists also unintentionally disturb the chimpanzees' natural behaviors by surrounding them, or standing between a mother and her offspring. Sometimes relatively shy chimpanzees avoid people and go into dense bush or remain in the bush when people approach. In one case, a female tourist was hit by a male chimpanzee because she was unable to move out of the route of the male's charging display.

Tourists themselves may be able to reduce the incidence of the above-mentioned problems if they are well informed about the existence of, and the reasons for, the regulations. However, tourists have no direct control over another problem, which is that two or more groups may meet in the same place at the same time to observe the same chimpanzees because neither group is aware of the plans of the other group. Although park regulations specify that different tourist groups should observe different parties, because chimpanzees have fission-fusion social structures, two different tourist groups sometimes meet in the forest as a result of following different parties.

Finally, to reduce stress on the chimpanzees and to foster genuinely sustainable ecotourism at Mahale, we propose the following five recommendations.

1. The number of groups per day must be strictly limited to three. This can be achieved by introducing a unified booking system controlled by the park management. For the time being, park officials should mediate constant communication among tourist camps. If the park management needs more annual revenue, higher entrance fees or introduction of "the chimpanzee fee" (Nishida and Mwinuka 2005) will be a definite option.
2. Only one group of tourists should be allowed to visit the chimpanzees at any one time. This can best be achieved when the first recommendation (i.e., the limit on groups allowed per day) is strictly followed. The park management should coordinate and arrange the timing of observations by different tourist groups.
3. The total number of people in a group should be reduced. Capacity building of park guides is crucial. This would eliminate the need for camp guides. The presence of capable park guides will bring benefits to both tourist companies and conservation.
4. The presence of tourists should not be allowed to disturb the behaviors of chimpanzees. When chimpanzees approach them, tourists should gently step back until they have enough distance before they take photographs. Tourists must be reminded that chimpanzees have the ability to severely injure humans. Tourists should receive

better instructions before they are allowed to observe chimpanzees.

5. Some hygienic measures should be considered, such as presenting updated vaccination certificates for potentially very dangerous diseases and the washing of the soles of people's shoes to prevent bringing in or taking out germs.

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Distribution of the Green Monkey (*Chlorocebus sabaues*) in the Coastal Zone of Côte d'Ivoire

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Abstract: The green monkey (*Chlorocebus sabaues*) of West Africa ranges from the north-west coast of Senegal to the White Volta in Ghana. In Côte d'Ivoire, *C. sabaues* was thought to be mainly distributed through the savannah and savannah-forest mosaic habitats north of the rain forest zone. During primate surveys in the forest zone of southern Côte d'Ivoire we were unable to confirm the presence of *C. sabaues* in any of the forest reserves; however, we did find the species in a littoral forest outside its expected range. *Chlorocebus sabaues* was also reported from two other forests in the coastal region of Côte d'Ivoire. The discovery of these three populations is surprising and shows that there is an urgent need for more surveys in the region. The absence of *C. sabaues* in areas where it was reported 30 years ago is another alarming indication concerning the conservation of primates in Côte d'Ivoire.

Key words: *Chlorocebus sabaues*, distribution, habitat, conservation status, Côte d'Ivoire

Résumé: L'aire de répartition du callitriche (*Chlorocebus sabaues*) de l'Afrique de l'Ouest s'étend de la côte nord-ouest du Sénégal à l'est jusqu'à l'Ouest du Volta blanc à l'ouest. En Côte d'Ivoire, son aire de distribution avait été confinée à la savane et aux mosaïques savane-forêts au Nord. Au cours de l'inventaire des primates dans la zone forestière au Sud de la Côte d'Ivoire, nous n'avons pu confirmer la présence de *Chlorocebus* spp. dans aucune des réserves forestières. Cependant, nous l'avons observé dans la forêt du littoral, en dehors de son aire de répartition connue. Aussi, la présence de *Chlorocebus* spp. a été rapportée dans deux autres forêts de la zone côtière de Côte d'Ivoire. La découverte de ces trois populations est surprenante et montre qu'il y a un besoin urgent de mener davantage d'inventaires dans la région. L'absence de *Chlorocebus* spp. dans des régions où il avait été rapporté plus de 30 ans auparavant est un autre signal alarmant concernant la conservation des primates de Côte d'Ivoire.

Mots clés: *Chlorocebus sabaues*; distribution; habitat; statut de conservation; Côte d'Ivoire

Introduction

Savannah monkeys (*Chlorocebus*) are among the most widespread of the African primates and inhabit large parts of sub-Saharan Africa (Hill 1966; Wolfheim 1983; Lernould 1988; Kingdon 1997). They are found across the continent from north-west Senegal to Eritrea, Djibouti and Somalia, as well as southward over much of southern Africa. *Chlorocebus* spp. live in a wide variety of habitats but show preference for savannahs and savannah forest mosaics and are not found in the moist forests of Central and West Africa, or in the deserts of south-west Africa (Hill 1966; Wolfheim 1983; Lernould 1988; Kingdon 1997). Numerous morphotypes have been

described (Hill 1966; Napier 1981; Kingdon 1997; Groves 2001). Their taxonomic status and phylogenetic relationships remain unclear and the taxonomy of the entire genus is in urgent need of a revision (Groves 2001). Even the generic name is under discussion. Groves (2001) resurrected *Chlorocebus* Gray, 1870, but Grubb *et al.* (2003) retained the genus name *Cercopithecus* Linnaeus, 1758, regarding the former to be a synonym. In the present paper we follow Groves (2001) and accept the generic name *Chlorocebus*. Among the various taxa of the genus we find the grivet (*Chlorocebus aethiops*), which occurs from south-eastern Sudan through Ethiopia into Eritrea, the Bale Mountains monkeys (*C. djamdjamensis*) from a restricted area in the highlands of Ethiopia, the vervet

(*C. pygerythrus*) from southern Ethiopia into the southern part of Africa, the malbrouck (*C. cynosuroides*) from southern D. R. Congo, central Zambia and Angola, the tantalus monkey (*C. tantalus*) in northern central Africa from the Volta River in Ghana east into Sudan, Uganda and north-western Kenya, and the green or callithrix monkey (*C. sabaesus*) in West Africa from Mauritania and Senegal to the western bank of the Volta River in Ghana and Burkina Faso (Kingdon 1997; Groves 2001).

The ecology and behaviour of savannah monkeys have been studied mainly in East and southern Africa (for example, Struhsaker 1967; Henzi and Lucas 1980; Seyfarth *et al.* 1980; Wrangham and Waterman 1981; Cheney and Seyfarth 1983, 1987; Isbell *et al.* 1991; Barrett *et al.* 2006), with considerably less information available for *C. sabaesus* and *C. tantalus* (for example, Dunbar 1974; Galat and Galat-Luong 1976, 1977; Kavanagh 1978; Galat 1983; Harrison 1983; Nakagawa 2000, 2003). A number of studies of *C. sabaesus* have been conducted on Caribbean Islands, where introduced animals have established populations (Poirier 1972; Fairbanks 1978; Horrocks 1986; Boulton *et al.* 1996).

For Côte d'Ivoire, several authors have reported the distribution of *C. sabaesus* as being limited to the Guinean savannah north of the forest zone (Booth 1956, 1958; Hill 1966; Galat and Galat-Luong 1980; Lernould 1988; Kingdon 1997). Its presence in the Comoé National Park, north-eastern Côte d'Ivoire, for example, is well documented (Gerling and Bokdam 1973; Balzamo *et al.* 1980; Fischer *et al.* 1999-2000, 2002). The distribution of *C. sabaesus* in southern Côte d'Ivoire, however, is not well known; only Tahiri-Zagrët (1976) had reported it to occur in parts of the southern forest zone. Here we report a population of *C. sabaesus* from the littoral forest of Iles Ehotilé National Park (NP), south Côte d'Ivoire; a site outside the previously described range of this species.

Methods

Between 2000 and 2006 we conducted surveys in 23 protected forests in southern Côte d'Ivoire (for a complete list see Gonedélé Bi *et al.* submitted). All 23 forests have habitat suitable for primates (Gonedélé Bi 2008). The surveys

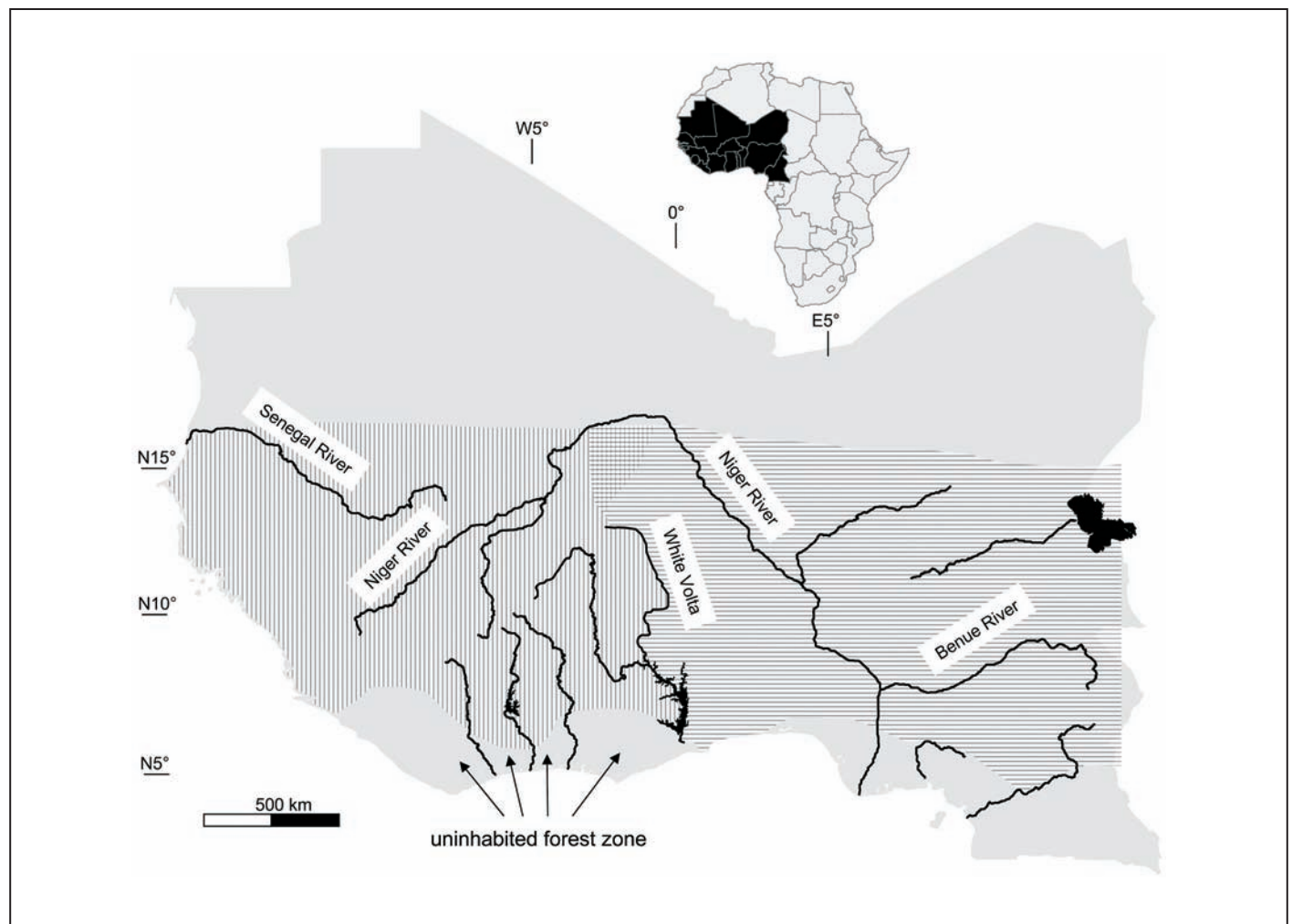


Figure 1. Distribution of savannah monkeys in West Africa, based on maps in Lernould (1988) and Kingdon (1997) (left, vertical hatching = *C. sabaesus*; right, horizontal hatching = *C. tantalus*; cross-hatching = possible overlap of both taxa). In Ghana, Côte d'Ivoire and Liberia, *C. sabaesus* is confined to the savannah habitats north of the rain forest region (but see Tahiri-Zagrët 1976).

included extensive walks in the forests, along with interviews with hunters, officials and other local people from villages surrounding the forests (for further details see Gonedélé Bi *et al.* 2006, 2008). We carried out foot surveys over 99 days in the 23 forests, with a mean of 4.13 survey days (range: 1 to 14 days) for each of the forests. In total, we covered 2,673 km.

For each survey we formed three teams, each composed of a researcher and a local guide recruited among hunters, former hunters, or staff of the local bio-monitoring programmes. The three teams surveyed different zones of the forests simultaneously so that a relatively large area was covered within a short period. Surveys normally lasted nine hours (between 06:30 and 17:30) with a break from 12:00 to 14:00. During surveys we walked slowly (1–1.25 km/hour) and quietly along old logging roads and paths. We noted all visible

or acoustic signs of primates, determined the species present and recorded geographic positions with a global positioning system. We were familiar with the appearance and behavior of *C. sabaues* from previous encounters with the species in Marahoué National Park (6°01'W, 7°07'N) and in Soko Forest Grove (2°44'W, 7°58'N) (Fig. 1). We had also seen and examined *C. sabaues* carcasses in local markets.

Results

During our survey in Iles Ehotilé National Park (550 ha) we sighted a group of five *C. sabaues* on Elouamin Island (3°18'W, 5°09'N, *c.* 95 ha), one of the five islands comprising this reserve. We also found dead specimens of *C. sabaues* among the bushmeat offered in a market in Assomlan, a

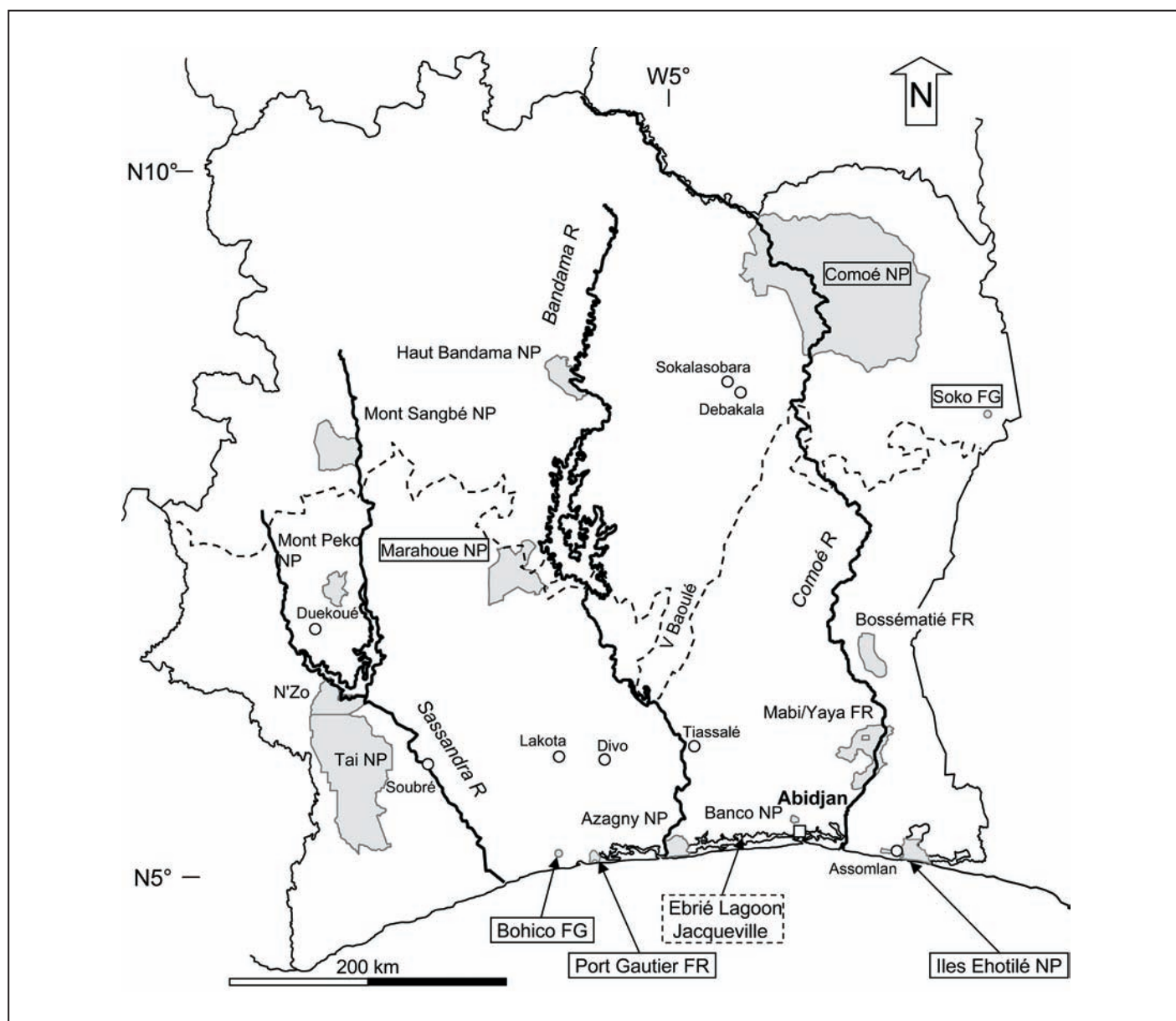


Figure 2. Geographical position of sites in Côte d'Ivoire where we encountered *C. sabaues* (Marahoué National Park, Soko Forest Grove) or where it was reported outside its expected range, (Iles Ehotilé National Park, Port Gautier Forest Reserve, and Bohico Forest Grove). Galat (1983) observed *C. sabaues* near Jacquerville. Dashed line = approximate northern limit of evergreen forest zone (Peltre 1976).

village adjacent to the National Park (c. 0.7 km from the border). The presence of *C. sabaesus* in two other areas, the Port Gauthier Forest Reserve (5°27'W, 5°08'N, c. 2,500 ha) and the Bohico Forest Grove (5°32'W, 5°08'N, c. 5 ha) was reported by several villagers. However, in the Port Gauthier Forest Reserve we failed to find *C. sabaesus* during one week of surveying. For the Bohico Forest Grove, we have only information from interviews. All three sites are in the forest zone of southern Côte d'Ivoire, outside the reported range of *C. sabaesus* (Fig. 2). We were unable to confirm the presence of *C. sabaesus* in any of the forest reserves in southern Côte d'Ivoire (see Gonedélé Bi *et al.* submitted) besides Soko Forest Grove and Marahoué National Park and the three reserves at the coast.

Discussion

The occurrence of *C. sabaesus* in the coastal forest zone of Côte d'Ivoire is surprising since, according to a number of authors, it is confined to the savannah and savannah forest mosaics of the drier northern parts of the country, such as Comoé, Marahoué National Park and Sokala-Sobara, near Dabakala (Booth 1956, 1958; Hill 1966; Galat and Galat-Luong 1980; Lernould 1988; Kingdon 1997). Galat and Galat-Luong (in press) draw the southernmost boundary of the geographical range of *C. sabaesus* at the south of the "V Baoulé" savannah (for example, near Lakota, Divo and Tiassalé) and exclude forest areas such as Tai, Duékoué, Soubré and the coastal forests. By contrast, Tahiri-Zagrët (1976) reported *C. sabaesus* from Tai, Duékoué and Soubré. Galat (1983), however, reported *C. sabaesus* from a site in the coastal area; a group on the edge of the Ebrié Lagoon, near Jacqueville (c. 40 km west of Abidjan), whereas Tahiri-Zagrët (1976) wrote that *C. sabaesus* does not occur in the coastal areas of southern Côte d'Ivoire. A possible reason for this contradictory information may be that local people call both Lowe's monkey *Cercopithecus lowei* and *C. sabaesus* "little black" or "little dark monkeys", and often mistake one for the other (Anh Galat-Luong and Gerard Galat, pers. comm.). Hence, it might be that the *C. sabaesus* reported from the forest area is indeed *Cercopithecus lowei*.

During our surveys (Gonedélé Bi *et al.* 2006; Gonedélé Bi 2008; Galat and Galat-Luong, in press; Gonedélé Bi *et al.* in press) we did not encounter *C. sabaesus* in any protected area in the forest zone where the species was reported by Tahiri-Zagrët (1976). This suggests that *C. sabaesus* has never occurred in these areas, has been locally extirpated, or that it is so rare that we did not find it. Our observations have confirmed the presence of *C. sabaesus* near the coast, namely in Ile Ehotilé National Park.

At this site (and also at the two other sites where *C. sabaesus* was reported by villagers) *C. sabaesus* seems to be restricted to swamp forests and mangroves. The use of mangroves by *C. sabaesus* has also been reported for populations in Senegal, Sierra Leone and Ghana (Galat and Galat-Luong 1976; Galat 1983; Grubb *et al.* 1998; Galat and Galat-Luong in

press). The current distribution of *C. sabaesus* in Côte d'Ivoire appears to be discontinuous: a northern savannah population and a coastal mangrove population with a distribution gap of about 300 km in the interjacent forest zone.

The disjunct distribution of *C. sabaesus* in Côte d'Ivoire is puzzling, and Galat and Galat-Luong (pers. comm.) suggest that those living in and near the mangrove swamps of the coast descended from introduced pets. The lagoon forests where *C. sabaesus* occurs are along the former north-south road to Abidjan and/or near important points of interest for tourists (exotic botanic garden and seaside resorts). These areas may have been used by foreigners to release pets before leaving the country. Due to the considerable adaptability and the ability of *C. sabaesus* to colonize mangrove swamps, the released monkeys would be expected to survive and reproduce in these areas (Galat and Galat-Luong 1976; Poirier 1972; McGuire 1974). Mangrove swamps in West Africa are becoming increasingly important refuges for large mammals as human populations increase (Galat-Luong and Galat 2007; Gonedélé Bi *et al.* 2008).

A second hypothesis sees the colonization of the coastal area by *C. sabaesus* as a consequence of relatively recent human-caused conversion of the rain forest into a forest agriculture mosaic. Savannah monkeys are known for invading cultivated forests (Kavanagh 1980) and since large parts of the forests in southern Côte d'Ivoire are already converted, there might now be a corridor for *C. sabaesus* to reach the coastal forests. If so, we would expect to also find *C. sabaesus* in areas between the northern savannah and the coastal region, wherever the forest has been converted to cultivation. There are, however, no reports of *C. sabaesus* from this region.

It might also be possible that the present populations of *C. sabaesus* in the littoral forests are relicts from a former continuous distribution of the species from the northern savannah belt to the coastal areas. There is some evidence that climatic fluctuations during the Pleistocene caused several retreats and expansions of rain forest in Côte d'Ivoire (Hewitt 2000). It is highly likely that savannah covered parts of southern Côte d'Ivoire during this period, thus connecting the northern savannah with the coast (Maley 1996; Ray and Adams 2001; Leal 2004). Under such conditions, it would have been possible for *C. sabaesus* to disperse from northern savannahs south to coastal areas, where they subsisted in mangrove habitats after the regrown rainforest isolated them from their northern conspecifics. A population genetic study, comparing northern and southern *C. sabaesus* is underway to test hypotheses about the origin of the southern population and about the time of their isolation.

Due to their wide distribution and large numbers, none of the five *Chlorocebus* spp. is regarded as threatened, with the exception of *C. djamdjamensis* (see IUCN 2008). In Côte d'Ivoire, however, even a generally common and adaptable species, such as *C. sabaesus*, has been apparently extirpated from large parts of its former range. This provides another example of the inadequate efforts to preserve primates in Côte d'Ivoire. The presence of *C. sabaesus* in the littoral forests of Côte d'Ivoire, on the other hand, demonstrates that more

survey work has to be done to document primate diversity and distribution in this region of the country. The information obtained from such surveys needs to be considered in the development of conservation measures for the region.

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Body Measurements for the Monkeys of Bioko Island, Equatorial Guinea

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Abstract: Bioko Island, Equatorial Guinea, has a rich (eight genera, 11 species), unique (seven endemic subspecies), and threatened (five species) primate fauna, but the taxonomic status of most forms is not clear. This uncertainty is a serious problem for the setting of priorities for the conservation of Bioko's (and the region's) primates. Some of the questions related to the taxonomic status of Bioko's primates can be resolved through the statistical comparison of data on their body measurements with those of their counterparts on the African mainland. Data for such comparisons are, however, lacking. This note presents the first large set of body measurement data for each of the seven species of monkeys endemic to Bioko; means, ranges, standard deviations and sample sizes for seven body measurements. These 49 data sets derive from 544 fresh adult specimens (235 adult males and 309 adult females) collected by shotgun hunters for sale in the bushmeat market in Malabo.

Key Words: Bioko Island, body measurements, conservation, monkeys, morphology, taxonomy

Introduction

Comparing external body measurements for adult individuals from different sites has long been used as a tool for describing populations, subspecies, and species of animals (see, for example, Eisentraut 1973; Dandelot 1974). Although most of Africa's primate taxa were first collected, described and named well over 100 years ago (Waterhouse 1838; Groves 2001, 2005; Grubb *et al.* 2003), identification was usually based on phenotypic characters (for example, color, pattern, texture, and length of the pelage) and measurements of the teeth and skull. While external body measurements from fresh specimens were sometimes available, the samples generally comprised but one or a few specimens. External body measurement data sets that are adequate for statistical analyses are still absent for many of the African primates. Indeed, for some species not even one full set of standard body measurement data from a fresh adult male specimen has been published (for example, golden-bellied mangabey *Cercocebus chrysogaster*, white-naped mangabey *Cercocebus lunulatus*, Sanje mangabey *Cercocebus sanjei*, kipunji *Rungwecebus kipunji*, southern talapoin monkey *Miopithecus talapoin*, dryad monkey *Cercopithecus dryas*, rolaway monkey *Cercopithecus rolaway*, djam-djam *Chlorocebus djamdjamensis*, and Udzungwa red colobus *Procolobus*

gordonorum), and surprisingly few such data exist even for some of the more widespread species (for example, Allen's swamp monkey *Allenopithecus nigroviridis*, northern talapoin monkey *Miopithecus ogouensis*, and grivet *Chlorocebus aethiops*).

Bioko Island (formerly Fernando Poo), Equatorial Guinea, is a continental island located about 32 km off the coast of Cameroon in the Gulf of Guinea (Fig. 1). Bioko (03°48'–03°12'N; 08°25'–08°57'E) has a surface area of about 2,017 km², an altitudinal range of 0–3,008 m, and a high mean annual rainfall that ranges from about 200 cm on the north coast to >1000 cm on the south coast. The primate fauna of Bioko is diverse, unique, and threatened (Basilio 1952; Eisentraut 1973; Oates 1988, 1996; Butynski and Koster 1994; Oates *et al.* 2004; Hearn *et al.* 2006). Based on all recent taxonomies (for example, Kingdon 1997; Groves 2001, 2005; Grubb *et al.* 2003), there are 11 species of primates on Bioko, five of which are threatened (Table 1; IUCN 2009). Of these, one is Critically Endangered, Pennant's red colobus *Procolobus pennantii* (Fig. 2), and two are Endangered, Preuss's monkey *Allochrocebus preussi* and drill *Mandrillus leucophaeus* (Fig. 3 and front cover of this issue of *Primate Conservation*). Nine of the 11 species of primate present on Bioko are usually regarded as represented either by subspecies endemic to Bioko (seven subspecies), or by subspecies

endemic to Bioko and to a small region on the immediately adjacent mainland (two subspecies; Eisentraut 1973; Oates 1988; Gautier-Hion *et al.* 1999; Groves 2001, 2007; Grubb *et al.* 2003; Oates *et al.* 2004). Of the nine subspecies of primates on Bioko, six are Endangered and three are Vulnerable (Table 1; IUCN 2009).

The primary threat to the monkeys of Bioko has long been, and remains, hunting with shotguns for the bushmeat trade (Fig. 4; Butynski and Koster 1994; Fa *et al.* 1995; Hearn *et al.* 2006). For its size, and based on current taxonomy (for example, Grubb *et al.* 2003), there is probably no single site in the world with more taxa of threatened primates than Bioko. In view of this situation, the IUCN/SSC Primate Specialist Group's action plans for African primates have consistently given high priority to the conservation of Bioko's primate fauna (Oates 1986, 1996; Lee *et al.* 1988).

One serious impediment to the conservation of Bioko's primates is the inadequate understanding of the taxonomic status of every one of the 11 primate taxa. It is safe to say that there is no community of primates in Africa for which there is more taxonomic confusion and uncertainty, or for which there is greater urgency for answers to taxonomic questions. For example, there is debate as to whether Pennant's red colobus and Allen's galago *Sciurocheirus alleni* of Bioko are endemic at the species (Groves 2001, 2005, 2007) or at the subspecies levels (cf. Dandelot 1974; Hill and Meester 1974; Napier 1985; Grubb *et al.* 2003). Similarly, it is far from clear as to whether the putty-nosed monkey *Cercopithecus nictitans*, crowned monkey *Cercopithecus pogonias*, and Demidoff's dwarf galago *Galagoides demidovii* on Bioko are endemic subspecies (Oates 1988; Gautier-Hion *et al.* 1999; Groves 2001). Until the many taxonomic questions surrounding the primate fauna of Bioko are resolved, it will remain difficult to set priorities for conservation, not just for the primate fauna of Bioko but also for the related primate fauna of western Central Africa (i.e., Nigeria, Cameroon, Equatorial Guinea, Gabon; Gautier-Hion *et al.* 1999; Oates *et al.* 2004).

That the taxonomic status of the primates on Bioko remains uncertain is partly due to the lack of significant samples of external body measurements both from Bioko and from mainland populations (Fig. 1). A review of the literature indicates that sets of external body measurement data for Bioko's primate taxa range from none (for example, Thomas's dwarf galago *Galagoides thomasi*) to six (for example, crowned monkey and red-eared monkey *Cercopithecus erythrotis*).

Here we present a new, large, set of seven body measurements for each of the seven species of monkeys present on Bioko.

Methods

From August 2006 into October 2007, we obtained five body and two tooth measurements from 1,039 monkeys in the Malabo ('Semu') Bushmeat Market. Malabo, the capital of Equatorial Guinea, is on the north coast of Bioko. More

than 90% of the approximately 200,000 people on Bioko live in Malabo and in nearby towns and villages. Recently killed (as well as 'smoked') monkeys obtained for the bushmeat trade by shotgun hunters are brought to the Malabo Bushmeat Market daily from all parts of Bioko. The body measurement data presented in this article come solely from monkeys brought to this market. No measurements were taken from smoked monkeys.

The vast majority of the immature monkeys were readily separated from the overall sample based on their small body size. Where there was a question as to whether a specimen

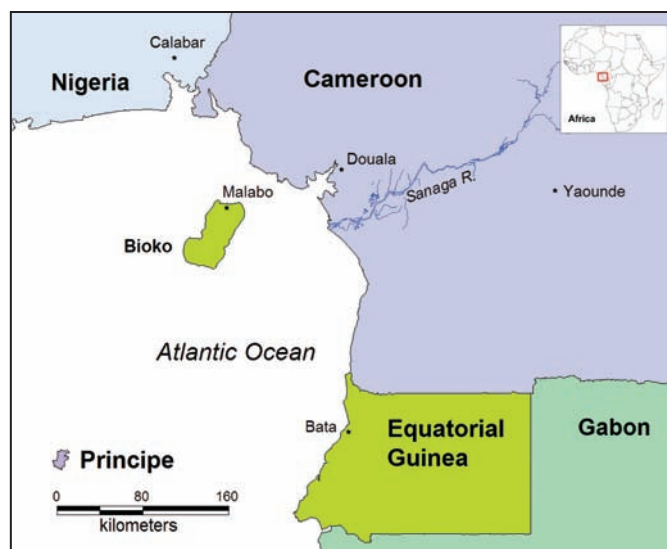


Figure 1. Location of Bioko Island (formerly Fernando Poo), Equatorial Guinea, in the Gulf of Guinea, western Central Africa.



Figure 2. Adult male Bioko red colobus *Procolobus pennantii pennantii*. This is a Critically Endangered species (IUCN 2009) and one of five threatened species of monkey on Bioko Island, Equatorial Guinea. This subspecies is limited to the southern-western corner (ca. 20%) of Bioko Island. Photograph © Tim Laman.

was immature or adult, the specimen was separated from, or included in, the ‘adult sample’ based on length of the canines, and on length of the nipples or width of the scrotum. Of the 1,039 monkeys measured, 544 were adults (235 adult males and 309 adult females).

Total body weight (mass) was recorded to the nearest 0.1 kg and the six linear measurements were recorded in millimeters (mm). The four linear body measurements were taken in the standard fashion (Martin *et al.* 2001):

Head-body length – From tip of nose to proximal base of tail (when tail is bent up at a right angle to the body).

Tail length – From the proximal base of tail (when tail is bent up at a right angle to the body) to the distal end of the last

tail vertebra (i.e., exclude protruding hairs). Tails which were incomplete were not measured.

Hindfoot length – From back edge of heel to tip of longest toe.

Ear length – From notch at base of inner ear to farthest point on edge of pinna. Damaged ears were not measured.

In addition to the above four measurements, *Upper canine length* and *Lower canine length* were recorded (from gum line to tip of canine). Broken canines were not measured.

Unfortunately, red colobus and black colobus *Colobus satanas* are usually eviscerated by hunters soon after being shot. For these two species the weights of the eviscerated



Figure 3. Juvenile Bioko drill *Mandrillus leucophaeus poensis*. This is an Endangered species (IUCN 2009) which is represented on Bioko Island, Equatorial Guinea by this endemic subspecies. This is the preferred prey of the commercial bushmeat hunters on Bioko. The total number of drills on Bioko as of 2009 is unlikely to be more than 4,000. Photograph by BBPP/Andrea Durcik. A photograph of an adult male Bioko drill appears on the front cover of this issue of *Primate Conservation*.



Figure 4. Bioko Preuss's monkey *Allochrocebus preussi insularis* for sale at the Malabo (‘Semu’) Bushmeat Market, Bioko Island, Equatorial Guinea. This is an Endangered species (IUCN 2009) which is represented on Bioko by an endemic subspecies. Note the short tail which is characteristic of this subspecies. All five of Bioko's threatened species of monkey are sold at this market. No fewer than 2,940 monkeys were sold here during the first 6 months of 2009. Photograph by Thomas M. Butynski.

Table 1. The primates of Bioko Island, Equatorial Guinea, and their degree of threat status at the species and subspecies levels (IUCN 2009). The taxonomy and vernacular names used here follow Grubb *et al.* (2003) except that we allocate Preuss's monkey to the genus *Allochrocebus*, not to the genus *Cercopithecus*.

Vernacular name	Scientific name	Degree of threat of species	Degree of threat of subspecies
Bioko black colobus*	<i>Colobus satanas satanas</i>	Vulnerable	Endangered
Bioko red colobus*	<i>Procolobus pennantii pennantii</i>	Critically Endangered	Endangered
Bioko drill*	<i>Mandrillus leucophaeus poensis</i>	Endangered	Endangered
Bioko Preuss's monkey*	<i>Allochrocebus preussi insularis</i>	Endangered	Endangered
Bioko red-eared monkey*	<i>Cercopithecus erythrotis erythrotis</i>	Vulnerable	Vulnerable
Golden-bellied crowned monkey	<i>Cercopithecus pogonias pogonias</i>	Least Concern	Vulnerable
Stampfli's putty-nosed monkey	<i>Cercopithecus nictitans martini</i>	Least Concern	Vulnerable
Bioko needle-clawed galago*	<i>Euoticus pallidus pallidus</i>	Least Concern	Endangered
Bioko Allen's galago*	<i>Sciurocheirus alleni alleni</i>	Least Concern	Endangered
Demidoff's dwarf galago	<i>Galagoides demidovii</i>	Least Concern	-
Thomas's dwarf galago	<i>Galagoides thomasi</i>	Least Concern	-

*Recognized by Grubb *et al.* (2003) as subspecies endemic to Bioko.

Table 2. Seven sets of body measurements from fresh adult male and adult female specimens for each of the seven species of monkeys on Bioko Island, Equatorial Guinea (August 2006–October 2007).

Species	<i>Colobus satanas satanas</i>							
Sex	Male				Female			
Measures	Mean	Range	SD	N	Mean	Range	SD	N
Head-body length (mm)	595.4	510–675	35.6	37	576.4	500–680	39.0	48
Tail length (mm)	759.0	690–840	39.5	37	741.8	600–825	54.4	47
Hindfoot length (mm)	174.4	160–188	7.1	38	169.7	154–190	8.1	46
Ear length (mm)	32.3	28–40	3.4	38	30.4	26–36	2.0	47
Upper canine length (mm)	14.7	10–18	2.0	28	6.4	4–10	2.0	43
Lower canine length (mm)	10.7	6–14	1.9	28	5.0	3–8	1.4	43
Weight (kg)	10.3	7.3–13.1	1.8	12	8.2	6.6–10.0	1.2	7
Weight when eviscerated (kg)	7.8	5.0–11.0	1.7	19	6.8	5.2–8.2	0.7	27

Species	<i>Procolobus pennantii pennantii</i>							
Sex	Male				Female			
Measures	Mean	Range	SD	N	Mean	Range	SD	N
Head-body length (mm)	504.7	470–554	23.0	12	518.8	470–583	24.0	48
Tail length (mm)	587.2	520–630	32.1	12	639.1	600–710	31.4	48
Hindfoot length (mm)	153.8	142–162	6.7	12	157.7	140–176	8.2	51
Ear length (mm)	29.8	26–35	2.9	12	29.8	26–33	1.7	50
Upper canine length (mm)	17.1	15–20	1.9	12	6.8	4–12	2.2	41
Lower canine length (mm)	11.8	10–16	2.0	12	4.7	3–8	1.3	40
Weight (kg)	11.0	–	–	1	10.0	–	–	1
Weight when eviscerated (kg)	6.2	5.2–8.0	0.9	9	5.8	5.0–7.0	0.6	38

Species	<i>Mandrillus leucophaeus poensis</i>							
Sex	Male				Female			
Measures	Mean	Range	SD	N	Mean	Range	SD	N
Head-body length (mm)	668.1	600–740	41.8	28	537.7	470–620	40.7	41
Tail length (mm)	84.7	65–100	10.4	31	63.3	40–90	9.3	40
Hindfoot length (mm)	181.8	160–200	11.5	30	149.1	140–165	7.7	38
Ear length (mm)	37.2	30–44	3.6	30	31.7	28–41	3.1	41
Upper canine length (mm)	35.7	28–40	4.6	26	7.3	4–16	3.1	30
Lower canine length (mm)	21.8	18–28	2.5	30	5.7	3–11	2.3	30
Weight (kg)	20.0	14.5–27.0	3.5	26	8.5	6.5–12.0	1.5	39

Species	<i>Allochrocebus preussi insularis</i>							
Sex	Male				Female			
Measures	Mean	Range	SD	N	Mean	Range	SD	N
Head-body length (mm)	483.9	420–540	32.0	23	402.8	370–440	19.1	40
Tail length (mm)	552.7	500–600	29.4	23	476.6	410–560	41.2	42
Hindfoot length (mm)	140.1	130–150	5.5	24	123.1	110–145	8.8	42
Ear length (mm)	31.0	28–35	2.4	23	29.0	24–35	2.3	43
Upper canine length (mm)	17.3	13–22	2.4	23	10.7	6–16	2.6	39
Lower canine length (mm)	11.8	10–14	1.4	24	7.0	3–12	2.5	41
Weight (kg)	5.5	4.7–6.5	0.5	24	3.5	2.9–4.7	0.5	42

Table 2. *continued*

Species	<i>Cercopithecus erythrotis erythrotis</i>							
Sex	Male				Female			
Measures	Mean	Range	SD	N	Mean	Range	SD	N
Head-body length (mm)	420.1	390–500	23.2	66	384.1	360–430	15.9	64
Tail length (mm)	608.7	490–730	50.7	67	552.7	460–660	41.2	62
Hindfoot length (mm)	124.0	112–144	7.8	67	113.5	105–130	6.0	64
Ear length (mm)	27.6	24–31	1.9	67	27.2	22–32	2.0	64
Upper canine length (mm)	14.0	12–18	1.6	65	10.4	6–14	2.6	59
Lower canine length (mm)	9.1	7–12	1.1	64	7.0	5–11	1.7	59
Weight (kg)	3.7	3.0–5.6	0.6	64	3.1	2.4–5.4	0.5	64

Species	<i>Cercopithecus pogonias pogonias</i>							
Sex	Male				Female			
Measures	Mean	Range	SD	N	Mean	Range	SD	N
Head-body length (mm)	407.0	370–480	19.3	46	372.4	340–410	18.4	40
Tail length (mm)	624.0	560–730	36.1	47	556.6	480–610	33.7	42
Hindfoot length (mm)	123.8	118–140	6.5	47	114.2	110–120	4.3	41
Ear length (mm)	26.9	22–30	1.9	47	26.5	22–30	1.6	41
Upper canine length (mm)	14.2	10–20	2.1	48	10.2	7–14	2.2	32
Lower canine length (mm)	9.6	6–13	1.7	48	7.3	4–12	2.6	33
Weight (kg)	3.7	3.0–5.1	0.6	45	2.8	2.2–3.8	0.4	39

Species	<i>Cercopithecus nictitans martini</i>							
Sex	Male				Female			
Measures	Mean	Range	SD	N	Mean	Range	SD	N
Head-body length (mm)	484.6	420–570	44.9	14	439.1	400–500	28.0	20
Tail length (mm)	739.8	700–790	26.3	13	647.9	558–700	46.3	19
Hindfoot length (mm)	138.9	130–150	7.3	14	125.1	112–132	6.1	18
Ear length (mm)	30.2	28–35	2.3	15	28.6	26–32	1.6	20
Upper canine length (mm)	15.7	12–20	2.8	13	9.4	6–12	1.8	16
Lower canine length (mm)	10.7	10–12	1.0	13	6.5	4–10	2.2	18
Weight (kg)	5.1	4.0–6.0	0.6	14	4.1	3.0–5.6	0.7	20

individuals were recorded and are presented here in addition to a smaller number of weights from intact specimens.

Results and Discussion

The means, ranges, standard deviations, and sample sizes for these seven body measurements are presented in Table 2. Through the presentation of these data we hope to encourage and facilitate further research on the systematics and diversity of the monkeys of Bioko, and of their counterparts on the mainland. These data should also provide insights into the extent to which insular primate taxa undergo size changes (the 'Island Rule' or 'Foster's Rule') and changes in degree of sexual size dimorphism (Isaac 2005; Bromham and Cardillo 2007). As such, we plan to apply these data in a series of papers that will examine the validity of the current taxonomy for the monkeys of Bioko and of the possible effects of insularity on this primate fauna. In this planned series of papers, the body measurement data presented here will be supported (1) by a thorough review of the literature for each taxon, (2) by our detailed descriptions of the pelage pattern and coloration of the primates of Bioko, (3) by reference to the hundreds of close-up photographs that we have taken of the primates of Bioko, and (4) by our observations on the ecology and behavior of free-living primates on Bioko.

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Primate Crop-raiding: A Study of Local Perceptions in Four Villages in North Sumatra, Indonesia

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Abstract: The main threat to the survival of primates in Sumatra is habitat destruction, but there is also an increasing problem of conflict with local people due to crop-raiding. This study characterizes the perceived impacts of primate crop-raiding in four villages in North Sumatra. Ninety-eight semi-structured interviews were conducted to collect data on (i) crop species, (ii) which vertebrates are thought to damage crops, (iii) the perceived extent to which each vertebrate species damages crops, and (iv) the preventive measures taken in the four villages. Farmers reported an average of 16 different crop species; 85.7% had rubber plantations. Crop-raiding by wildlife was reported by 94.9% of the interviewees as the single most important determinant of crop yields. Thirteen vertebrates were reported causing damage to cultivars; most important were squirrels, porcupines, pigs, deer, elephants and primates. However, primates were perceived as damaging crops differently from the other vertebrate species. Long-tailed macaques (*Macaca fascicularis*) and Thomas' leaf monkeys (*Presbytis thomasi*) were considered to be the most destructive crop-raiders in all locations. Contrary to what was expected, only a small proportion of farmers complained about the Sumatran orang-utan (*Pongo abelii*). The interviewees reported twenty different crop protection techniques. Shouting was the most common.

Key words: Primates, crop-raiding, Sumatra, human-wildlife conflict

Introduction

The Sumatran landscape is currently dominated by agriculture with isolated patches of rainforest (Kinnaird *et al.* 2003). Oil palm is grown as a monoculture, and provides habitat for 20% or less of the previously resident animals (Laidlaw 1995; Heang and Boo Liat 1998). The remaining forest patches cannot support all the previously resident animals. As a result, primates, for example, are pushed nearer to human settlements and raid crops in farms (Brown and Jacobson 2005).

The province of North Sumatra used to be Indonesia's primary rubber-producing area, but in the 1990s most of the estates were converted to palm oil plantations (Gérard and Ruf 2001). The area covered by oil palm plantations in Indonesia is 3,107,986 ha, and the Indonesian government is planning to expand oil palm plantations by an additional 4 million ha in Sumatra (Brown and Jacobson 2005). Large-scale plantation activities are one of the main causes of the ongoing degradation of primate habitat and food resources, which in turn increases crop-raiding in smaller-scale agricultural plots of local farmers. Indonesian householders plant various

species of trees, including durian (*Durio zibethinus*), cempe-dak (*Artocarpus integer*), jengkol (*Archidendron pauciflorum*), petai (*Parkia speciosa*) and mango (*Mangifera* spp.), as commercial and household crops along the perimeter of their cash crop plantations (rubber and oil palm). Primates eat these fruits, resulting in conflict between wildlife and farmers.

In tropical and subtropical regions, the extension of farming into the forest interior makes wild animals become farm pests, and the degree of tolerance of the damage caused changes over time (Knight 2001). Indeed, many farmers expect a certain amount of loss due to wildlife damage, but tolerance of this declines with the increasing use of pesticides and new technological inputs which promise preventability (Knight 2001). Previous research into human-wildlife conflict has tended to focus on either directly measuring the extent of the losses at sites where damage is occurring or interviewing the victims of the damage about their losses (Conover 2002). It is important to record absolute crop losses experienced at the individual, village and district levels because individual farming households can experience different crop losses even within the same village (Hill 2004). For instance, the distance between the farm and the forest boundaries and the number

of neighboring farms are highly likely to affect vulnerability to crop-raiding by wildlife (Hill 2000). Additionally, farmers are not equally exposed to human-wildlife conflict situations; age, gender, farm location, ethnicity, cultural rules, crop assemblages, and the behavioral and ecological characteristics of wildlife can all influence vulnerability to crop damage by wildlife (Hill 2004). The extent and intensity of damage may also vary depending on the cropping patterns, wildlife population density and behavior, and food availability in wild habitats (Sekhar 1998). Crop losses to wildlife may have various impacts on farming households. They include high guarding investment, disruption of schooling for children who have to help guard fields, increased risk of injury from wildlife, and increased risk of contracting diseases such as malaria (Hill 2004). Crop-damage depends also on the species that are involved in this activity. Indeed, different species may specialize on different types of crop and different plant parts or development stages (Osborn and Hill 2005). Certain species may cause more damage than others. For instance, primates and elephants can have a significant impact on crop yields due to the extreme agility of many primate species and to the large size and nocturnal/crepuscular activity of elephants (Osborn and Hill 2005). Our knowledge and understanding of the issues surrounding human-primate conflict interactions is based predominantly on research completed at a number of African sites in, for example, Tanzania (Gillingham and Lee 2003), Uganda (Hill 2000, 2004; Naughton-Treves *et al.* 1998; Saj 2001), and Zanzibar (Siex and Struhsaker 1999). In Asia, crop depredation by wild animals has been studied in India (for example, Sekhar 1998; Rao *et al.* 2002; Chhangani and Mohnot 2004), in Indonesia (for example, Nyhus *et al.* 2003; Priston 2005), and in Nepal (for example,

Chalise 2005). In India, crop damage is very common along the immediate periphery of wildlife sanctuaries and national parks (Chhangani and Mohnot 2004), as is the case at many other sites in Africa and Asia.

To date, comparatively little information has been published about crop raiding by primates in Indonesia, and information is largely limited to unpublished reports and anecdotes. In addition, the literature focuses on orangutans (*Pongo abelii*) and very little is known about conflicts involving other non-human primates. When their habitat is lost, orangutans stay within the area even to the extent of risking starvation (Singleton *et al.* 2002). Under such circumstances, this renders them even more likely to raid householders' gardens. Householders are reported to respond to crop-raiding by shooting the apes (I. Singleton pers. comm.). However, while conflicts between farmers and orangutans are being reported, there are no systematic records and no centralized database, so little verified information is available in order to examine exactly what is happening. This study provides baseline data on human-primate conflicts in areas where non-human primates, including orangutans, and farmers co-exist.

Methods

Study sites

The study took place in four locations in the Langkat district of Sumatra: Sampan Getek, Tangkahan, Bukit Lawang, and Telaga Said (see Fig. 1). The village of Sampan Getek is located at 3°43'35"N, 98°11'26"E. Singleton *et al.* (2002) identified eight orangutans in this area, which appears to be completely isolated from the forests of the Leuser Ecosystem. The area in which these animals roam is less than 1,000 ha



Figure 1. Location of the study sites in North Sumatra. Map by Ian Singleton©.

and is surrounded by mature palm oil plantations. It is thought that the orangutans survive by ‘hiding’ in the more scrubby valleys, eating what fruits are available there and avoiding encounters with humans as much as they can (Singleton *et al.* 2002). Tangkahan is located in Padang Tualang sub-district, at 3°41'9"N, 98°4'31"E. The village is on the border of the Gunung Leuser National Park, but is otherwise surrounded by palm oil plantations. The village of Bukit Lawang is located in Bohorok sub-district, at 3°32'58"N, 98°7'33"E. Until 1995, Bukit Lawang was the site of an orangutan rehabilitation center that released ex-captive orangutans into the Gunung Leuser National Park. The center was closed in 1995, due mainly to new regulations governing species' reintroduction that disallowed the release of the orangutans into areas already containing existing wild populations (Orangutan Reintroduction and Protection Workshop 2001; IUCN/SSC Re-introduction Specialist Group 1995). In addition, the site has become a mass-tourism destination that is not compatible with a re-introduction programme. Nevertheless, semi-wild orangutans still live in the area surrounding the center and are fed on platforms twice a day in order to give tourists an opportunity to view them (Rijksen and Meijaard 1999). In this village householders consider that orangutan raids on gardens are particularly problematic. The village of Telaga Said is located at 3°37'47"N, 98°16'45"E; an area of about 1,000 ha surrounded by palm oil plantations and completely isolated from the forests of the Gunung Leuser National Park. Orangutan habitat in this location is restricted to about 10 ha, bordered by a river on one side and by palm oil plantations on the other. Conflicts between orangutans and farmers are reported in this village, especially during the durian fruiting season.

Data collection

Semi-structured interviews were used to investigate the farmers' perceptions of the crop-raiding issue. The framework for interviews was adapted from Gillingham and Lee (2003)

(see Table 1). The interviews were conducted in Bahasa Indonesia with the help of a local translator. These semi-structured interviews were conducted with either the household head or the household head's wife. The local population was advised of the aims of the study, how the results would be used and who was funding it, following the Oxford Brookes University Code of Practice on Ethical Standards. Each interviewee was informed that sensitive information and personal characteristics would not be included in any reports or publications, against his/her wishes (Christensen 1992). Each interview took approximately 20–30 minutes to complete, and focused on farming strategies and experiences of crop-damage by vertebrates (following Hill 1997). Notes were taken during the interview, including information about the informant's manner, number of interruptions, and physical surroundings.

Study sample

Ninety-eight interviews were completed between 17 May and 24 June 2005. In each site, we conducted an average of eight interviews a day. We reviewed the daily interviews every evening. The reported age of respondents ranged from 18 to 113 years, with a mean of 41.9. Two thirds of the respondents were men. The majority originated from North Sumatran ethnic groups (62% of respondents were Karo and 6% were Melayu people), others were ethnically from the islands of Java (29%) or Borneo (3%). Sixty-seven percent identified themselves as Christian and 33% as Muslim. The majority of respondents (61%) reported that they had lived in the study site for more than 25 years (24% between 26 and 35 years and 37% for longer than 36 years).

Data analysis

The data collected from the interviews are presented as the percentage frequency of respondents giving each response in the case of multi-response questions. SPSS for Windows (version 11) was used to analyse all data. Weighted ranks

Table 1. Framework of the semi-structured interview.

The number of the interview, date, location (name of the village), time at which the interview was completed and the sex of the interviewee were first recorded. The interviewee was then asked the following questions:	
1.	How old are you?
2.	What is your ethnicity?
3.	What is your religion?
4.	How long have you been in this village?
5.	What is your position in the household?
6.	How far is your garden from your home?
7.	Which types of crops do you cultivate in your garden?
8.	Do you sell your harvest, or is it solely for household consumption?
9.	Does anything limit your crop yields?
	• Respondents who answered “yes” to this question were then asked: “Which of kind of problems limit crop yields in your gardens?” Respondents were asked to rank the problems in order of importance.
	• If respondents listed wild animals as one of the problems, they were asked “Which animals are problematic for you?” They were asked to rank the four species that cause the most damage in order of importance.
	• Respondents who reported primates as causing crop-damage were then asked: “When was the last time that a primate caused crop-damage to your garden?” and “Which species of primate did you see crop raiding in your garden?”
10.	Which methods do you use to protect your crops?
11.	Have you ever seen/heard someone shooting/trapping a crop raider?
	• Respondents who answered “yes” to this question were then asked: “Did this person kill the animal or injure it?”

were calculated from the ranks farmers assigned to different wildlife species to indicate how species compared with each other with respect to the crop damage they caused. To calculate the weighted rank, each individual rank given by the interviewees was assigned a score: rank 1=1, rank 2=2, rank 3=3, and rank 4=4, and the formula below was used to calculate weighted rank for each species.

$$\text{Overall ranking} = \Sigma (\text{score} \times n) / N$$

n = number of respondents ranking the species
N = total number of respondents in the sample

The Kendall Coefficient of Concordance was used to assess degree of concordance in farmer rankings across the four village sites (Siegel and Castellan 1988).

Results

Only 21.4% of respondents had two gardens and 3.1% had three. Most smallholders (85.7%) reported they cultivated rubber; all the farmers interviewed in Telaga Said grew this crop (Table 2). Interviewees reported planting various species of trees at the margin of their cash crop plantations (rubber and oil palm). Durian trees were grown by the majority of interviewees (62.3%). In Tangkahan, the large majority of the farmers interviewed (85.7%) were growing this crop. Approximately half the interviewees cultivated cempedak, jengkol and petai (55.1%, 51.2% and 49.2%, respectively). Of the 30.6% of interviewees who reported they cultivated oil palm, the largest proportion (48.6%) was found in Tangkahan.

The number of different crops grown on each farm varied greatly (mean standard variation=30.73; n=98). The majority grown were sold in local markets, where people bought

food mainly for their households (see Table 3). All the villagers interviewed were fully dependent on the proceeds of their gardens for their livelihood needs. The average area of rubber and oil palm fields was about 2 ha (1.9 ha and 2.2 ha, respectively; see Table 3). The annual incomes generated from the main cash crops (durian, rubber and oil palm) were very variable, depending on the quality of the production and the price at the local market. Nevertheless, the mean income generated by rubber plantations was reported to be about two times higher than the mean income from an oil palm plantation.

Most farmers (94.9%) reported that crop-raiding by wildlife was the single most important limit to their yields. Other reported constraints on agricultural productivity were insect pests and fungi. There is a small degree of variation in the farmers' ranking of crop raiding species across the 4 village sites but analysis using the Kendall Coefficient of Concordance indicates that there is a significant degree of concordance between farmer rankings of problem species across the four villages ($W=0.556$, $N=8$, $p<0.05$). Locally, primates were considered more problematic than any other wildlife species; long-tailed macaques ranked first as the most severe crop-raiding species in terms of raid frequency and economic impact, and the next four were Thomas' leaf monkey, pig-tailed macaque, Griffith's silver langur, and orangutan, respectively (see Table 4). Thomas' leaf monkeys were reported damaging fruits of 11 crop species (see Table 5). They were also reported eating the flowers of durian and petai trees (53.1% and 37% responses, respectively). Long-tailed macaques were reported to cause damage to 14 crop species (Table 6). Even though they damaged fruits crops the most often, they also damaged flowers of petai, durian and banana trees. Sumatran orangutans were reported causing damage to only six species of fruit tree (Table 6); only a small proportion of respondents claimed that they damage banana, jengkol and mango trees.

Squirrels were indicated as the most destructive to the oil palm plantations, with 76% of oil palm growers reporting

Table 2. Distribution of the most common crop species in each study site (percentage of interviewees).

Crop species	Sampan Getek (n=39)	Tangkahan (n=35)	Bukit Lawang (n=10)	Telaga Said (n=14)	Mean (N=98)
Rubber (<i>Hevea brasiliensis</i>)	84.6	80.1	90	100	85.7
Durian (<i>Durio zibethinus</i>)	48.8	85.7	60	42.9	62.3
Cempedak (<i>Artocarpus integer</i>)	66.7	34.4	50	85.8	55.1
Jengkol (<i>Archidendron pauciflorum</i>)	28.3	59.9	60	86.2	51.2
Petai (<i>Parkia speciosa</i>)	48.9	43.1	60	57.8	49.2
Mango (<i>Mangifera</i> spp.)	15.4	51.6	40	36.4	33.8
Oil palm (<i>Elaeis guineensis</i>)	23.1	48.6	10	21.4	30.6
Banana (<i>Musa</i> spp.)	12.8	34.3	30	36.1	25.6
Rambutan (<i>Nephelium lappaceum</i>)	10.2	34.3	10	64.1	26.5
Cocoa (<i>Theobroma cacao</i>)	10.2	31.5	10	0	16.3
Mangosteen (<i>Garcinia mangostana</i>)	5.2	2.9	20	14.4	11.3
Pineapple (<i>Ananas</i> spp.)	5.2	14.4	20	0	5.1
Maize (<i>Zea mays</i>)	2.5	5.6	0	0	3
Papaya (<i>Carica papaya</i>)	0	5.6	0	7.7	3

Table 3. Area, number of trees grown and destination of the crops per species cultivated. The values in parentheses are the standard deviation.

	Mean area (ha)	Mean number of trees	Destination of crops (% responses; N = 98)		
			Household consumption	Sale	Both
Banana	N/A	18.4 (10.3)	12	60	28
Cempedak	N/A	19.5 (10.6)	16.4	78.2	5.5
Cocoa	N/A	128 (90.8)	0	100	0
Corn	0.06 (0.04)	N/A	0	100	0
Durian	N/A	15.2 (8.6)	7.8	89.1	3.1
Jengköl	N/A	10.1 (5.6)	5.9	84.3	9.8
Mango	N/A	4.3 (3.5)	41.7	41.7	16.7
Mangosteen	N/A	3.9 (2.6)	18.2	81.8	0
Oil palm	2.2 (0.8)	116.7 (63.1)	0	100	0
Papaya	N/A	6.3 (3.2)	33.3	33.3	33.3
Petai	N/A	5.9 (3.1)	10.9	89.1	0
Pineapple	N/A	131 (49.9)	0	100	0
Rambutan	N/A	6.5 (3.6)	19.2	65.4	15.4
Rubber	1.9 (1.1)	237.1 (144.9)	0	100	0

Table 4. Wildlife species reported to damage crops at each of the four villages. In **bold** are the percentages of farmers reporting crop-raiding of each species (per study site and the mean for the four study sites). The values in parentheses are the rankings of the farmers for each species reported to damage crops: the first value is the farmer's overall ranking of the crop raiders; numbers after the semi-colon are the weighted ranking scores of the various species. N/A = Not assessed.

Species	Villages in which farmers lived								Mean (n = 98)	
	Sampan Getek (n = 39)		Tangkahan (n = 35)		Bukit Lawang (n = 10)		Telaga Said (n = 14)			
Civet (<i>Viverricula</i> spp.)	5%		18.5%		20%		0%		10.6%	
Common long-tailed macaque (<i>Macaca fascicularis</i>)	85%	(1.64 ; 1)	91%	(1.31 ; 1)	90%	(1.43 ; 1)	92%	(1.27 ; 1)	88.6%	(1.46; 1)
Fruit bat (Pteropodidae)	1.5%		4.5%		10%		8%		4.4%	
Griffith's silver langur (<i>Trachypithecus villosus villosus</i>)*	12%	(2.60 ; 3)	9%	(3.33 ; 6)	20%	(3.50 ; 6)	92%	(2.60 ; 5)	23.2%	(2.65 ; 4)
Pig (<i>Sus scrofa</i>)	26%	(3.44 ; 7)	48%	(4.00 ; 8)	0%	(N/A ; 7)	15%	(2.00 ; 2)	29.6%	(3.25 ; 6)
Pig-tailed macaque (<i>Macaca nemestrina</i>)	1.5%	(N/A ; 8)	42%	(2.61 ; 3)	80%	(2.67 ; 4)	42%	(2.39 ; 4)	29.8%	(2.48 ; 3)
Plantain squirrel (<i>Callosciurus notatus</i>)	47%	(2.81 ; 4)	78%	(2.72; 4)	70%	(2.67; 4)	100%	(3.77; 6)	68.0%	(3.77; 7)
Porcupine (<i>Hystrix brachyura</i>)	12%	(3.20 ; 6)	18%	(3.20 ; 5)	0%	(N/A ; 7)	0%	(N/A ; 8)	11.2%	(N/A ; 8)
Sumatran elephant (<i>Elephas maximus sumatrensis</i>)	0		15%		0%		0%		5.3%	
Sumatran orang-utan (<i>Pongo abelii</i>)	58%	(2.85; 5)	49%	(3.60; 7)	80%	(2.12; 2)	21%	(4.00; 7)	51.7%	(2.95; 5)
Sun bear (<i>Helarctos malayanus</i>)	0%		11%		10%		0%		4.9%	
Thomas' leaf monkey (<i>Presbytis thomasi</i>)	81%	(1.78; 2)	85%	(2.17; 2)	90%	(2.50; 3)	92%	(2.27; 3)	84.9%	(2.06; 2)

*sensu Brandon-Jones et al. (2004)

Table 5. Damage caused by the Thomas' leaf monkey and the Griffith's silver langur to each fruit tree species.

	Thomas' leaf monkey				Griffith's silver langur			
	No damage	Fruits	Fruits+ flowers	Fruits+ flowers+ leaves	No damage	Fruits	Fruits+ flowers	Fruits+ flowers+ leaves
Banana (N=25)	20	72	4	4	72	28	0	0
Cempedak (N=55)	27.3	69.1	0	3.6	78.2	21.8	0	0
Cocoa (N=17)	88.2	11.8	0	0	100	0	0	0
Corn (N=3)	66.7	33.3	0	0	66.7	33.3	0	0
Durian (N=64)	32.8	7.8	53.1	6.3	90.6	1.6	7.8	0
Jengköl (N=51)	37.3	54.9	2	5.9	78.4	19.6	0	2
Mango (N=36)	47.2	44.4	2.8	5.6	88.9	8.3	0	2.8
Mangosteen (N=11)	72.7	27.3	0	0	90.9	9.1	0	0
Papaya (N=3)	66.7	33.3	0	0	66.7	33.3	0	0
Petai (N=46)	30.4	26.1	37	6.5	78.3	13	8.7	0
Pineapple (N=5)	100	0	0	0	100	0	0	0
Rambutan (N=26)	38.5	57.7	0	3.8	76.9	23.1	0	0

that they attacked the fruits. Almost 45% of crop growers reported that long-tailed macaques raided the fruits. Pigs, porcupines and elephants were reported damaging the roots of the oil palm trees. About 64% of rubber growers claimed that Thomas' leaf monkeys damaged the leaves. According to 8.3% of the growers, orangutans were responsible for damage to the bark of the rubber trees.

Most tree crops are reported to be most vulnerable to damage by wildlife during their annual fruiting season (for example, durian, mango, cempedak; see Table 7). However, some crops (rubber, oil palm, banana, cocoa, coconut, papaya, pineapple) were reportedly damaged throughout the year. All farmers said that primates and squirrels were diurnal crop raiders. The other vertebrates (pigs, porcupines and elephants) were nocturnal crop raiders.

Seventeen crop protection techniques were reported being used to respond to crop-raiding by wildlife (see Table 8). About half of the farmers interviewed reported that they shouted to prevent crop raiding. Shouting was the most commonly used protection in Sampan Getek, Tangkahan and Bukit Lawang (respectively 41%, 66% and 50%). In Telaga Said, the percentage of farmers using guns to injure or kill wildlife was higher than in the other villages (57.1% in Telaga Said, 2.6% in Sampan Getek, 8.6% in Tangkahan and 10% in Bukit Lawang).

Thirty-nine percent of interviewees claimed that they had never done any harm to a primate, and that they had never seen others doing so either (Table 9). More interviewees reported the occurrence of farmers killing primates in Telaga Said than in other locations, and they were mostly long-tailed macaques.

Discussion

Crop losses due to excess rainfall or lack of fertilizer and pesticide, though a common claim in developing countries (Rao *et al.* 2002), were not mentioned by the farmers interviewed. The survey data presented here revealed that villagers consider wildlife crop-damage as the most significant limitation to their agricultural production. Some fruit trees

were reported damaged every month (for example, banana and papaya trees) because they have no definite fruiting season (Soemarwoto *et al.* 1985). However, most cultivated fruit tree species (cempedak, jengkol, durian and petai trees) were reported to be most vulnerable during the peak of fruiting (May to August). As a result, it is highly likely that conflicts were exacerbated during those four months. Data collection occurred during this time, and the probability that the incidence of crop-raiding was just then particularly high may have influenced the perceptions and responses of the interviewees. Damage to the two main cash crops (oil palm and rubber) was reported to be a problem year round; a situation which makes the control of damage by wildlife on those two species even more difficult.

Fruits were the plant parts most often reported damaged by primates. However, although primates caused damage to the fruits and leaves of rubber trees, farmers did not perceive this as a problem because it does not affect latex production, which is all they exploit. Squirrels raid fruits of all the crop species, but according to the interviewees, squirrels are less of a problem because they eat much less than the primates. A similar situation was reported from Uganda by Saj *et al.* (2001). Another important aspect that may influence people's perceptions is the size of the animal. Bell (1984), for example, found that larger animals attracted greater attention from the farmers because farmers assume that small animals eat fewer fruits and therefore cause less damage than do the larger species. Small species, however, usually have a larger population (for example, insect pests) and consequently can cause significant damage when they raid gardens.

Damage to oil palm fruits was mainly attributed to squirrels and long-tailed macaques. Only a small proportion of oil palm growers complained about the other primate species; a situation which is surprising when compared with the oil palm raiding issue in Malaysia. Indeed, orangutan raids in oil palm plantations are commonplace in the lower Kinabatangan, Malaysia (Brown and Jacobson 2005). Nevertheless, it seems that orangutans do not like oil palm fruits and they raid plantations only when there is little alternative (M. Ancrenaz pers. comm. 2005). In Sabah, there are a number of places where

Table 6. Damage caused by the long-tailed macaque, the pig-tailed macaque and the Sumatran orangutan to each fruit tree species.

Crop species	Long-tailed macaque			Pig-tailed macaque			Sumatran orangutan				
	No damage	Fruits	Fruits+ flowers	No damage	Fruits	Fruits+ flowers	No damage	Fruits	Bark	Fruits+ bark	Nest
Banana (n = 25)	28	64	8	84	16	0	92	8	0	0	0
Cempedak (n = 55)	25.5	74.5	0	85.5	14.5	0	67.3	16.4	7.3	9.1	0
Cocoa (n = 17)	52.9	47.1	0	94.1	5.9	0	100	0	0	0	0
Corn (n = 3)	0	100	0	66.7	33.3	0	100	0	0	0	0
Durian (n = 64)	34.4	53.1	9.4	81.3	17.2	1.6	39.1	34.4	0	10.9	15.6
Jengkol (n = 51)	21.6	78.4	0	78.4	21.6	0	84.3	9.8	2	3.9	0
Mango (n = 36)	22.2	77.8	0	83.3	16.7	0	97.2	2.8	0	0	0
Mangosteen (n = 11)	27.3	72.7	0	72.7	27.3	0	100	0	0	0	0
Papaya (n = 3)	33.3	66.7	0	0	100	0	100	0	0	0	0
Petai (n = 46)	32.6	52.2	15.2	80.4	17.4	2.2	69.6	30.4	0	0	0
Pineapple (n = 5)	40	60	0	80	20	0	100	0	0	0	0

small forest patches have been surrounded by oil palms and then further encroached upon. Any orangutans remaining in these isolated patches are often crowded together and forced to eat oil palms to survive. In Sumatra to date, there are few or no reports of orangutans eating palm oil, although that is not to say that they never do, nor that it will not happen in the future as more and more of their forest is lost (I. Singleton pers. comm. 2005). Damage to the bark caused by pigs, porcupines and elephants does not have a direct impact on the oil palm fruit harvests, but if severe it may kill the palm. These three nocturnal species (pig, porcupine, elephant) should also, therefore, be considered as of some threat to oil palms.

Table 7. Months when crops are the most vulnerable.

Crop species	Month
Banana	Every month (depending on when the tree was planted)
Cempedak	May–August
Cocoa	Every month
Coconut	Every month
Corn	Depends on when the tree was planted
Durian	July–August
Jengköl	February, June–July
Kemiri	February
Lemon	March, May, September
Mango	April–May
Mangosteen	July–August
Oil palm	Every month
Orange	March, June, December
Papaya	Every month
Petai	February, June–July
Pineapple	Every month
Rambutan	August, October
Rubber	Every month

The long-tailed macaque was considered to be the most destructive crop-raiding species in all the four villages. It was reported raiding fruits of all crop species grown. The reasons why it is such a successful pest are probably its omnivorous diet and its behavioral adaptability to changing habitat (Aldrich-Blake 1980; Altmann and Muruthi 1988).

Thomas' leaf monkey was most often perceived as the second most destructive crop-raiding species. It was reported to damage fruits at all stages of maturation, and also flowers and leaves, and is therefore believed to damage crops year round. A small number of farmers complained about the Sumatran orangutan, citing especially damage to cempedak, durian and petai yields. Contrary to what we expected, less than half of the durian growers reported damage to their crops from orangutans. They did complain, however, that the damage was always considerable because of the orangutan's large size.

Active deterrence methods to guard crops against diurnal crop-raiding species were used by a majority of the farmers. Guarding oil palm and rubber plantations by night would perhaps contribute to the reduction in the damage caused by pigs, porcupines, deer and elephants. However, guarding fields results in increased risks of being injured by wild animals or contracting malaria (Hill 2004), and is very time consuming. Alternatively farmers can cooperate in a system of rotating "guard duty" (Osborn and Parker 2003), which would help reduce costs to individuals. In the longer term, conserving large blocks of forest and reducing forest-edge habitat should be a management priority (Naughton-Treves *et al.* 1998). Unfortunately, current trends in encroachment and fragmentation tend to make forest boundaries much longer than would be preferable. In Tangkahan, where some farmers own fields along the edge of the Gunung Leuser National Park, the

Table 8. Percentage of farmers reporting which method they use to prevent crop damage.

Animal species targeted by the protection method	Protection methods	Sampan Getek (n=39)	Tangkahan (n=35)	Bukit Lawang (n=10)	Telaga Said (n=14)	Mean (n=98)
	<i>Passive deterrence methods</i>					
Primate	▪ Using scarecrow	7.7	2.9	0	14.2	5.6
Primate	▪ Making fires	7.7	5.7	0	0	5.1
Orangutan	▪ Putting metal around trees	2.6	5.7	20	0	5.1
Primate; squirrel	▪ Suspending cans	2.6	5.7	0	0	3.1
Primate; civet cat	▪ Hiding fruits	5.1	2.9	0	0	3
Porcupine	▪ Spreading tar on the bark	2.6	2.9	0	0	2.1
Porcupine	▪ Spreading sulphur around trees	0	0	0	7.1	1
Pig; deer; elephant	▪ Fencing	7.7	0	0	0	3.1
Primate	▪ Cutting neighbouring trees	0	0	10	0	1
	<i>Active deterrence methods that do not cause harm to wildlife</i>					
Primate	▪ Guarding	10.3	5.7	0	0	6.1
Primate	▪ Shouting	41	65.7	50	14.3	46.9
Primate; squirrel	▪ Throwing stones/ wood	39.5	25.7	30	21.4	31
Primate	▪ Using guns as warning	15.4	11.4	20	7.1	13.2
	<i>Active deterrence methods that can injure/kill wildlife</i>					
Primate	▪ Chasing with dogs	7.7	37.1	0	14.3	18.4
Primate	▪ Shooting	2.6	8.6	10	57.1	13.3
Primate	▪ Trapping	2.6	2.9	0	14.2	4.1

Table 9. Percentage of interviewees reporting the occurrence of farmers harming primates at each study site (* LTM = long-tailed macaque; PTM = pig-tailed macaque; TLM = Thomas' leaf monkey; GSL = Griffith's silver langur; OU = orangutan).

	Sampan Getek (n = 39)	Tangkahan (n = 35)	Bukit Lawang (n = 10)	Telaga Said (n = 14)	Mean (n = 98)
Never seen/done anything	71.8	28.3	4	0	39.1
Saw someone killing a LTM* to eat it	5.1	11.2	0	0	6.0
Saw someone killing a TLM* to eat it	0	5.7	0	14.3	4.1
Saw someone killing a LTM* and discarding the carcass	2.6	2.9	0	0	2.1
Saw someone injuring a LTM*	2.6	5.7	0	0	3.1
Saw someone catching a LTM* to keep it as pet	8.1	14.3	1	0	8.4
Saw someone catching a TLM* to keep it as pet	2.6	5.7	0	0	3.1
Saw someone catching a PTM* to keep it as pet	2.6	0	0	0	1.0
Saw someone catching an OU* to keep it as pet	2.6	2.9	0	0	2.1
Killed a LTM* and discarded the carcass	0	0	0	35.8	5.1
Killed a TLM* and discarded the carcass	0	0	1	14.3	2.1
Killed a GSL* and discarded the carcass	0	0	0	21.4	3.1
Killed an OU* and discarded the carcass	0	0	1	0	0.1
Injured an OU*	0	2.9	1	7.1	2.1
Caught a LTM* and kept it as a pet	0	8.9	0	7.1	4.2

creation of a non-agricultural buffer zone might not be possible. Consequently, a monoculture of unattractive crops (for example tea or coffee; Naughton-Treves *et al.* 1998) might act as a buffer to discourage primate crop-raiding. Alternative buffer crops could also be medicinal plants not raided by wildlife (Rao *et al.* 2002). Deliberate preservation of wild food species (for example *Ficus* spp.) could also reduce primate crop-raiding in some seasons (Naughton-Treves *et al.* 1998). In all circumstances, highly palatable crops should not be cultivated on the edge of primate habitat (Naughton-Treves *et al.* 1998). However, some crop species (for example, rambutan, cempedak and cocoa) thrive in conditions that mimic their natural habitat, i.e., in the cool and shady understorey of the rainforest (Cobley 1976; Rice *et al.* 1991). Young durian trees also must be shaded. Yaacob and Subhadrabadhu (1995) recommended (i) inter-planting durian trees with banana or papaya trees to provide shade and (ii) not cultivating durian in the forest.

Many farmers considered that shooting or trapping primates was the most successful preventative measure, although shooting them was not the most commonly reported method used except in Telaga Said. Islam forbids the consumption of primate meat, but Muslim farmers in Telaga Said reported killing primates and then either discarding the carcass or giving it to Christian neighbours. The small number of farmers using guns in Tangkahan and Bukit Lawang may be attributed to the proximity of the Gunung Leuser National Park and the presence of wildlife tourism facilities. In Tangkahan, the village community agreed in 2001 to turn the site into an environmentally-oriented tourist destination (Jakarta Post 2003). The Gunung Leuser National Park Office entrusted local people in a small concession area within the Park with the management of ecotourism, and local youths were trained as tourist guides (Jakarta Post 2003). The direct involvement of the local community in ecotourism may prevent the use of guns as a protection method at this particular site. In Sampan Getek, surveys of orangutans have been done since late 2001 (Singleton *et al.*

2002a), and it is possible that the occasional presence of biologists and conservationists in this area might be preventing the use of guns as a preventative measure against primates. The figures presented here about primate shooting are probably not, however, a true picture of the exact frequency of gun use. Interviewees may have considered that confessing to using guns would get them into trouble because they were aware that some primates are officially protected.

The large proportion of interviewees reporting they had never harmed or seen another individual harming primates may reflect a fear of being punished for such acts. Consequently it is very possible that farmers under-reported the incidence of using lethal methods to deter wildlife from damaging their crops. Only a small proportion of farmers complained about the Sumatran orang-utan, but conservationists should be vigilant. Given the Critically Endangered status of the Sumatran orangutan, one might suggest relocating problem animals. However, this procedure is very expensive and risky, and is not guaranteed to be successful (Singleton *et al.* 2002a). We suggest that orangutans should be removed only in extreme cases, after detailed study, and only if suitable alternative habitat is available elsewhere. However, it is important to be aware that eradication of all crop-raiding primates from an area, though perceived by farmers as the most effective protection method, is usually only a short-term solution, as other primate groups may quickly invade the newly available home range (Strum 1987; Osborn and Hill 2005).

Conclusion

Wild vertebrates, and especially primates, were believed to inflict substantial losses on crops in the four villages surveyed. The perceived impacts of primate crop-damage on local people are of great importance for conservationists, because if local people attach a negative value to wildlife they will not support its continued existence in the region (Gillingham and Lee 2003; Hill 2004). Primates are considered

to be particularly successful crop-raiders because they can cross fences with ease (Newmark *et al.* 1994; Hill 2002) and often wait for the farmers to leave before raiding their gardens (Kavanagh 1980). As a result, farmers may be only marginally successful in preventing their crop-damage (Saj *et al.* 2001). An immediate concern, therefore, is the development of effective, non-lethal, humane methods to mitigate human-primate conflict.

The study presented here provides a snapshot of the primate crop-raiding issue in four villages of North Sumatra. More extensive fieldwork is needed to examine more fully some of the issues we have outlined, and could contribute to the creation of a centralized database on the human-wildlife conflict issue in North Sumatra. There is a need for further studies to cover a much larger area. Successful measures to protect primates using agricultural areas in Indonesia will be a central issue as increasing amounts of the remaining natural forests are put under cultivation. Lowland wildlife species are likely to be even more at risk than other animals because they live on the edge of the remaining forest; exactly where local people live.

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The Distribution, Status and Conservation of Hoolock Gibbon, *Hoolock hoolock*, in Karbi Anglong District, Assam, Northeast India

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Abstract: In India, the hoolock gibbon, *Hoolock hoolock*, is found only in a small part in the northeast, south of the Brahmaputra River and east of the Dibang River. This article describes its distribution, habitat and conservation and also compares its relative status over the past 15 years in Karbi Anglong, the largest district of Assam in India. The hoolock gibbon still occurs over a large part of the district but in depleted numbers. It has been recorded from altitudes of less than 100 m to above 1,300 m. Hoolocks have become rarer due to habitat loss and hunting and, except for a few protected areas and larger reserved forests, they are found in scattered groups, where they may not survive for long. Karbi Anglong has the largest known habitat and estimated population of the species in Assam. A rough population estimate indicates that the total numbers of hoolock gibbons today could be between 2,400 and 3,200. This can be compared to an estimate in 1991–1992 of 3,500–4,800. The hoolock gibbon is protected by law and occurs in all the five protected areas and in at least 20 reserved forests and 14 proposed reserved forests in the district. Of these, its continued presence is doubtful in at least four reserved forests and one proposed reserved forest. The creation of further protected areas, adequate protection of existing protected areas, control of *jhum* cultivation and poaching, and awareness and involvement of churches and village headmen in conservation are recommended.

Key words: Hoolock gibbon, *Hoolock hoolock*, northeast India, Assam, Karbi Anglong, Mikir Hills, Dhansiri

Introduction

The hoolock gibbon, *Hoolock hoolock*, is the only ape found in the Indian subcontinent. Adult males and juveniles of both sexes are black with white eyebrows. When subadult, the pelage of the females changes to greyish and then to a tan color, which they retain as adults. The range of the hoolock gibbon is between the Brahmaputra and Salween rivers, covering parts of northeast India, eastern Bangladesh, northern Myanmar (Burma), and a small area of southern China (McCann 1933; Groves 1972; Choudhury 1987). In India, it is confined to the northeast, where it is restricted to the south of the Brahmaputra River and east of the Dibang River (Parsons 1941; Choudhury 1987). Its range in northeast India was shown incorrectly in Corbet and Hill (1992). The type locality of the species is the Garo Hills in Meghalaya (originally recorded as Assam), India (Harlan 1831).

A fair amount of published information is now available on hoolock gibbons in Assam and other areas of India (see McCann 1933; Tilson 1979; Choudhury 1987, 1990, 1991,

1996, 1998, 2000, 2003, 2006; Chhetri *et al.* 2007) and there are number of synoptic works on primates or wildlife in general which also mention the species (Pocock 1939, 1941; Prater 1948; Choudhury 1988, 1997, 2001; Menon 2003; Groves 2005). Some unpublished theses and reports also provide important data (see Choudhury 1989; Misra *et al.* 1994; Kakati 1997; Das 2002). Prouty *et al.* (1983) and Mootnick and Groves (2005) reviewed the taxonomy of the species, while Takacs *et al.* (2005) reviewed that of the family Hylobatidae. Choudhury (1993) provided information on the gibbons in Karbi Anglong. In this article, I describe the distribution, habitat, status, and conservation of the hoolock gibbon in Karbi Anglong, the largest district of Assam in northeastern India, and also discuss its status over the past one and a half decades.

Study Area

The district of Karbi Anglong (25°32'–26°37'N, 92°09'–93°53'E; 10,330 km²) is in central Assam, northeast India

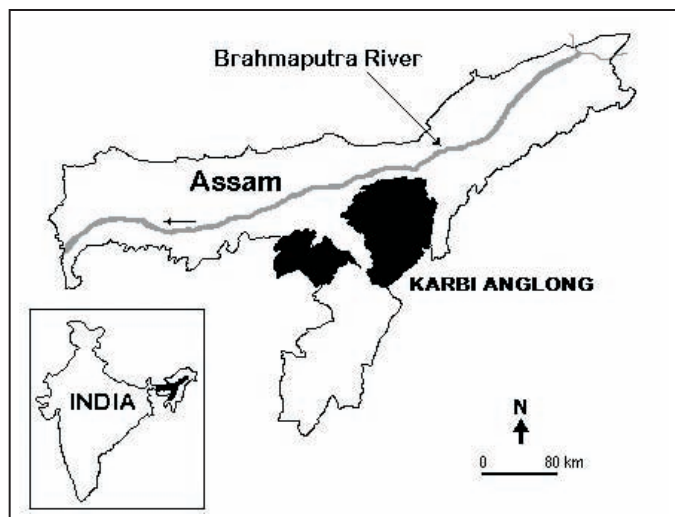


Figure 1. The location of Karbi Anglong in Assam, India. Map by Anwaruddin Choudhury, 2008.

(Figs. 1, 2). Formerly the area was called the Mikir Hills. The district consists of two disjunct regions separated by Hojai-Lanka plains. The eastern region comprises the Diphu and Bokajan Subdivisions, and the western region is the Hamren Subdivision. The area is mostly rugged and hilly, being part of an Archaean plateau (known as Karbi Plateau). The East Karbi Plateau is like an isolated 'block' linked to the main plateau through lower undulating terrain called the Lumding-Dhansiri Gap. The West Karbi Plateau is contiguous with the larger Meghalaya Plateau. There are three small plains, all formed by different rivers, namely, the Kopili, Dhansiri and the Jamuna. The highest point of Karbi Plateau is 1,360 m above sea level, and has been called the 'Hoidu Parbat' (*Hoidu* means hoolock gibbon in local Karbi parlance) by Choudhury (1993). The second highest peak is Singhason or Chenghehison Parbat (1,357 m above sea level).

The climate is tropical with hot and wet summers (May to September) and cool and generally dry winters (December to February). Annual rainfall ranges from 800 to 2,800 mm. The bulk of the rain falls during the summer monsoon (May to September). Rainfall is very variable because the southern Karbi Anglong is in a rain shadow area. The temperature ranges from less than 5°C in higher areas in winter to more than 35°C in summer (often reaching 37°C in the plains).

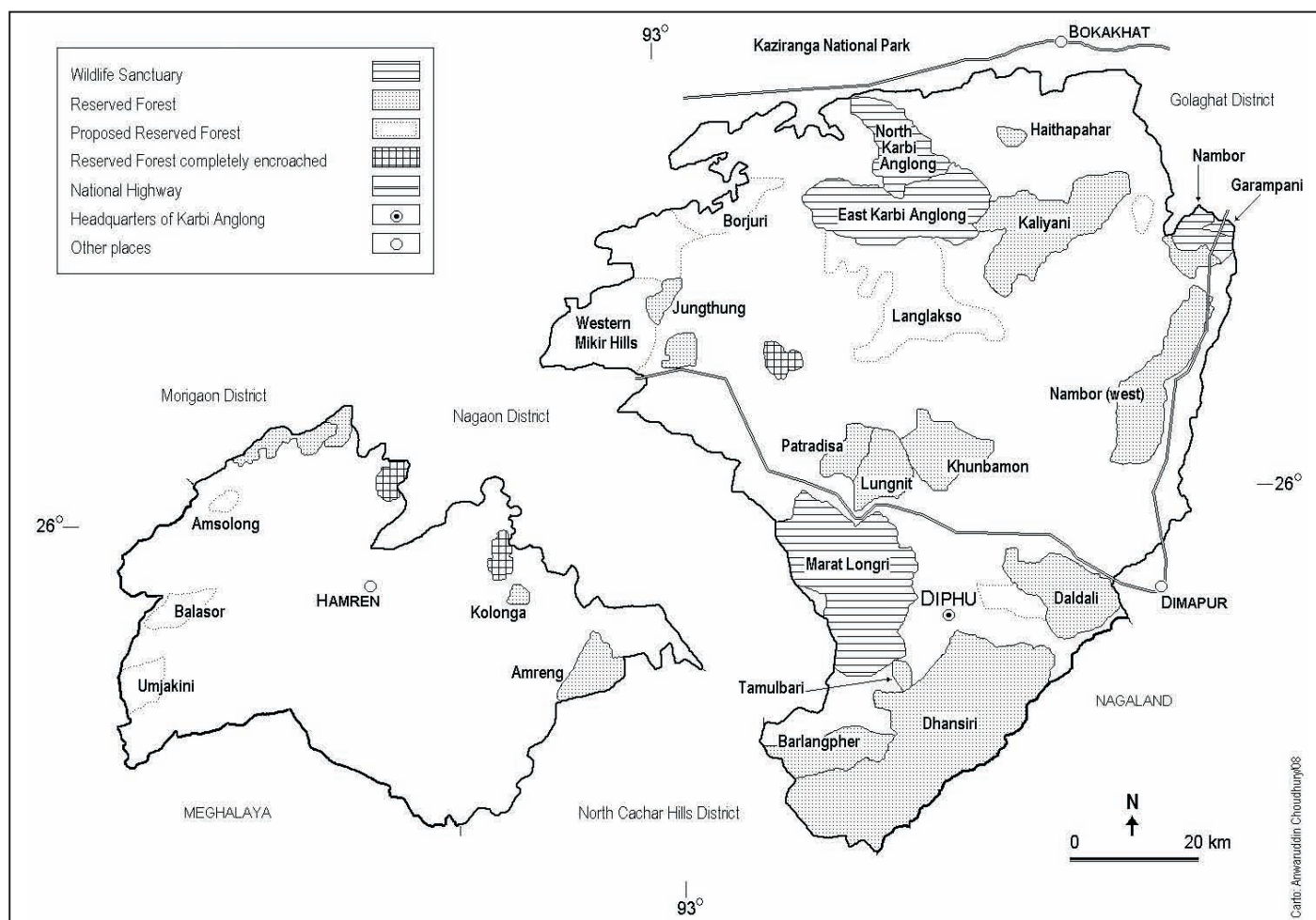


Figure 2. Karbi Anglong, showing some general features and the wildlife sanctuaries and reserved forests. Map by Anwaruddin Choudhury, 2008.

Methods

From 1991 to October 2007, I carried out field surveys in areas where hoolock gibbons occur in the district of Karbi Anglong as part of a broader survey of wildlife of northeast India. The presence of gibbons was ascertained by direct sightings or by hearing their calls, as well as through finding preserved skulls in the tribal villages and by interviewing local forest staff, villagers, and hunters (using visual aids such as photos and drawings). Direct observations and censuses were carried out along trails (mostly during foot-transect), roads (by car), and rivers (by boat).

The data were obtained during numerous field surveys carried out since 1991, and particularly between April 1991 and June 1992, when I was posted as Additional District Magistrate at Diphu, the headquarters of Karbi Anglong. Visits were sporadic after that time, but were made every year until October 2007.

Distribution

Hoolock gibbons are still widely distributed in Karbi Anglong (Fig. 3.). The species has been recorded all over the hilly and forested areas of Bokajan and Diphu subdivisions

in the eastern part of Karbi Anglong. It vanished many years ago, however, from the flat plains of Howraghat and Bokajan, due to forest clearing for human settlements. The range is largely contiguous in the central, northern and southern areas where there are still large tracts of forest. In Hamren Subdivision in the western part of the district, the range is discontinuous with only a number of fragmented pockets remaining. Hoolock gibbons have been recorded in all the wildlife sanctuaries of the district: East Karbi Anglong, Garampani, Nambor, North Karbi Anglong and Marat Longri. The protected areas, reserved forests and proposed reserved forests where they are known to occur are listed in Table 1. The larger reserved forests and proposed reserved forests still containing gibbons are Dhansiri, Langlokso, Nambor (west block), Khunbamon and Borjuri (in Table 1). They are still found in some unclassified forests as well, but in small numbers. In Hamren Subdivision of western Karbi Anglong and in parts of the central highlands in eastern Karbi Anglong populations were found to be small and isolated, in forest patches dispersed through *jhum* (slash-and-burn shifting cultivation of the hill tribes) fields. Isolated groups can be found near Habang (Umwang), Bait-halangso, Karbi Langpi, Jirikinding, Amtereng and Amkarlu, and a few gibbons still survive in sacred groves, such as those in Killing Sarpo.

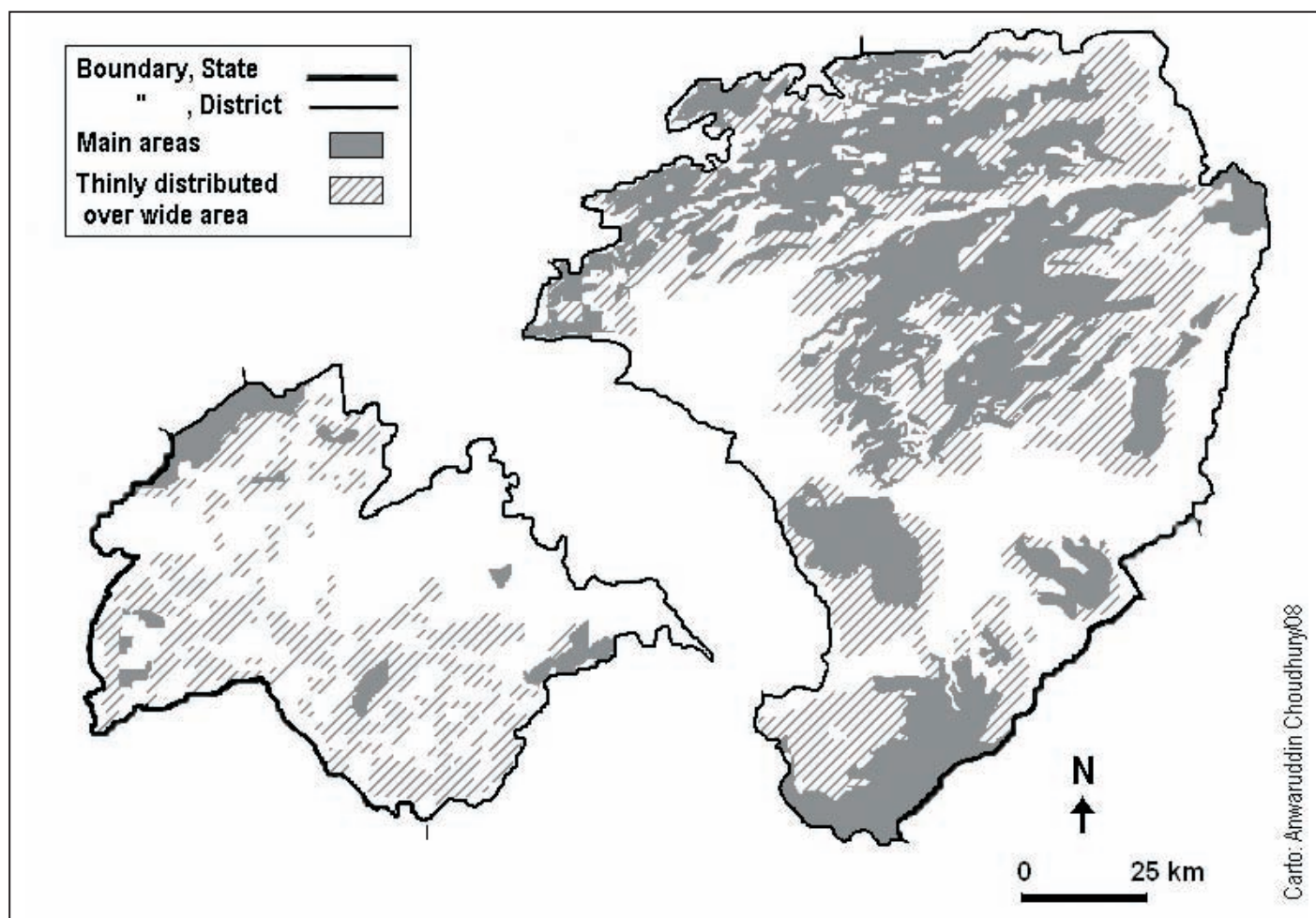


Figure 3. The current distribution of the the hoolock gibbon (*Hoolock hoolock*) in Karbi Anglong. Map by Anwaruddin Choudhury, 2008.

Table 1. Protected areas, reserved forests and other areas with known hoolock gibbon, *Hoolock hoolock*, populations. Population range: A=>200; B=100–200; C=50–100; D=20–50; E=<20; ? = current occurrence doubtful.

	Area (km ²)	Population		Remarks
		1991–92	Post-2003	
Wildlife Sanctuaries				
East Karbi Anglong	221.8	A	A	Formerly Mikir Hills RF. Largely in good condition. Small areas under <i>jhum</i> . Contiguous with North Karbi Anglong WS and Kaliyoni RF.
Garampani	6.0	D	D	Some felling. A busy National Highway passes through. Contiguous with Nambor-Doigrung WS of Golaghat district and Nambor WS.
Marat Longri	451.0	A	A	Formerly Disama, Kaki, Inglonggiri and Miyungdisa RFs. Encroachment, <i>jhum</i> and felling of trees. Contiguous with Lumding RF of Nagaon district.
Nambor	37.0	B	C	Some encroachment and felling of trees. Contiguous with Garampani WS.
North Karbi Anglong	96.0	B	B	Encroachment, <i>jhum</i> and felling of trees. Contiguous with East Karbi Anglong WS and Kaziranga National Park.
Reserved Forests (including District Council Reserved Forests)				
Amreng	57.0	B	D	Encroachment (up to 17 km ² in 2007) and felling of trees.
Amreng 1 st Addition	5.8	E	E	Separated from Amreng RF by an all weather road. Amreng 1 st Addition was notified as a District Council RF (*see footnote below).
Barlangpher	77.3	C	D	Large-scale <i>jhum</i> and also encroachment. Contiguous with Dhansiri RF.
Daldali	123.3	A	B	Encroachment and felling of trees. Poaching from Nagaland.
Dhansiri	770.4	A	A	Encroachment (up to 33 km ² by 2007) and felling of trees but due to sheer size, still contains the single largest contiguous habitat. Poaching from Nagaland. Contiguous with Intanki National Park of Nagaland.
Haithapahar	54.4	C	D	Encroachment, <i>jhum</i> and felling of trees.
Jungthung	33.0	B	D	Some encroachment, <i>jhum</i> and felling of trees.
Kaliyoni	209.0	A	B	The entire valley area is under encroachment. Elsewhere <i>jhum</i> , felling of trees and poaching. Contiguous with East Karbi Anglong WS.
Khunbamon	166.0	A	C	Encroachment (up to 99.3 km ² by 1997; some evicted), <i>jhum</i> , felling of trees and poaching.
Kolonga	17.35	C	E	Encroachment (up to 10 km ² by 1997; some evicted but still about 8 km ² is under encroachment in 2007) and felling of trees.
Lungnit	118.0	B	D	Encroachment (up to 35.3 km ² by 1997; some evicted), <i>jhum</i> and felling of trees. Contiguous with Patradisa RF.
Mahamaya	5.6	?	?	Last seen in 1976, a pair.
Matipung	13.3	E	?	Encroachment being located close to Diphu town; also <i>jhum</i> and felling of trees.
Nambor (north block)	11.0	D	E	Large-scale encroachment (out of total area of 54 km ² , 43 km ² forms Garampani and Nambor wildlife sanctuaries leaving only 11 km ² as RF).
Nambor (west block)	166.3	A	B	Encroachment, felling of trees and poaching.
Patradisa	67.0	B	D	Encroachment (up to 13.5 km ² by 1997; some evicted) and <i>jhum</i> . Contiguous with Lungnit RF.
Sarchim/Charchim	133.1	C	E	Encroachment, <i>jhum</i> and felling of trees.
Sildharampur	21.5	E	?	Encroachment and felling of trees. A busy National Highway passes through.
Sinanadi/Chinanadi	19.8	E	E	Encroachment (up to 10 km ² by 1997), now cleared but small gibbon habitat.
Tamulbari	13.9	D	E	Encroachment and felling of trees. Separated from Dhansiri RF by a railway track.
Proposed Reserved Forests				
Amreng, 2 nd addition	55.3	D	E	Encroachment and felling of trees.
Amsolong	74.7	C	D	Encroachment, <i>jhum</i> and felling of trees.
Balasar	82.8	C	D	Encroachment (up to 20 km ² by 1997), <i>jhum</i> and felling of trees. Contiguous with forests in Meghalaya.
Bokajan	9.8	E	?	Encroachment and felling of trees.
Borjuri	139.0	A	C	Encroachment, <i>jhum</i> and felling of trees.
Dolamara	5.5	D	E	Encroachment, <i>jhum</i> and felling of trees.
Haithapahar	54.0	E	E	Encroachment, <i>jhum</i> and felling of trees.
Hapjan	35.25	E	E	Encroachment and felling of trees.
Kalapahar	9.8	D	E	Encroachment, <i>jhum</i> and felling of trees.
Kaziranga	33.9	D	E	Encroachment and felling of trees.
Langlokso	534.7	A	A	Large-scale encroachment (up to 214 km ² ; some evicted but still about 64 km ² is under encroachment in 2007) and felling of trees. Also, ginger cultivation, <i>jhum</i> and poaching. Contiguous with East Karbi Anglong WS.
Parkup Pahar	27.7	D	E	Encroachment, <i>jhum</i> and felling of trees.
Tikok	25.3	C	E	<i>Jhum</i> and felling of trees.
Umjakini	36.8	D	E	Encroachment (up to 6 km ² by 1997), <i>jhum</i> , felling and poaching. Contiguous with forests in Meghalaya.
Western Mikir Hills	173.0	B	C	Encroachment, <i>jhum</i> and felling of trees.

Table 1. continued on next page

Table 1., continued

	Area (km²)	Population		Remarks
		1991–92	Post-2003	
Unclassed Forests				
Scattered across Singhason area outside Langlokso		A	C	Large-scale <i>jhum</i> , ginger cultivation and human settlement (not necessarily encroachment as these are not reserved forests), also poaching.
Scattered across Hamren sub-division		A	C	Same as above.
Gandhipur		E	?	c.5 km east of Hidipi
Tapat		E	?	
Umpabeng		E	?	
Habang (Umwang)		E	E	
Killing Sarpo sacred grove		D	E	Traditionally protected but small in size.

WS=wildlife sanctuary; RF=reserved forest; PRF=proposed reserved forest.

* Earlier District Council RFs were declared and controlled by the District Council while the other reserved forests were administered by the State Government. Now all the reserved forests are controlled by the Council (now upgraded from District Council to Autonomous Council). Hence, there is no longer any functional difference between these two categories of reserved forests.

In all, there are eight large fragmented forests and more than 15 smaller isolated pockets in eastern Karbi Anglong, and two large forests and more than 20 smaller isolated pockets in western Karbi Anglong (Table 2). Dhansiri RF, East Karbi Anglong Wildlife Sanctuary (formerly Mikir Hills RF), and Marat Longri Wildlife Sanctuary (especially Disama and Kaki RF areas) are the key areas for hoolock gibbons, with extensive forests and relatively large populations. Nambor (north block) forests, including Nambor and Garampani wildlife sanctuaries, are also important, but their small size, along with encroachment and felling of trees, are threats to their integrity and to the small gibbon populations remaining. Some of the forests there are contiguous with those in neighboring states (Nagaland and Meghalaya) and districts (Golaghat, Morigaon, Nagaon and North Cachar Hills) (see ‘Remarks’ in Table 1).

Habitat

A strictly forest-dwelling primate, the hoolock gibbon is found in two major types of habitat in Karbi Anglong: tropical moist deciduous and tropical semi-evergreen forests. The deciduous forests of most parts of north-eastern India, however, are not pure stands of deciduous trees but contain large numbers of evergreen trees, and there are also patches of semi-evergreen forests within the deciduous biotope forming a mosaic. The hoolock gibbon has also been observed in bamboo thickets amidst semi-evergreen or deciduous forests. Suitable gibbon habitat in the form of tropical semi-evergreen forest occurs in patches on the northern slopes and in the Nambor forests towards the north-east. The ‘hollong’ *Dipterocarpus macrocarpus* (its western limit of distribution is north-east Karbi Anglong), *Terminalia myriocarpa*, *Duabanga sonneratioides*, *Artocarpus chaplasi* and *Mesua ferrea* are some of the notable trees. Tropical semi-evergreen forest also occurs elsewhere in patches, especially along streams.

A large part of the gibbon’s range is covered with tropical moist deciduous forest. The ‘sal’ *Shorea robusta*, and the emergent *Tetrameles nudiflora*, *Gmelina arborea*, *Dillenia scabrela*, and *Bombax ceiba* are some of the species typical of these forests. The main bamboo species are *Dendrocalamus hamiltonii* and *Oxytenanthera nigrociliata/parvifolia*. Sal, however, does not occur in Dhansiri RF. An endemic species, *Mansonia dipikae* occurs in the southern part of Karbi Anglong, including Dhansiri RF. Grasslands and pine groves (*Pinus kesiya*) predominate in the tablelands of West Karbi Anglong (Hamren Subdivision), and gibbons do not occur there. The hoolock gibbons evidently prefer the mixed patches and pockets of evergreen forest. Bamboo and scrub have invaded large areas in the abandoned *jhums* in the deciduous forests. Gibbons can also be found in small forest patches in plantations of deciduous species such as ‘teak’ *Tectona grandis*, but not in the pure teak areas.

Most of the present habitat of the hoolock gibbon in Karbi Anglong is in the hills or in rugged or low undulating terrain. This is mainly because of the destruction of forest for intensive paddy cultivation in the low lying plains. Hoolock gibbons have been recorded at altitudes of less than 100 m to more than 800 m in the western part (Hamren) and up to 1,300 m above sea level in Singhason and the adjacent highlands. The known “area of occupancy” (see IUCN 2008) of hoolock gibbons in Karbi Anglong is about 8,000 km² (down from around 9,000 km² in 1991–92) of which about 4,000 km² (down from around 5,500 km² in 1991–1992) could be considered to still have a viable gibbon metapopulation. Only scattered individuals or isolated groups are found in the remaining 3,500 km².

Status

Although quite widespread, the hoolock gibbon has become rare in the district except for a few protected areas and reserved forests. Choudhury (1993) mentioned that it was

common in reserved forests such as Dhansiri, Nambor and Mikir Hills, but the situation is today very different except for a few pockets where numbers still remain relatively high. In the Dhansiri and Nambor forests, there is encroachment and felling of trees with occasional poaching. Mikir Hills RF was declared as a wildlife sanctuary following a recommendation by Choudhury (1993), but there is no enforcement. In the areas where *jhum* cultivation is extensive, lone animals or groups can be isolated by 5 or 10 km from other groups. Similarly, even in some reserved forests with encroachment, their distribution is sparse and scattered, for example, in Khunbamon and Nambor (west block). In Hamren Subdivision, the area of occupancy is around 2,000 km², but in about 1,600 km² it is encountered only in widely separated valleys and hilltops in isolated and very fragmented small groups which really have no possibility of long-term survival. In the remaining about 400 km² also the density is nowhere near Dhansiri RF.

With small numbers thinly distributed across large areas, population estimates are difficult to obtain except for some protected areas and reserved forests. We have some idea of crude density for certain sites I surveyed in six areas in 1991–1992. Density estimates were as follows: Dhansiri RF (between Langcholi and Nailalung) 6.3 individuals/km²; Dhansiri RF (near Diphu), 0.7 individuals/km²; Dhansiri RF (Khelma), 1.67 individuals/km²; in the Nambor (north block), Garampani Wildlife Sanctuary, 3.5 individuals/km²;

Mikir Hills RF, now East Karbi Anglong Wildlife Sanctuary (between Chaprasi Rongphar to Haru Lauri Anglong and Borlangso), 4.5 individuals/km²; and the Miyungdisa DC RF, now part of Marat Longri Wildlife Sanctuary, 1.67 individuals/km².

The mean of these density estimates is 2.43 individuals/km². The highest estimate was 6.3 individuals/km² in one part of Dhansiri RF, and the lowest was 0.7 individuals/km² in another part of the same RF. Excluding the high density estimates for Dhansiri RF, Mikir Hills RF and Garampani, indicated a population (including those that are widely scattered) of 3,500–4,800 gibbons in Karbi Anglong in 1991–1992. In recent years, obtaining similar estimates has proved impossible due to social unrest and extremist activities except in the Nambor (north block)–Garampani Wildlife Sanctuary. This site indicated a decline from 3.5 individuals/km² in 1991–1992 to 2.5 individuals/km² in 2003–2004. In Dhansiri RF (between Langcholi and Nailalung) there was no apparent decline, but there are evidently sharp declines in Dhansiri RF between Diphu and Monglumukh and at Khelma, even though a quantitative assessment was not possible. Habitat loss since 1991–1992 has been severe due to encroachment in Dhansiri RF between Diphu and Monglumukh. The current population could be inferred to be between 2,400 and 3,200; well below the estimate of around 3,500–4,800 gibbons in 1991–1992.

Table 2. Large fragmented habitats of hoolock gibbon in Karbi Anglong and their long-term conservation value.

	Area (km ²)		Remarks
	Total	Suitable habitat	
Dhansiri – Barlangper RF	847.7 (950.0 with Sarkihading forests)	600.0 (650.0 with Sarkihading forests)	Separated from Tamulbari and Daldali RFs by a railway track, and from Intanki NP by Dhansiri River. It is, however, contiguous with the forests of Sarkihading range of North Cachar Hills district.
North Karbi Anglong WS East Karbi Anglong WS Kaliyoni RF Langlokso PRF Kaziranga PRF	1094.6	600.0	Separated from other areas by <i>jhum</i> , degraded habitat and human settlements. The human settlements along the Kaliyoni River and then along the road to Samelangso are going to divide this large habitat into two within the next half decade or so.
Marat Longri WS	451.0 (675.0 with Lumding RF)	400.0 (550.0 with Lumding RF)	Separated from Lungnit and Patradisa RFs by the Jamuna River, from Tamulbari RF by an all weather road. It is, however, contiguous with the dense forests of Lumding RF of Nagaon district. An all weather road (Diphu-Lankaijan road) passes through it but is still narrow and not busy.
Khunbamon RF Lungnit RF Patradisa RF Tikok PRF	376.3	150.0	Separated from Marat Longri WS by the Jamuna River, and from other areas by <i>jhum</i> , degraded habitat and human settlements.
Jungthung RF Borjuri PRF Western Mikir Hills PRF Parkup Pahar PRF	372.7	250.0	Separated from other areas by <i>jhum</i> , degraded habitat and human settlements.
Daldali RF Matipung RF Hapjan PRF	171.85	100.0	Separated from Dhansiri RF by a railway track, and from other areas by degraded habitat, human settlements and a national highway. Poaching from Nagaland.
Nambor (west block) RF	166.3	90.0	Separated from Nambor WS by Namburnadi tea garden, and from other areas by cultivations, degraded habitat, human settlements and a national highway. Poaching from Nagaland.
Nambor WS Garampani WS Nambor (north block) RF	54.0 (178.5 with Nambor-Doigrung WS)	35.0 (90.0 with Nambor-Doigrung WS)	Isolated. A busy national highway passes through. It is contiguous with Nambor-Doigrung WS (excluding the Lower Doigrung part, which is isolated) of Golaghat district.

RF = Reserved Forest, PRF = Proposed Reserved Forest, WS = Wildlife Sanctuary, NP = National Park

Conservation Issues

Forest loss and fragmentation

Forest destruction through tree felling, encroachment, *jhum*, and monoculture tree plantations is a major threat to the survival of the hoolock gibbon in Karbi Anglong. The forest cover in northeast India is disappearing at an alarming rate. More than 1,000 km² of forest was destroyed annually in the northeastern region of India (including Assam, Arunachal Pradesh, and other states) during the 1970s and 1980s (data from the National Remote Sensing Agency). In Assam, dense forest cover has declined from 19.9% of the geographical area in 1980–1982 to 16.4% in 2004–2005 (India, NRSA 1983; India, FSI 2005).

In Karbi Anglong, the dense forest cover was 6,044 km² or 58.5% of the geographical area in 1996–1998 (India, FSI 1999). In 2004–2005 this had dropped to 4,489 km² or 43.5% (India, FSI 2005), a net decrease of 1,109 km², or nearly a fifth in less than a decade. Encroachment is a major problem in the reserved forests, and *jhum* cultivation is an important cause of forest loss and fragmentation in hilly areas such as Karbi Anglong. The Nambor and Garampani Wildlife Sanctuaries are cut into two by a busy national highway, which the gibbons are unable to cross. The number of fragmented units is as follows: 23+ in East Karbi Anglong, and 22+ in West Karbi Anglong. This is excluding the scattered groups and individuals spread all over in the abandoned *jhums* and heavily degraded tracts, for which estimates of fragmentation are extremely difficult to obtain.

Poaching

Hoolock gibbons are hunted for food by many of the tribes of Karbi Anglong and adjacent areas. Members of the largest tribe of the area, the Karbis (formerly called the Mikirs), however, do not normally kill gibbons because local folklore has it that the gibbon is a 'Karbi who was sent to the jungle for his misdeeds'. But today, there are members of the younger generation who occasionally kill them. Other tribes from Karbi Anglong who kill gibbons for their meat are the Rengma Nagas, Kukis, Hmar, Paite, Biate, Chakmas, Khasis, and Jaintias. From across the border, various Naga tribes from Nagaland often hunt gibbons in Karbi Anglong, especially along the roads and near the border. In the past, traditional weapons such as snares and self-made muzzle-loaders were used, but the last two decades has seen the increased use of automatic firearms.

Conservation Measures Taken

The hoolock gibbon is protected under Schedule-I of the Wild Life (Protection) Act of India, which prohibits its killing or capture, dead or alive. Enforcement, however, is virtually nonexistent, even in the protected areas. Most locals are unaware of its legal status. It is listed as "Endangered." on the IUCN Red List of Threatened Species (IUCN 2008). The Autonomous Council in Karbi Anglong has already shown

its positive attitude by accepting proposals for protected areas (Choudhury 1993) and as many as five wildlife sanctuaries have been declared, all of which have hoolock gibbon populations.

Discussion

Karbi Anglong is among the most important districts in northeast India for the long-term conservation of the hoolock gibbon. The reasons are the large size of the district, the relatively large areas that still sustain dense forest, and the traditional beliefs of the Karbis that discourage hunting. The district's importance in this respect is rivaled only by the Lohit and Lower Dibang Valley districts of Arunachal Pradesh where large continuous tracts of dense forest also remain and the main local tribes (Idu and other Mishmi tribes) likewise do not hunt the hoolock gibbon.

Karbi Anglong still has 4,489 km² of dense forest (canopy cover 40% or more), which is 43.3% of the dense forest left in the hoolock gibbon's range in Assam, south of the Brahmaputra River (India, FSI 2005). Conservation measures in this district are crucial for the long-term protection of this species.

The population estimates reported by Choudhury (1989) were made without full surveys being carried out in some key areas in Karbi Anglong, such as Mikir Hills RF (now North Karbi Anglong Wildlife Sanctuary), Disama and Kaki RFs (now part of Marat Longri Wildlife Sanctuary) and Dhansiri RF. As for other areas bordering Nagaland, it was presumed that Dhansiri being located right on the border had heavy poaching. However, during actual field work it was found that the location of Intanki National Park of Nagaland just across the



Figure 4. A young hoolock gibbon (*Hoolock hoolock*) in Hamren sub-division. Immatures are black irrespective of their sex. The pelage of females gradually changes to grayish and then to tan. Photograph by Anwaruddin Choudhury.

border had acted as a buffer, and there was negligible poaching in Dhansiri. Hence, in the period 1988–1992 I believe there were in fact more than at least 9,500–10,800 hoolocks in Assam.

The decline of the hoolock population in Karbi Anglong between 1991–1992 and 2004–2006 by approximately one-third is corroborated by the loss of dense forest cover, which dropped from 6,044 km² in 1996–1998 (India, FSI 1999) to 4,489 km² in 2004–2005 (India, FSI 2005), a loss of 25.7% in less than 10 years. The dense forest recorded by the Forest Survey of India (India, FSI 2005) includes, however, all forests with crown cover of 40% or more, i.e., teak plantations, village woodland, and scattered tiny forest fragments. Hence a sizeable portion is unsuitable for gibbons. In much of the dense forest where the habitat is still ideal, gibbons have long since vanished due to hunting. On the other hand, a few groups do still survive in degraded areas. Hence, in some areas along the border of Nagaland and Meghalaya, and those inhabited by Kukis, Nagas, Chakmas, Khasis and Jaintias in Karbi Anglong, the extent of dense forest may not have much bearing on gibbon abundance and distribution. In the Karbi and Dimasa Kachari areas, they are generally not molested and hence still occur even in small patches. In the hilly area outside the reserved forests, isolated gibbons in fragments isolated due to *jhum*, the gibbons are hunted down within a short time.



Figure 5. A male hoolock gibbon (*Hoolock hoolock*) in a canopy clearing, Dhansiri Reserved Forest. Photograph by Anwaruddin Choudhury.

In parts of larger areas such as Dhansiri RF and Marat Longri and East Karbi Anglong Wildlife Sanctuaries, the gibbons have survived largely because of relative inaccessibility rather than protection or popular belief of Karbi people. Throughout its range in Karbi Anglong, the gibbon is sympatric with other primates, including the Assamese macaque (*Macaca assamensis*), stump-tailed macaque (*M. arctoides*), pig-tailed macaque (*M. nemestrina*), Rhesus macaque (*Macaca mulatta*), capped langur (*Trachypithecus pileatus*), and slow loris (*Nycticebus bengalensis*).

Owing to the rapid growth of the human population, areas under *jhum* cultivation and the demand for firewood is increasing. The human population in Karbi Anglong grew from 0.38 million in 1971 to 0.81 million in 2001, i.e., more than double in just three decades. Since the bulk of the rural population practice *jhum* as their main occupation, and new villages and hamlets appear constantly, the large-scale destruction of natural habitat seems inevitable.

Despite all these constraints, Karbi Anglong is among the few areas that has the following advantages for protecting the hoolock gibbon:

- Still large contiguous habitat and a relatively numerous population for long-term conservation;
- the largest tribe, the Karbis, do not hunt it; and
- there is already a network of protected areas.

Adequate protective measures for the reserved forests and wildlife sanctuaries, the creation of some new protected areas, and the reduction of hunting through community awareness and enforcement are the measures needed.



Figure 6. Selective logging and clearance for human habitation (such as this case in Dhansiri reserved forest) and shifting cultivation are serious threats to gibbon habitat in Karbi Anglong. Photograph by Anwaruddin Choudhury.



Figure 7. A national highway passes through the Nambor forests (Garampani and Nambor wildlife sanctuaries) separating its gibbon population besides facilitating poaching by hunters from Nagaland. Also in the photo are a wild tusker *Elephas maximus* and the author. Photograph by Nur Hussain.

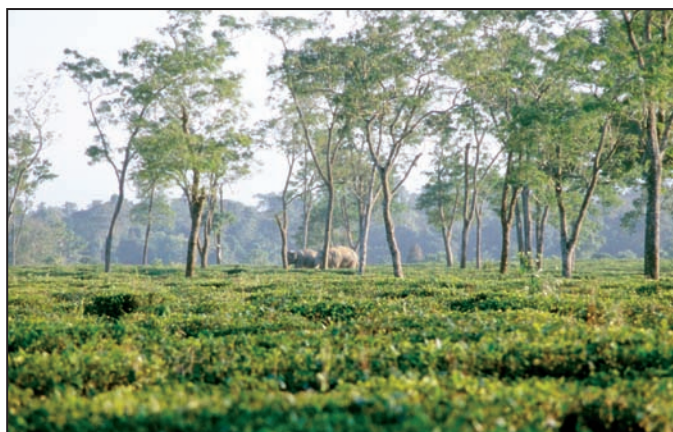


Figure 8. Much gibbon habitat has been lost to tea plantations, especially in eastern Karbi Anglong. Photograph by Anwaruddin Choudhury.



Figure 9. Exploring the interior of Karbi Anglong required crossing such fords across many unbridged rivers, such as this one on the Horgati River. Photograph by T. K. Barman.

Recommendations

A number of important known habitats for hoolock gibbons that are outside the protected area network should be declared as wildlife sanctuaries, conservation reserves and community reserves. Dhansiri should be changed from a reserved forest to a wildlife sanctuary. Dhansiri has also been recommended for a 'tiger reserve' (Choudhury 1992, 1993). Amreng RF, Kolonga RF, Jungthung RF, Tamulbari RF and the proposed Balasor RF are important areas for the hoolock gibbon and should receive better protection. Some of the smaller scattered gibbon refuges, including the sacred grove of Killing Sarpo should be declared 'community reserves' for the development of eco-tourism with community involvement.

Existing protected areas should be better protected, with increased staff, anti-poaching camps and regular patrolling. Measures should be taken to control *jhum* cultivation as well as hunting for meat. Awareness campaigns should involve the churches and the village headmen to promote conservation measures, and programs should be set up for the regular monitoring of the gibbon populations in select sites, such as those at Garampani-Nambor, Marat Longri, North Karbi Anglong and Dhansiri.

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Status of Western Hoolock Gibbon (*Hoolock hoolock*) Populations in Fragmented Forests of Eastern Assam

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Abstract: A survey was carried out at 14 sites to investigate how forest fragmentation affects populations of the Endangered western hoolock gibbon. Encounter rates were used as an index of gibbon population densities, and gibbon group size and age class ratios as an index of the status of the population. The 28-day survey was carried out in May and June 2002 in the Doomdooma, Dibrugarh, Digboi and Tinsukia Forest Divisions of Upper Assam. Sites comprised 11 forest fragments in two size classes (small <5 km² and medium 20–30 km²) and three large forest-tracts (>100 km²) that served as controls. Two survey teams, each of three to five people, sampled each site over two days, walking between 6–21 km/site. Encounter rates for gibbon groups were lowest (0.09/km) in the small forest fragments, increasing as the forest size increased (0.23/km in 20–30 km² forest fragments and 0.58/km in the controls). Similar trends were recorded with group sizes. The smallest groups (mean 2.5, n = 2) were in the small fragments. Larger groups were found in the mid-size fragments (mean 3.29, n = 24) and the three large forest-tracts (mean 3.9, n = 28). Although infant-to-female ratios were similar among size classes, the total young (infant and juvenile)-to-female ratio was as low as 0.5 in the <5 km² size class forests. The ratios were higher in the 20–30 km² and 100 km² size classes; 1.28 and 1.46, respectively. Ways that forest fragmentation affects hoolock gibbons are discussed.

Key Words: Assam, forest fragments, western hoolock gibbon, *Hoolock hoolock*, population estimates

Introduction

The western hoolock gibbon (*Hoolock hoolock*) occurs in the western-most extreme of the distribution of the 16 gibbon species currently recognized (Geissmann 2007). Its range between the Brahmaputra and Chindwin rivers takes in three countries; Bangladesh, India and Myanmar. Preliminary surveys in Myanmar indicate that it occurs at least as far south as Rakhine Yoma in south-west Myanmar (Geissmann *et al.* 2008). It has been on the list of the World's 25 Most Endangered Primates since 2006 (Walker *et al.* 2007), with the global population estimated to be about 5,000 animals: 2600 to 4450 in India (Molur *et al.* 2005; Choudhury 2006), and about 200 in Bangladesh (Molur *et al.* 2005). The Myanmar population although not well known might be significant, but further surveys are needed there. Most populations of the western hoolock are isolated and small, with 80% of those assessed in India and Bangladesh harbouring fewer than 20 individuals, and over half having fewer than 10 (Walker *et al.* 2007).

The decline of the hoolock gibbon has been caused by the destruction, degradation and fragmentation of its forests for settled and shifting agriculture, plantations, logging, fuel-wood collection, and development projects such as mining, roads, and railways. Poaching of wildlife, including gibbons, for food and trade is common among the hill tribes of north-east India (Srivastava 1999; Choudhury 2006) leading to empty forests even where the habitat might be intact.

This survey was part of a longer study by the first author (Kakati 2004) to investigate the effects of forest fragmentation on the hoolock gibbon. It is recognized that the fragmentation of large, contiguous and undisturbed forests into small patches is one of the most serious threats to biodiversity. The effects of isolation into small forest patches are compounded by certain highly specialized gibbon life-history characteristics, notably frugivory, arboreality, territoriality and monogamy. The specific objective of our 14-site survey was to compare encounter rates, group sizes and age-sex ratios of western hoolock gibbon in lowland forest fragments with those in large, relatively undisturbed forest.

Study Area

We conducted the survey in lowland tropical evergreen and semi-evergreen forest fragments and continuous forest in the Digboi, Doomdooma, Dibrugarh and Tinsukia Forest Divisions of eastern Assam, north-east India, within a 40-km radius of the oil town of Digboi (Fig. 1). The original vegetation of these fragments was Assam Valley Tropical Wet Evergreen Forest (category 1B/C1) (Champion and Seth 1968), also called the Upper Assam *Dipterocarpus-Mesua* forest. The forest reserves have old mixed plantations with a number of deciduous species. Soil type is old alluvium of the Brahmaputra and Dehing rivers. The topography is flat *kurkani*, characterized by earthen mounds and dissected by seasonal streams in the Doomdooma and Tinsukia Forest Division, and undulating to hilly in the Dibrugarh and Digboi Forest Division reserves. Altitude ranges from 122 m to 475 m above sea level. The climate is tropical monsoonal characterized by high humidity and rainfall (2,226–2,372 mm). The monsoons last from June to September; July is the month of heaviest rainfall. There is a relatively dry period from November to February. Average temperature ranges from 6°C to 38°C. There are from 119 to 164 rainy days/year (Das 1965; Chand 1990; Choudhury 1995). Sympatric primates at the survey sites include the Assamese macaque (*Macaca assamensis*), northern pig-tailed macaque (*Macaca leonina*), rhesus macaque (*Macaca mulatta*), capped langur (*Trachypithecus pileatus*), and Bengal slow loris (*Nycticebus bengalensis*).

We selected the 14 forest reserves on the basis of size and similarity of habitat and topography (see Table 3 in

'Results'). They were categorized as small (<5 km²), medium (20–30 km²) and large (>100 km²). The reserves were in matrices of tea-plantations, paddy cultivation and villages. All were within the geographical coordinates given in Table 1.

Table 1. Geographic coordinates circumscribing the survey sites.

	Latitude (N)	Longitude (E)
North	27° 45' 26.4	95° 44' 41.8
South	27° 09' 23.0	95° 27' 03.8
West	27° 24' 32.0	95° 21' 01.7
East	27° 21' 43.3	96° 00' 50.3

Table 2. Characteristics used to classify hoolock gibbons into different age and sex classes.

Age Category	Sex	Distinguishing characteristic(s)
Adult (8 years +)	Male	Black coat, scrotum prominent, usually scowling expression
	Female	Light brown/blonde coat
Sub adult (6–8 years)	Male	Black coat, slightly smaller than adult male, scrotum distinct, facial expression gentler
	Female	Coat colour lightening from black to brown, smaller than adult female
Juvenile (2–6 years)	Male	Black coat, size small, small scrotal sac seen if inspected carefully
	Female	Black coat, size small, no scrotum
Infant (0–2 years)	Male & female	Carried by adult female. White to light grey coat colour (<1 year), dark grey or black (1–2 year old). Cannot determine sex in the field.

Table 3. Encounter rates for western hoolock gibbon, *Hoolock hoolock* (63 encounters of 59 different groups) at each site and in each forest size class.

		At Each Site			In Each Size Class		
Site (RF)	Size (km²)	No. of encounters	Distance walked (km)	Encounter rate (per km)	No. of encounters	Distance walked (km)	Av. encounter rate (per km)
Forest size class <5 km² (n = 4)							
Phillobari	3.17	0	6.25	0.00	3	32.47	0.09
Nalani	3.74	0	6.05	0.00			
Tokowani	5.02	1	11.27	0.09			
Borajan	5.05	2	8.91	0.22			
Forest size 20–30 km² (n = 7)							
Tarani	20.4	1	16.05	0.06	28	123.34	0.23
Buridihing	22.95	8	20.81	0.38			
Kumsang	22.52	3	16.38	0.18			
Kakojan	23.47	9	15.2	0.59			
Doomdooma	28.81	4	16.12	0.25			
Tinkopani	30.34	0	20.39	0			
Dirok	30.42	3	18.4	0.16			
Forest size >100 km² (n = 3)							
Dehing East	129.00	16	17.85	0.90	32	55.13	0.58
Dehing West	280.00	6	21.06	0.28			
Jeypore	108.00	10	16.24	0.62			
All sites (n = 14)							
	712.90	63	210.97				

Methods

The 28-day survey was carried out in the months of May and June 2002 by two survey teams, each of 3–5 people. Observations in the previous year had shown that both fragment and large forest groups call on a similar number of days during the rainy season of fruit abundance, while there is variation in calling rates during the drier winter season (Kakati 2004). The survey was thus conducted in the rainy season to avoid this possible source of bias in the detection of groups through calls. Each team comprised the team leader, 1–2 field assistants and 1–2 forest guards. All observations were recorded by the team leaders. We sexed and classified all hoolock gibbons seen into one of four age classes (adult, sub-adult, juvenile or young) based on body size and coat color (Table 2). With interbirth intervals being 2–3 years, a group can have two juveniles of different ages (*c.* 2–4 years and *c.* 4–6 years) differing slightly in size. We included both in a single category (juvenile), however, because of the difficulty in accurately differentiating between the two during a brief sighting.

We conducted 54 census walks covering 211 km across 14 sites. Each of the 14 sites was sampled over two days, with four walks/site (except in the Kumsang and Phillobari Reserve Forests [RF], where we did only three). We used existing foot-trails or elephant-trails when possible, and roads were avoided as far as practicable. We used 24 forest roads in whole or part during 34 out of 54 walks. Of these, four had daily vehicular traffic, six were occasionally used and 14 were either unfit

for road traffic or had been abandoned. We covered between six and 21 km/site at an average speed of 1.3 km/hour. We walked shorter distances (*c.* 8 km) in the smallest fragments because of their small sizes compared to the larger fragments and the large forests (*c.* 17 km)/site. The surveys were begun at 05:05–07:55 and ended between 07:00 and 12:45. They were completed in 199 min on average (range: 50–352 min).

We located gibbons by seeing them from the trail ($n = 26$) or by homing in on calls ($n = 37$). We tried to locate all calling groups when their distance from the trail was estimated to be less than 500 m. Groups usually call for about 15 minutes; sufficient time to find them, although not all were located. Nine of the 37 calling groups were located at >500 m from the trail. The mean sighting distance from the trail for the groups we saw was $45.16 \text{ SE} \pm 5.3 \text{ m}$ (range 5–110 m, $n = 26$) (Fig. 2a). The average distance over which calls were heard and which led to their subsequent location was $336 \text{ SE} \pm 35.2 \text{ m}$ (range 68–905 m, $n = 37$) (Fig. 2b). We also recorded the time and angle from the trail, and distance was noted as <500 m (very loud), 500–1,000 m (clear), $>1,000$ m (faint), and >2000 m (very faint) for all groups heard calling but not located (tracked down) during the walks. The GPS Trackmaker Version 11.7 software (Ferreira 2002) was used to plot this information on maps and to carry out spatial analyses of the survey data.

A minimum estimate of groups heard calling in each reserve was arrived at by deducting from the total count of calls heard, the following categories of calls: (a) calls of groups sighted after homing in on their calls, (b) calls of groups possibly outside the reserve boundary, and (c) all possible repeat

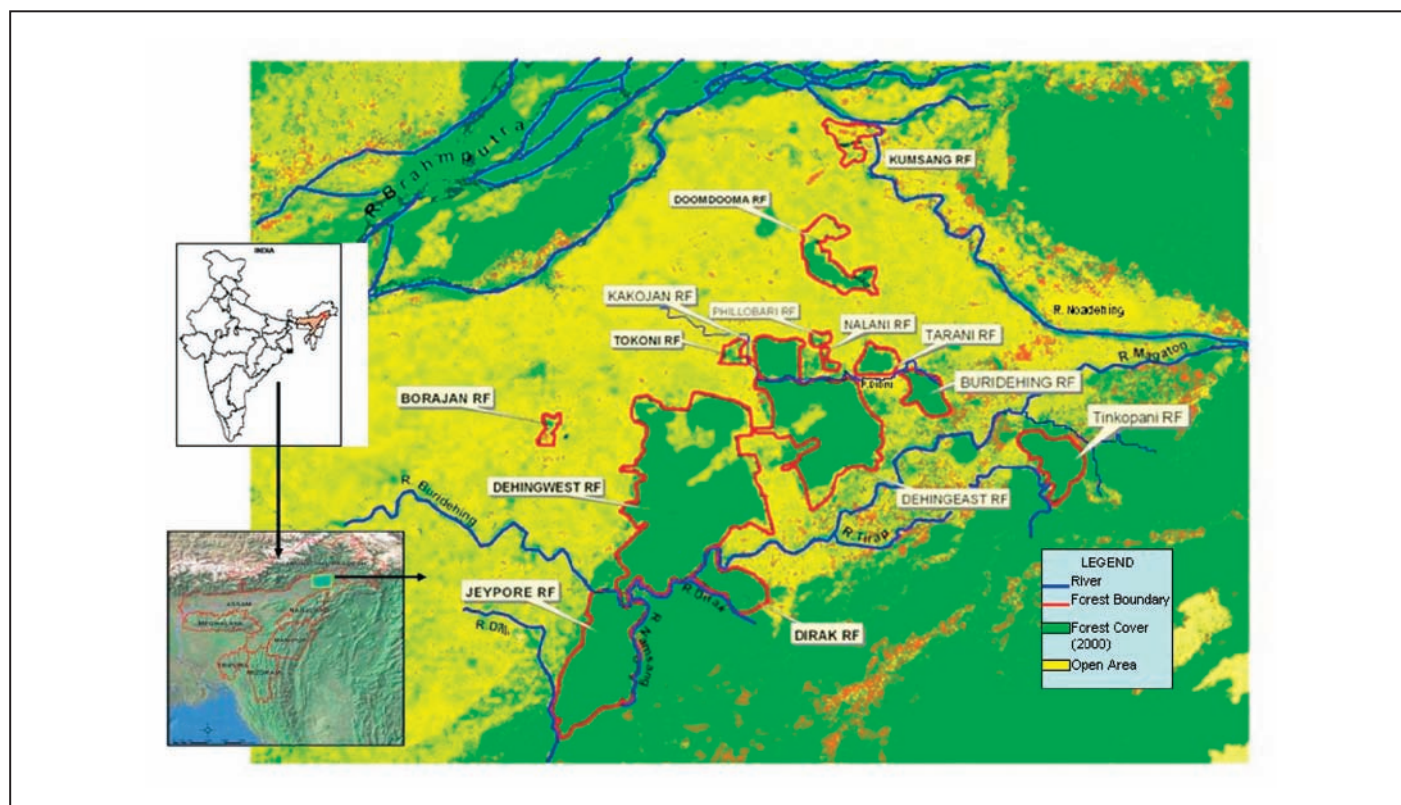


Figure 1. Satellite image (2000) of eastern Assam showing the forest patches surveyed for western hoolock gibbons, *Hoolock hoolock*.

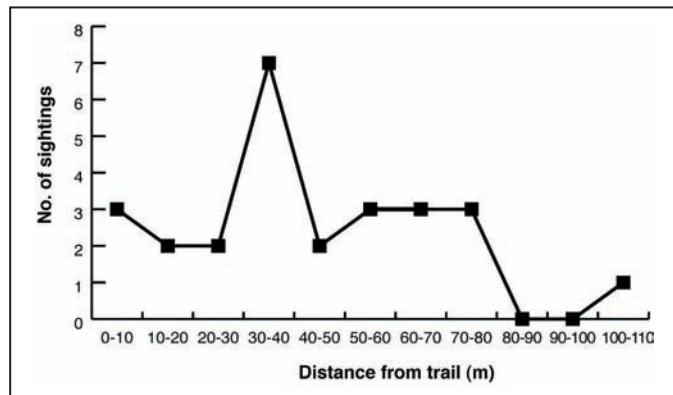


Figure 2a. Detection distance from trail for gibbon groups that were directly sighted from the trail (n=26).

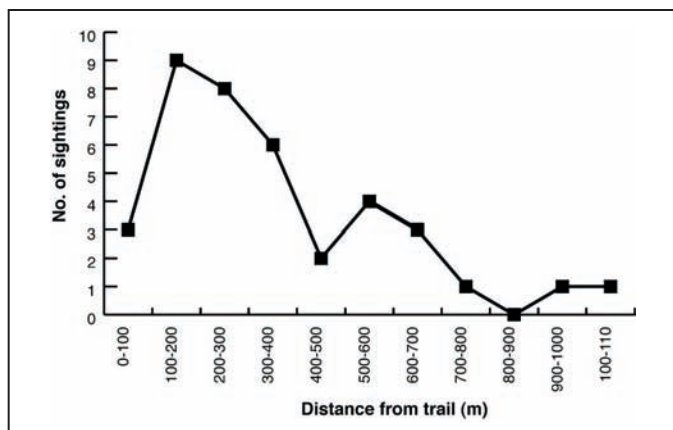


Figure 2b. Detection distance from trail of gibbon groups located by homing in on their calls (n=37).

calls (likely to have originated near the point where a group was encountered or likely to have originated from near where another call had been pin-pointed previously).

Survey of India maps and Assam Forest Department maps of the reserves were used for reference. All routes walked and the locations of all gibbon groups sighted (n=63) were recorded with a hand-held Garmin 12 Global Positioning Systems (GPS) and mapped with the software GPS Trackmaker (Ferreira 2002).

Waypoints along the route were marked on the GPS units at distances of every 140 m on average (n = 1495). We assessed tree-cover in a roughly 20-m-radius at each waypoint, scoring it on a broad scale of four categories: Open—no trees, scrub with ferns, weeds or bamboo; Low—scattered trees, crown of nearest tree to observer not connected to another tree or to only one other tree crown at most; Medium—middle-storey trees, and crown of nearest tree to observer connected to crowns of two or more trees; Good—three-tiered forest with large emergent trees and dense middle storey, and crown of nearest tree to observer connected to other crowns on all sides. The medium and good cover categories were combined for analysis since both classes represented adequate canopy continuity for gibbon movement. Forest cover change at all the sites was analysed from satellite images of the years 1990 and 2000 using the ERDAS Imagine (5th edition, 1999) software.

We used the SPSS 11.0 for Windows software for statistical analysis. Spearman's Rank correlation test was used to test for the relationship between fragment size class and encounter rates. The Kruskal-Wallis One-Way Anova and

Table 4. Western hoolock gibbon, *Hoolock hoolock*, group sizes (n = 54 complete group counts) at each site and in each forest size class.

Site (RF)	At Each Site		In Each Forest Size Class		
	Av. group size	Range	No. of different groups	No. of groups Full counts	Av. group size
					(n=54)
Fragment size class <5 km² (n=4)					
Phillobari	0		2	2	2.5*
Nalani	0				
Tokowani	1.00				
Borajan	4.00				
Fragment size class 20–30 km² (n=7)					
Tarani	3.00		26	24	3.29 ± 0.22
Buridihing	3.42	2–5			
Kumsang	3.00				
Kakojan	3.44	1–4			
Doomdooma	4.00	3–5			
Tinkopani	0				
Dirok	1.50	1–2			
Fragment size class >100 km² (n=3)					
Upper Dehing East	4.14	3–5	31	28	3.96 ± 0.13
Upper Dehing West	3.80	3–4			
Jeypore	3.77	2–5			

*Standard error on this value was not calculated because the number of groups in the sample was inadequate.

Mann-Whitney U test were used to examine differences in encounter rate and group size among the three forest size classes. Means are reported with standard errors.

Results

Encounter rates

Hoolock gibbons were seen in 11 of the 14 reserves surveyed. Two of the three reserves where no gibbons were seen were small, and the third was a medium-sized fragment where just two groups were heard calling during our survey, and we suspect that poaching has wiped out most of the gibbons here.

The encounter rate was significantly correlated with forest size class (Spearman's $r_s = 0.69$, $p = 0.006$, $n = 14$), being lowest in the small fragment size class at 9 groups/100km, intermediate in the medium fragment size class with 23 groups/100km, and highest in the large forest size class ($>100\text{km}^2$) at 58 groups/100km (Table 3). The encounter rates were significantly higher in the large forests than in the small fragments (Mann-Whitney U = 0.00, $Z = -2.14$, $p = 0.032$, $n = 7$).

Group sizes

Group sizes differed among the three forest size classes (Kruskal-Wallis one-way ANOVA, $\chi^2 = 6.67$, $df = 2$, $p = 0.036$, $n = 54$). The smallest mean group size (2.5, $n = 2$) was in the smallest forests, intermediate group sizes (3.29 ± 0.22 , $n = 24$) were in the medium-sized forests, and the largest groups (3.9 ± 0.13 , $n = 28$) were in the largest forests (Table 4). The difference was significant between the medium and large forest size classes (Mann-Whitney U, $Z = -2.416$, $p = 0.016$, $n = 52$). The groups in the small forest fragments were smaller than the large forest groups by more than one individual/group—a potentially important discrepancy for a species with naturally small group sizes (mean 3.8, range 2–5, $n = 6$ [Kakati 2004]) and a long period (6–8 years) to maturity. The composition of each of the 59 groups seen are given in Table 5.

Age and sex ratios

Hoolock gibbon groups in the small fragment size class had the lowest numbers of immature animals (infant, juvenile and sub-adult)-to-adult animals ratio, i.e., only 0.66 immature/adult, whereas in the large forest size class, there were 0.98 immature/adult. The infant-to-adult female ratios were similar among the three forest size classes (Table 6). There were no juveniles in the small fragments surveyed. We combined the juvenile and infant categories into 'young' and re-calculated the young-to-adult female ratio. There was 0.5 young/adult female in the small fragment size class compared to 1.46 young/adult female in the large forest size class, indicating that juvenile survival has been severely affected in the small fragments. The adult male-to-female sex-ratios in the small, medium and large forest size classes were 1:2 ($n = 2$ groups), 1:0.91 ($n = 24$ groups) and 1:1.03 ($n = 28$ groups), respectively.

Minimum estimate of number of groups

The minimum number of hoolock gibbon groups in each forest was calculated based on actual encounters of groups and the most conservative, minimum estimate from the number of other groups heard calling (Table 7). A total of 178 calls were recorded across sites, from which 79 counts were deducted. It was estimated, therefore, that there were at least 99 other gibbon groups at the sites, apart from the 59 groups sighted during the survey. The highest estimate was for Jeypore Reserve Forest (RF) (at least 38 groups), followed by Upper Dehing East Block RF (at least 30 groups). The middle-size (20–30 km^2) category of forests showed high variation, ranging from just two groups in the extensively encroached and degraded Kumsang RF to as many as 25 groups in the relatively less disturbed Kakojan RF. The estimates of two groups from Tinkopani RF and seven from Dirok RF, both bordering the Arunachal Pradesh state border, were unexpectedly low since both areas contained a high proportion of medium/good tree cover suitable for gibbons. Indications were that hunting has lowered the hoolock gibbon populations at these sites. It is possible that gibbons may call less in hunted areas and in sites

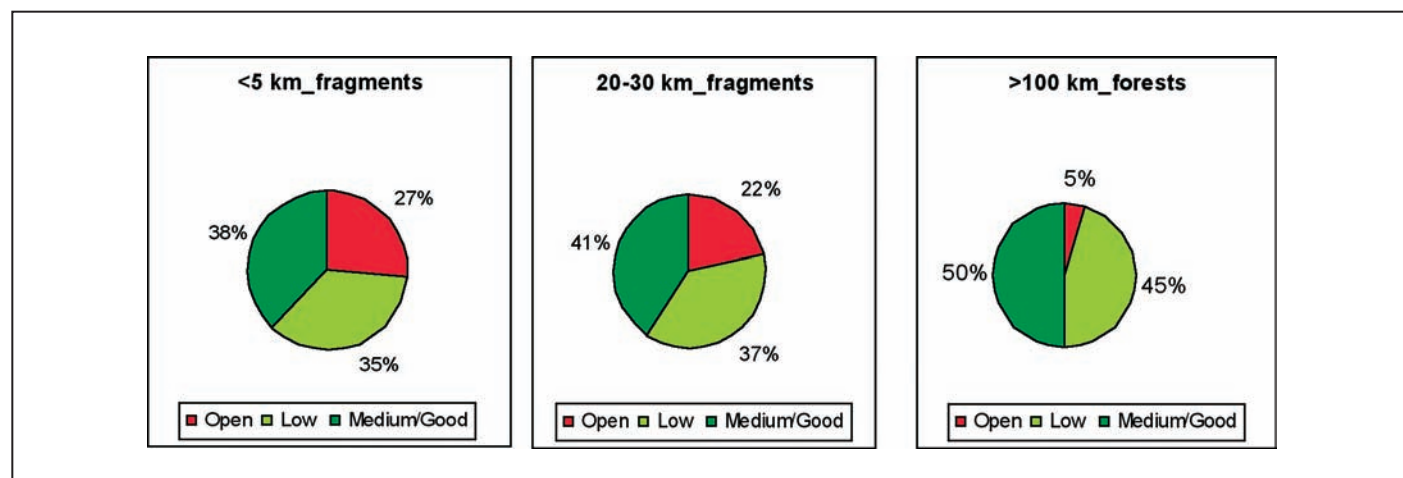


Figure 3. Proportion of points ($n = 1495$) in different tree-cover categories in each forest size class.

Table 5. Locality, group size and group composition of western hoolock gibbons, *Hoolock hoolock*, in eastern Assam (n = 59 groups; complete counts for 54 groups).

		No. of groups	Adult		Sub-adult			Juvenile			Infant	Group size
			Male	Female	Male	Female	?	Male	Female	?		
Forest size class <5 km²												
1	Tokowani	1	-	1	-	-	-	-	-	-	-	1
2	Borajan	1	1	1	-	1	-	-	-	-	1	4
Forest size class 20–30 km²												
3	Tarani	1	1	1	-	-	-	-	-	1	-	3
4	Buridihing	8	1	1	-	-	-	-	-	-		2
5			1	1	-	-	-	-	-	-	1	3
6			1	1	-	-	-	-	1	1	1	5
7			1	1	-	-	-	-	-	1	1	4
8			1	1	-	-	-	-	-	-	-	2
9			-		-	1	-	-	-	1	-	2+
10			1	1	-	-	-	1	-	1	1	5
11			1	1	-	-	-	-	-	-	1	3
12	Kumsang	2	1	1	-	-	-	1	-	-	-	3
13			1	1	-	-	-	-	1	-	-	3
14	Kakojan	9	1	1	1	-	-	-	-	1	-	4
15			1	1	-	-	-	-	1	-	1	4
16			1	1	-	-	-	-	-	1	1	4
17			1	1	-	-	1	-	-	1	-	4
18			1	1	-	-	-	-	-	-	1	3
19			1	1	-	-	1	-	-	-	1	4
20			1	1	-	-	-	-	-	1	1	4
21			1	1	-	1	-	-	-	-	-	3
22			-	-	1	-	-	-	-	-	-	1
23	Doomdooma	3	1	1	-	-	-	-	-	1	-	3
24			1	1	-	1	-	-	-	1	-	4
25			1	1	-	-	1	-	-	1	1	5
26	Dirok	3	1	1	-	-	-	-	-	1	-	3+
27			1	-	-	-	-	-	-	-	-	1
28			1	-	-	1	-	-	-	-	-	2
Forest size class >100 km²												
29	Upper Dehing East	15	1	1	-	-	1	1	-	-	-	4
30			1	1	-	-	-	-	-	1	1	4
31			1	1	-	-	-	-	-	1	-	3
32			1	1	1	-	-	-	-	1	1	5
33			1	1	-	1	-	-	-	1	1	5
34			1	1	-	-	-	-	-	1	1	4
35			1	1	-	-	1	-	-	2	-	5
36			1	1	-	-	-	-	-	1	1	4
37			1	1		1				1	1	5
38			1	1	-	-	-	1	-	1	-	4
39			1	1	-	-	-	-	1	-	-	3
40			1	1	-	-	-	-	-	-	-	2+
41			1	1	-	1	-	-	-	1	-	4
42			1	1	-	-	-	-	1	-	1	4
43			1	1	-	1	-	-	-	-	1	4

Table 5 continued on next page

Table 5, continued

		No. of groups	Adult		Sub-adult			Juvenile			Infant	Group size
			Male	Female	Male	Female	?	Male	Female	?		
44	Upper Dehing West	6	1	1	-	-	-	-	1	-	1	4
45			1	1	-	-	-	-	-	-	-	2+
46			1	1	-	-	-	-	-	1	1	4
47			1	1	1	-	-	-	-	1	-	4
48			1	1	-	-	-	-	-	-	1	3
49			1	1	-	-	-	-	-	1	1	4
50	Jeypore	10	1	1	-	-	-	-	-	-	-	2
51			1	1	-	1	-	-	-	1	-	4
52			-	1	-	-	1	-	-	1	-	3+
53			1	1	-	-	-	-	-	-	1	3
54			1	1	-	1	-	-	1	-	-	4
55			1	1	-	1	-	-	-	-	1	4
56			1	1	-	1	-	-	-	-	1	4
57			1	1	-	-	-	-	-	1	1	4
58			1	1	-	-	1	-	-	1	-	4
59			1	1	-	1	-	-	-	1	1	5
Total	n=11	59	55	55	4	14	7	4	7	32	29	207+

? Sex not identified

Table 6. Ratios of age and sex classes in 54 western hoolock gibbon, *Hoolock hoolock*, groups in the different forest size-classes.

Ratios	Forest Size Class (km ²) (No. of gibbon groups)		
	<5 (2)	20–30 (24)	>100 (28)
Immature (infant, juvenile, subadult): Adult	0.66	0.79	0.98
Infant : Female	0.50	0.52	0.60
Juvenile : Female	0.00	0.76	0.85
Young (Juvenile and Infant) : Female	0.50	1.28	1.46
Male : Female	0.50	1.09	0.96

Table 7. Estimate of minimum number of western hoolock gibbon, *Hoolock hoolock*, groups in each reserve from groups located and from calls only.

Area (RF)	No. of groups seen	No. of groups heard calling, but not tracked down	Minimum Estimate
Forest Size Class <5 km² (n=4)			
Phillobari	0	0	0
Nalani	0	0	0
Tokowani	1	1	2
Borajan	1	0	1
Forest Size Class 20–30 km² (n=7)			
Tarani	1	2	3
Buridihing	8	12	20
Kumsang	2	0	2
Kakojan	9	16	25
Doomdooma	3	4	7
Tinkopani	0	2	2
Dirok	3	4	7
Forest Size Class >100 km² (n=3)			
Upper Dehing East	15	15	30
Upper Dehing West	6	15	21
Jeypore	10	28	38
Total	59	99	158

where shortage of food may influence energy costs of calling. No groups were seen or heard in the two smallest fragments (Phillobari RF and Nalani RF), although a group was later found living in some village trees outside Nalani RF, with no canopy connection to the RF.

Forest cover

All the reserve forests had between 35% and 45% of the points sampled in the low tree cover category, indicating extensive disturbance due to logging in the past or from illegal logging more recently. The small and medium fragments had a high proportion of open areas without trees (22–27%) compared to the large forest (5%), while the largest forest-size class had the highest proportion of medium to good tree cover (50%) (Fig. 3). Assessments of tree cover for each of the reserves are represented graphically in Figure 4.

It is significant that 87% of the gibbon groups were found at medium to good cover locations, when only 42% of the total tree-cover scores fall in this category. No gibbons were seen in the open areas, and only 13% of the sightings recorded were in areas of low tree cover (Fig. 5).

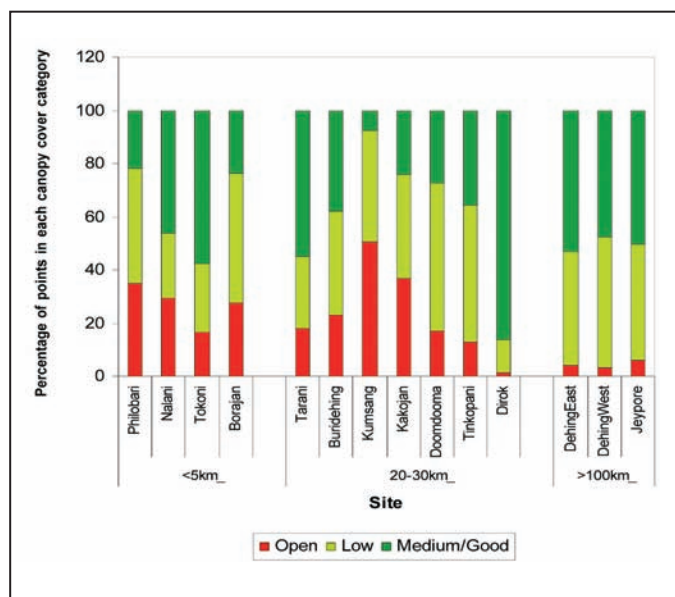


Figure 4. Percentage of points ($n=1495$) in three tree cover categories at each of the 14 sites.

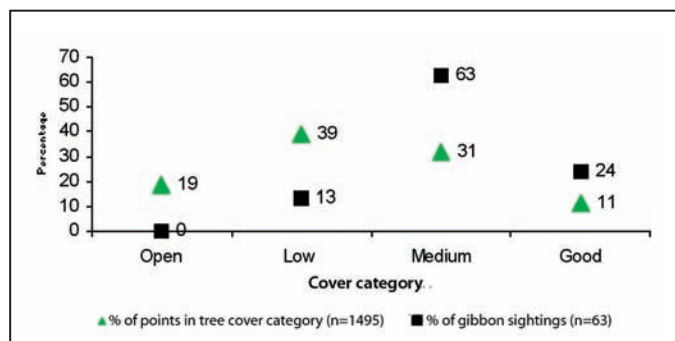


Figure 5. Proportion of gibbon sightings in the different cover classes plotted against the total proportion of points in each cover class.

A comparison of satellite images for the area between the years 1990 and 2000, a span of 10 years preceding the study, showed that the actual areas of forest cover lost in the small, medium and large forests were 0.5 ± 0.4 km², 1.3 ± 0.4 km² and 3.7 ± 1.0 km² respectively (Fig. 6a). As a proportion, however, the figures are 23.7 ± 15.7 , 16.1 ± 11.3 and $2.3 \pm 0.4\%$ for the small, medium and large forests respectively, indicating that the decadal rate of loss has been highest in the smallest fragments (Fig. 6b).

Discussion

Degradation in forest fragments was indicated by decreased canopy continuity, tree density, hoolock gibbon food tree density, and food species richness (Kakati 2004). This loss of food sources and the changes in the habitat structure are strongly implicated in the low encounter rates and smaller group sizes found in forest fragments during this survey.

All gibbons are fruit-pulp specialists (Chivers 1984) with simple stomachs and a very limited ability to digest leaf material. Unlike monkeys, they cannot cope with secondary compounds and toxins in leaves, and they prefer animal to plant protein in times of fruit-shortage (Vellayan 1981). Prolonged dependence on leaves can thus cause severe nutritional stress, especially in the juveniles. Although adults might persist and breed for a time in sub-optimal habitats, a sudden die-off may occur when eventually their nutritional tolerance threshold is breached.

Reductions in fruit supply and species richness of fruit trees due to disturbance and fragmentation have been documented in forest habitats (Johns 1986; Tabarelli *et al.* 1999). Reduced fruit supply in turn has been linked to low population densities and declines of primate frugivores in fragments in Mexico, the Atlantic forest of Brazil, and the Central Amazon (Estrada and Coates-Estrada 1996; Chiarello and Melo 2000; Gilbert and Setz 2001). The survival ability of several other rain-forest primates, including gibbons, has been negatively correlated to its degree of frugivory (Johns and Skorupa 1987). In undisturbed areas, the natural seasonal cycles of fruit shortage in tropical habitats may be offset by the presence of keystone species. For example, one study showed that <1% of the tree species sustained the entire frugivore community through a period of three months of low fruit availability in Cocha Cashu, Peru (Terborgh 1983). In a fragment forest, however, this vital availability might be easily disrupted. Across five intensive study sites in Assam, Kakati (2006) found that during the dry, winter season gibbon groups in all the forests shifted from a predominantly fruit diet to eating leaves. Even then, the gibbon groups in the medium-sized and large forests continued to eat at least 14–28% fruit (as percent time spent feeding), whereas the small fragment groups had practically no fruit at all for two months (January–February). Such extreme shortage of high-energy fruit could be the critical point on which hinges the survival of gibbons in small fragments.

Although the proportion of infants to adult females was similar between forest size classes (i.e., breeding rates were similar), groups in the small and medium-sized fragments had fewer juveniles/adult female than in the larger forests, suggesting that juvenile survival is compromised by fragmentation. Gibbon females in fragments probably suffer very high lactation costs because of inadequate diets in terms of both quality and quantity. Many mammals under conditions of seasonal food shortages are known to rear a reduced litter or females may not survive to breed again (Moir 1994). Weaning is thought to be the critical time that foods of particularly high nutrient density are required (Ofstedal 1991), and hoolock

gibbon infants might be surviving the entire two-year period of dependency on the mother at this high cost to her, only to die when they stop suckling.

Dispersal may be the time of greatest danger of mortality for sub-adults or displaced adults in fragments. Fragmentation causes deterioration of the habitat and the consequent attrition of suitable territories. Also, canopy discontinuity can prevent dispersing gibbons from accessing potentially suitable areas and make them vulnerable to injury or death from falls. Alfred and Sati (1990) recorded the disappearance of hoolock gibbons from 168 forest patches (0.14–2.7 km²) in *jhum* (slash and burn agriculture) matrices in the Garo Hills of

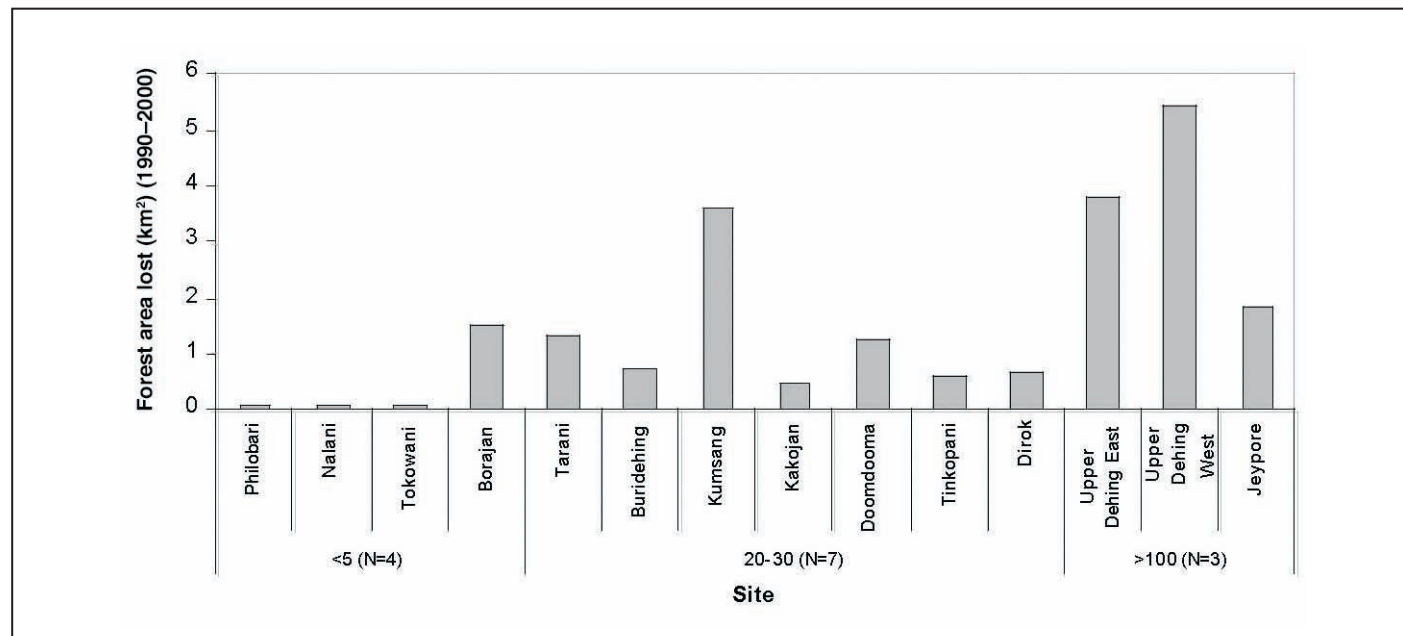


Figure 6a. Area of forest lost (km²) between 1990 and 2000, assessed from satellite images, in the 14 survey sites.

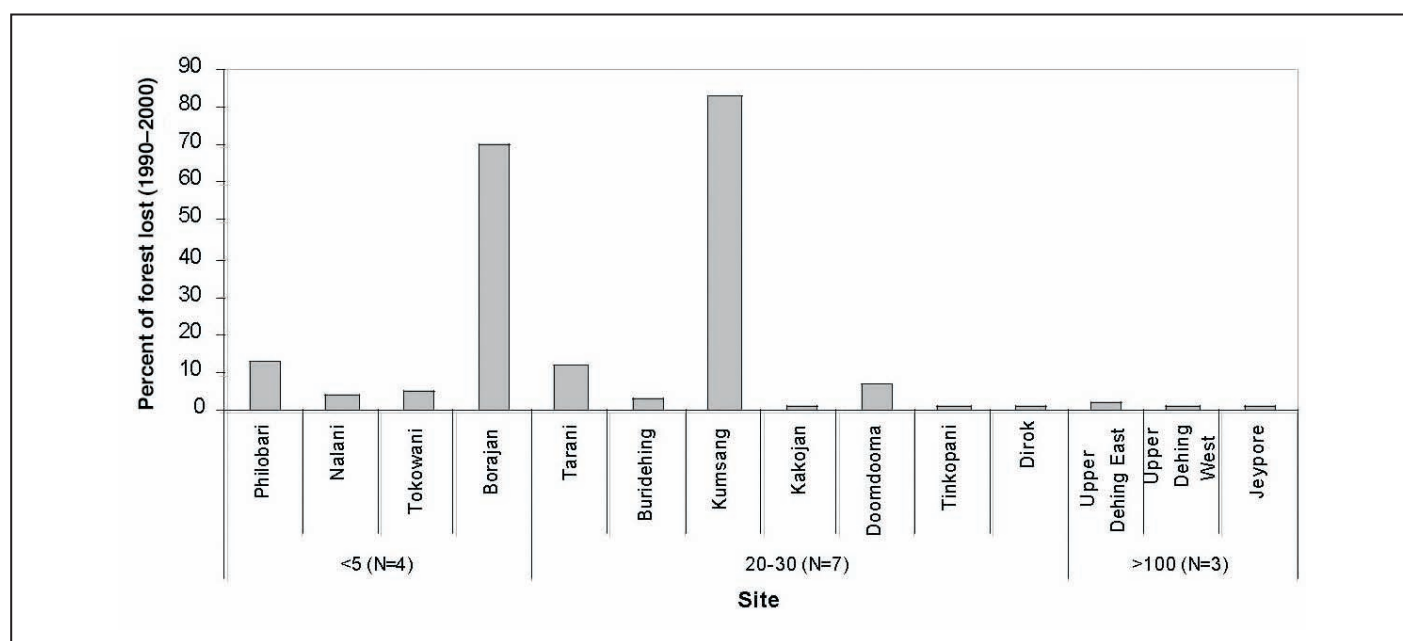


Figure 6b. Percentage of forest cover lost between 1990 and 2000, assessed from satellite images, in the 14 survey sites.

Meghalaya mostly in the 10 years preceding their study, and mainly because *jhum* cycles had shortened to <10 years and gibbon dispersal corridors (secondary forests and old growth bamboo) were no longer available. In the Upper Assam landscape studied, the sharp edges with the settled agriculture matrices leave no scope at all for gibbons to move between sites.

The extensive habitat degradation seen in fragments results from a number of factors. The reserve forests have a long history of logging (>70 years) which, although halted in 1996 by a region-wide ban by the Supreme Court of India, continues illegally and gives the forests little respite to recover. The effects are less clear in the large forests because relative to their size the damage is limited. Edge effects include hotter and drier micro-climates through exposure to the sun and wind and increased tree mortality on the periphery of the fragments (Ferreira and Laurance 1997). There are invasions of weeds and successional species (Laurance *et al.* 1998), reduction in genetic variation and vigour (Cascante *et al.* 2002), and changes in plant communities due to altered predator-prey relationships (Rao *et al.* 2001). Compared to continuous forest, fragments have also been found to have lower seedling abundance and seedling species diversity of the non-pioneer tree species (Benitez-Malvido 1998; Benitez-Malvido and Martinez-Ramos 2003). They also tend to have lower tree densities and tree species diversity and fewer large trees (Menon and Poirier 1996, Kakati 2004). Again, many forest trees depend on frugivores for dispersal and regeneration (Hamann and Curio 1999; Chapman and Onderdonk 1998). The disappearance of frugivores due to human activities such as logging or hunting will invariably affect recruitment of tree species, and the sooner and more severely in fragments.

The occurrence of most of the gibbon groups in areas of medium or good tree cover emphasizes their strong dependence on tall trees and closed canopies, not just for their food, but also in providing for adequate arboreal pathways to move around their home ranges and for protection from predators. Although they may persist in degraded habitats for a time, the outlook for their long-term survival in these areas is grim, as presaged by the series of recent local extinctions reported in Walker *et al.* (2007). One of the most impressive examples of gibbon population declines caused by rapid habitat degradation is in the Borajan fragment, which lost 70% of its forest cover between 1990 and 2000. Its gibbon population declined from 34 animals in 1995 (Srivastava 2006) to just nine in 2000 (pers. obs.).

In conclusion, it is valid to suppose that an intact 5 km² forest fragment can support 15–20 groups of hoolock gibbons, assuming each group had a home range of *c.* 25 ha (the average home range size in relatively undisturbed sites [*n* = 3, Kakati 2004]). Given that fragmented populations of the western hoolock gibbon make up a significant part of the surviving numbers of this endangered species, it is necessary to conserve these populations through forest restoration and the establishment of dispersal corridors wherever feasible. It

is also important to manage the commonly seen canopy disruptions within the forest, due for example, to roads, railways or power lines, by the establishment of canopy bridges, either natural or artificial, at distances of 50–100 m along the gap. Translocation to suitable, available habitats, to areas where they have been hunted out but where hunting can now be effectively prevented, can be a last resort for gibbons in completely degraded fragments.

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A Survey of Miller's Grizzled Surili, *Presbytis hosei canicrus*, in East Kalimantan, Indonesia

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Abstract: Miller's grizzled surili, *Presbytis hosei canicrus*, is the rarest of the four *hosei* subspecies, all endemic to the island of Borneo. From 5 March to 6 April 2008, we carried out a survey to examine the status of this monkey in the eastern part of Borneo, most particularly in Kutai National Park. We were unable to find any evidence for the continued existence of *Presbytis hosei* in the park. We were able to locate just one group occupying the riverbank habitat of Baai River, isolated by oil palm plantations in the District of Karangan, East Kalimantan. Forest fires, particularly in 1997–1998, and hunting for food and for their bezoar stones are probably the main causes of the probable extirpation of *P. hosei* in the Kutai National Park. Forest loss due to extensive oil palm plantations is the main threat to this species in the areas of Sangkulirang and Karangan.

Key words: *Presbytis hosei canicrus*, east Kalimantan, survey, habitat, forest fires, palm oil

Introduction

The grizzled sureli, *Presbytis hosei* (Thomas, 1889), is a Bornean endemic consisting of four subspecies (Brandon-Jones *et al.* 2004). Miller's grizzled surili, *Presbytis hosei canicrus* Miller, 1934, occurs in the northeastern part of Borneo, in Kutai National Park, Mt. Talisayan, and in the Karangan River basin in East Kalimantan, Indonesia (Supriatna and Wahyono 2000). It is now very rare, and has been listed as one of the world's 25 most endangered primates (Brandon-Jones 2006). It has been classified as Endangered by V. Nijman, E. Meijaard and J. Hon (assessors) on the 2008 *IUCN Red List of Threatened Species* (IUCN 2008). This assessment was based on an evident decline in the population due to extensive habitat loss, fragmentation and hunting. The assessors indicated, however, that the geographical boundaries of the range of *P. hosei* are unknown, and future surveys could result in it being reclassified.

Its former stronghold, Kutai National Park, has been largely wrecked, and only 5% of its forest remains. Its destruction was largely due to logging concessions, illegal settlement and massive and widespread forest fires. The population of *P. h. canicrus* believed to occur in this park, however, has never been surveyed (Meijaard and Nijman 2000). Here we report on a survey of this species in the Kutai National Park and other parts of its known geographic distribution.

Methods

From 5 March 2008 to 6 April 2008, we surveyed three sites in Kutai National Park (created in 1982, 198,629 ha)—Prevab (00°31'54.6"N, 117°27'54.0"E), Mentoko (00°34'04.0"N, 117°25'53.0"E), and Melawan (00°30'17.3"N, 117°26'45.5"E). We also traveled along the Sangata River (that forms the northeastern boundary of the park) starting from the town of Sangata lama downstream to the mouth of the river. We surveyed locations outside the Kutai National Park: Baai River from Pengadan village (01°15'10.9"N, 117°45'20.3"E); Karangan River from Perondongan village (01°20'15.1"N, 117°42'14.9"E); and another location in a karst area, north-east of Mt. Beriun (01°09'00.6"N, 117°22'48.8"). These locations are in the Districts of Sangkulirang and Karangan of the East Kutai Regency (Fig. 1). The surveys conducted on foot used line transect methods (Whitesides *et al.* 1988). We walked along existing trails (cut by the research station in Kutai National Park), and in some cases we set up new transects where trails were unavailable. Each transect was 2 to 5 km long. We recorded animal-to-observer distances using a laser rangefinder, and noted the number of individuals, group spread, age-class categories of the individuals seen, their behavior, and their responses to the observer. The surveys were begun at about 07:00 and would continue till about 17:30 h.

We also surveyed rivers that we presumed passed through areas with habitat suitable for *P. hosei*. We surveyed from 06:30 to 09:30 and 16:00–18:00. Our speed was 2–3 km/hour, and the surveys were repeated up to three times on different days. Vegetation types were recorded along the transects, as were any signs of disturbance or sources of threat to the monkeys. We interviewed local people so as to obtain information on the occurrence of *P. hosei*, the people's impressions of their abundance, and to gauge the situation concerning hunting and the threats that the species is facing.

Results and Discussion

Kutai National Park

Rodman (1978) carried out primate surveys in the Kutai National Park (then the Kutai Nature Reserve) in 1970–1971 (1 May 1970 to 31 July 1971) and 1975 (1 July to 24 August). His study site was on the Mentoko River, just south of the Sengata River in the northeast corner of the park (Fig. 1; location 2 in Table 1). In the 1971 study, Rodman (1978) recorded seven *P. hosei* groups (then referred to as *P. aygula*, following

Napier and Napier [1967]) in an area of 2.74 km², estimating a density 2.6 groups/km². From this, and based on a group size of eight, Rodman calculated a density of 20.4 individuals/km². Berenstain (1986) reported on the aftermath of forest fires in 1982–1983; fires that destroyed 3.5 million ha of mainly forested land in East Kalimantan. Only one-quarter of the 3 km² forest at Mentoko remained, but there were still six groups of *P. hosei* there. Azuma (1988) also reported that the species could still be found in Kutai National Park. Fires again ravaged East Kalimantan's forests in 1991/1992 and 1993/1994, and the El Niño-related fires of 1997–1998 destroyed a total of 5.2±0.3 million ha in the region (Siebert *et al.* 2001). Of this about 2.6 million ha of forest was burned, with varying degrees of damage but primarily affecting recently logged forests. By the time the rains started at the beginning of May 1998 almost the entire basin area in the Kutai district had been burned (Hoffmann *et al.* 1999; Siebert *et al.* 2001) and only 5% of the national park remained forested (Meijaard and Nijman 2000). Nijman (2001) failed to locate any *P. hosei* groups in a survey in the eastern part of the Kutai National Park in 2000.

Table.1. Results of the survey for *Presbytis hosei canicrus* in the 9 localities in the districts of Sangkulirang and Karangan in eastern Borneo: 5 March 2008 to 6 April 2008. Also listed are other primates recorded: slow loris (*Nycticebus coucang*), long-tailed macaque (*Macaca fascicularis*), pig-tailed macaque (*M. nemestrina*), proboscis monkey (*Nasalis larvatus*), the silvery leaf monkey (*Trachypithecus cristatus*), Müller's grey gibbon (*Hylobates muelleri*), and the Bornean orang-utan (*Pongo pygmaeus*).

	Locations	Transect length Coordinates	<i>P. hosei</i>	Threats/disturbance	Other primates
1	Kutai National Park - Prefab	5.5 km 00°31'54.6"N, 117°27'54.0"E	No	Forest fires, coal mining, encroachment, illegal logging, settlements, hunting, tourism	<i>M. fascicularis</i> (3 groups) <i>H. muelleri</i> (2 groups seen, 6 groups by calls) <i>P. pygmaeus</i> (9 individuals, 28 nests)
2	Kutai National Park - Mentoko	5.5 km 00°34'04.0"N, 117°25'53.0"E	No	Forest fires, coal mining, encroachment, illegal logging, settlements, hunting, tourism	<i>M. fascicularis</i> (1 group) <i>H. muelleri</i> (5 groups by calls) <i>P. pygmaeus</i> (2 individuals, 17 nests)
3	Kutai National Park - Melawan	10 km 00°30'17.3"N, 117°26'45.5"E	No	Forest fires, coal mining, encroachment, hunting, illegal logging	<i>M. fascicularis</i> (1 group) <i>M. nemestrina</i> (1 group) <i>H. muelleri</i> (2 group seen) <i>P. pygmaeus</i> (2 individuals)
4	Kutai National Park - Sangata River	16.69 km, by boat 00°26'16.1"N, 117°36'46.0"E 00°29'50.5"N, 117°31'37.4"E	No	Fish/shrimp farming, mining, settlements, hunting	<i>M. fascicularis</i> (7 groups) <i>N. larvatus</i> (5 groups) <i>T. cristatus</i> (2 groups) <i>P. pygmaeus</i> (7 nests)
5	Karangan River - Perondongan village	27.8 km, by boat 01°20'15.1"N, 117°42'14.9"E	No	Oil palm plantations, hunting, illegal logging, settlements	<i>M. fascicularis</i> (9 groups) <i>M. nemestrina</i> (4 groups) <i>N. larvatus</i> (9 groups) <i>T. cristatus</i> (2 groups) <i>H. muelleri</i> (1 group by call)
6	Baai River - Pengadan village	35.67 km, by boat 01°15'10.9"N, 117°45'20.3"E	Yes (5 ind.)	Oil palm plantations, hunting, illegal logging, settlements	<i>N. coucang</i> (1 pet) <i>M. fascicularis</i> (9 groups & 2 pets) <i>M. nemestrina</i> (3 groups) <i>N. larvatus</i> (7 groups) <i>H. muelleri</i> (2 groups by call and 1 pet) <i>P. pygmaeus</i> (1 individual, 11 nests)
7	Muara Bulan	01°15'37.2"N, 117°40'39.5"E	No	Oil palm plantations, hunting, illegal logging, settlements	<i>M. fascicularis</i> (3 pets) <i>P. pygmaeus</i> (2 pets)
8	Muara Entaik	01°11'44.9"N, 117°30'03.8"E	No	Oil palm plantations, hunting, illegal logging, settlements	<i>P. pygmaeus</i> (2 nests) <i>M. nemestrina</i> (1 group) <i>M. fascicularis</i> (4 groups)
9	NE Mt. Beriun	2.5 km 01°09'00.6"N, 117°22'48.8"E	No	Oil palm plantations, hunting, illegal logging, settlements	<i>P. pygmaeus</i> (3 individuals) <i>H. muelleri</i> (4 groups by call)

We surveyed three sites in Kutai National Park (Mentoko, Prefab, Melawan) and conducted a river-boat survey along the Sangata River over 20 days. We were unable to find any evidence that *P. hosei* could still be found there. In Mentoko the forests were entirely secondary, recovering after the forest fires. They were dominated mostly by species of *Macaranga* (Euphorbiaceae, and typically pioneer) and shrubs, and showed extensive and frequent clearings. The forests of Prefab and Melawan were also largely destroyed, although small primary forest patches remained, some not entirely burned and some relatively intact. These forest patches still supported other primates such as orangutans, Müller's gibbons, and pigtail macaques (Table 1). Besides the major loss of forest, there was significant disturbance from hunting, as well as forest degradation resulting from mining activities (the biggest coal mining corporation in east Kutai) close to Mentoko. Encroachment, illegal logging, hunting and illegal settlement are serious threats to Kutai National Park's future.

"Berangat" is the local vernacular name of *Presbytis hosei*, although our interviews showed that they can confuse them with other langur species. People we interviewed in Kabojaya village reported that *P. hosei* was frequently hunted until the late 1990s. They would hunt them for food and especially bezoar stones or "batu geliga" (intestinal concretions

valued for traditional medicine, see Nijman 2004). The price of these stones can reach US\$20–30 per gram.

The Sangata River, a refuge for wildlife since the forest fire (Berenstain 1986), has also been largely devastated. There are many illegal settlements and the riparian forest and mangroves along the Sangata River, from the town of Sangata lama to the mouth, have been converted for fish and shrimp farming and for agriculture. We found five groups of "Bekantan" (*Nasalis larvatus*) and also, in fields and plantations even, the silvery leaf monkey (*Trachypithecus cristatus*). Although sparse, vegetation in some areas would appear to still provide sufficient habitat for primates; orangutan nests were found in some places.

Hunting by immigrants that arrived with the logging, oil and coal companies undoubtedly contributed to the rarity of *P. hosei* in the Kutai region towards the end of the 1990s, but we believe that the massive forest fires of 1997–1998 were the main cause of the extirpation of the species in the areas of Kutai National Park that we surveyed. At 198,629 ha, the park is very large (Tresina *et al.* 2005) and of course more surveys are needed, and it is possible that *P. hosei* is still surviving in more remote forest patches.

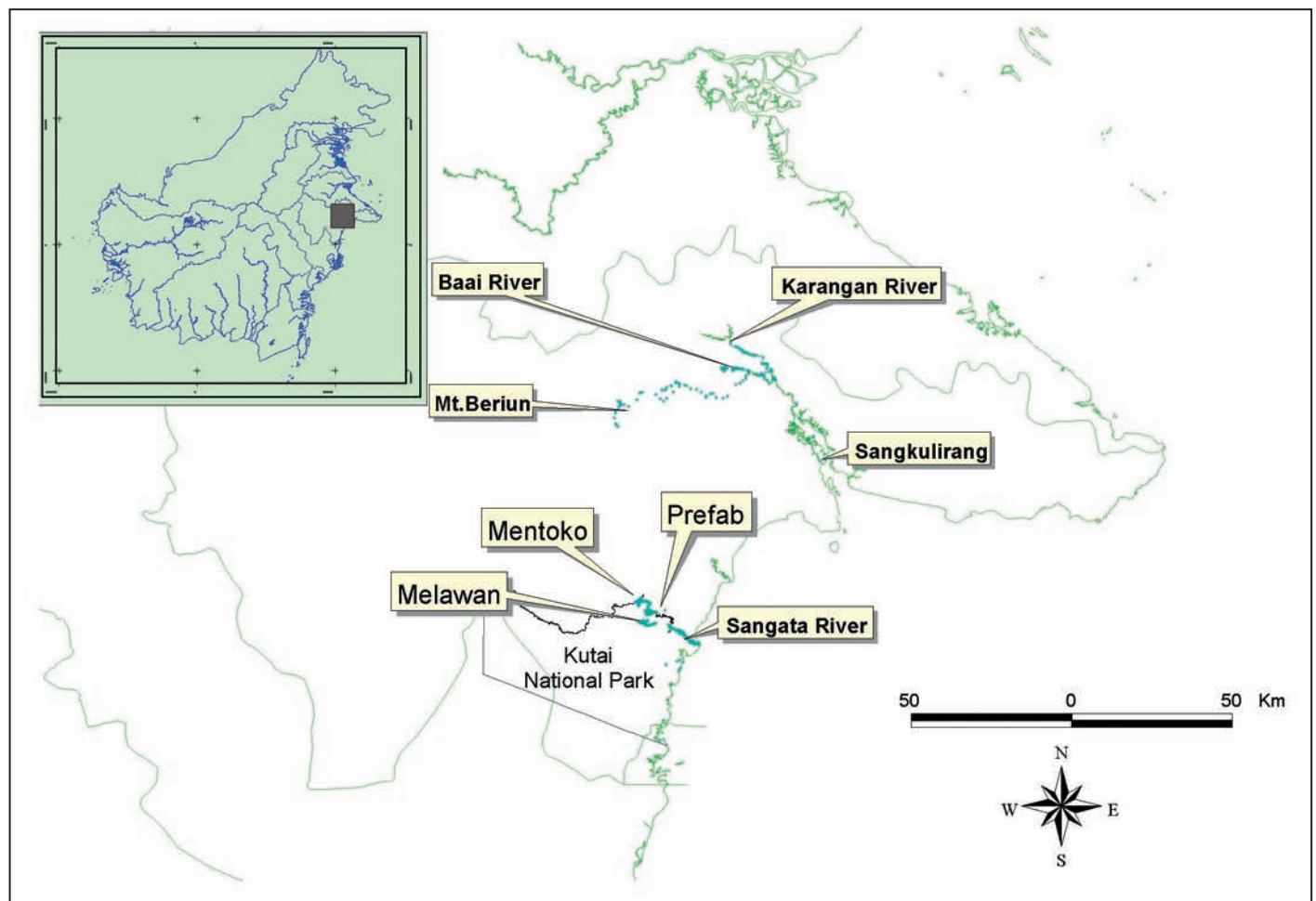


Figure 1. Survey sites for *Presbytis hosei canicrus* in the districts of Sangkulirang and Karangas in eastern Borneo.

Sangkulirang and Karangan

These areas are more than 100 km to the northeast of Kutai National Park. The occurrence of *Presbytis hosei canicrus* there was reported by Supriatna and Wahyono (2000). We were unable to find any forest along the road from Sangata (Kutai National Park) to Sangkulirang bay: the area was dominated by grassland, there is coal mining in the area, and also burgeoning oil palm plantations. Upstream of Sangkulirang River there are two major rivers, the Karangan River and Baai River (also called Pangadan River because it flows through the village of Pangadan). Karst mountains comprise the catchment area of these rivers. We took at least ten days to travel the Baai River (35.67 km), a portion of the Karangan River (27.8 km), and the area in the interior northeast of Gunung Beriun along a logging road belonging to PT. Penambangan logging company.

We found just one group of *P. hosei canicrus*. It was on the banks of the Baai River (01°15'10.9"N, 117°45'20.3"E), near the village of Pangadan. It was composed of five individuals (four adults and a juvenile). They were predominantly grey on the back and whitish on the ventral surface of the tail, on the abdomen and chest up to the neck, and lower face. Black hairs on the upper part of the cheeks and white hairs from the lower lips to the ears give the appearance of two angled lines of long black and white hairs extending back along the side of the face from the mouth to the ears. Their calls are grunt-like and distinct from other members of the genus *Presbytis* in the region (*P. fredericae* and *P. rubicunda*). *Presbytis hosei canicrus* was seen in a small patch of mangrove at the river edge. The vegetation included such as *Rhizophora apiculata*, *Nypa* sp. and *Bruguiera parviflora* near the water, and *Macaranga* sp., *Ficus* sp., and planted mango trees behind the mangrove. There was widespread clearance for oil palm cultivation in the area, and the group was surrounded by plantations. The chances of survival for the group seemed slim. They would evidently need to travel along the ground to reach other forest patches, and in doing so would be susceptible to dogs or to capture for their bezoar stones.

Deforestation is not limited to the lowlands, and extends to the higher elevations such as Gunung Beriun. There the forests have been replaced by *Acacia* and *Gmelina* plantations, and there is legal and illegal logging, and also clear cutting for oil palm plantations. These forests have no protected status, but it is possible that *P. hosei* still exists in remnant patches there. Hunting is evident in this area judging by the number of pets we found (orangutan, gibbon, macaque, and slow loris) (Table 1). Muarabulan villagers informed us that hunting for bezoar stones occurred in the distant past before they became Moslem, but that now the "stone monkey" (*P. hosei*) is so very rare that they believed it could no longer be found in the forests there.

Conclusion

It is quite probable that *Presbytis hosei canicrus* is already locally extinct in Kutai National Park due to the forest fires and hunting. Our surveys in the park and in Sangkulirang and Karangan lead us to conclude that the natural habitat of *Presbytis hosei canicrus* is also disappearing very rapidly due to the expansion of oil palm plantations. Surveys of the remaining areas where it may still occur in east Kalimantan are urgently needed in order to provide for a true assessment of this species which we now believe to be extremely endangered.

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Sequences of Tibetan Macaque (*Macaca thibetana*) and Tourist Behaviors at Mt. Huangshan, China

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Abstract: Previous research on Tibetan macaques (*Macaca thibetana*) at Mt. Huangshan, China, suggested that ecotourism can have detrimental consequences. This study identified sequences of behaviors that typically occur in macaque-tourist interactions to examine whether particular tourist behaviors precipitate monkey responses. Focal sampling was used to record relevant behaviors from tourists and 10 macaques over 28 data collection sessions in August 2006. Data collectors recorded whether each behavior occurred as part of a sequence. Sequences were defined as two or more behaviors in which each behavior occurred within five seconds of the previous behavior. Of 3,129 total behaviors, 2,539 (81.1%) were from tourists and 590 (18.9%) were from monkeys. Tourists initiated significantly more sequences than did macaques (412 [84.6%] versus 75 [15.4%]). Tourist *pointing*, *rail slapping*, *fleeing*, and *rock showing* occurred significantly more than expected in tourist-macaque sequences. *Points* were also among the most common tourist behaviors preceding macaque threats. By discouraging tourists from engaging in these behaviors, macaque threats could be reduced, thereby improving macaque-tourist interactions. These results may aid in the management of other macaque tourist sites to minimize stress-inducing interactions.

Key words: *Macaca thibetana*, tourism, stress, behavior sequences

Introduction

In recent years, anthropogenic ecological changes and increased human populations worldwide have led to heightened opportunities for interactions between human and nonhuman primates. Sponsel (1997) first coined the term *ethnoprimatology* to refer to the interconnections between human and nonhuman primates. In little more than a decade since then, studies in *ethnoprimatology* have become increasingly common (Riley 2006; Wolfe and Fuentes 2007). Many of these studies have examined the effects of primate ecotourism on the species it aims to conserve. Of wild primate populations at tourism sites, perhaps the most studied has been the genus *Macaca*. Macaques have a wide-ranging distribution, spanning East Asia to Northern Africa and Gibraltar, and their home ranges frequently overlap with human habitat and tourist sites (Fuentes 2004). Close contact between humans and macaques can have deleterious consequences for the health of both species. Macaques can transmit simian foamy virus, herpes B virus, simian T cell lymphotropic viruses, and simian retrovirus to humans (Engel *et al.* 2002; Jones-Engel *et al.* 2005; Wolfe *et al.* 2004). In turn, humans can transmit

measles, influenza, and respiratory pathogens to macaques (Jones-Engel *et al.* 2001). Human-macaque interactions can also lead to heightened intragroup aggression in macaques, injury to both macaques and humans, and missed or negative educational experiences for humans (Zhao and Deng 1992; Berman and Li 2002; Berman *et al.* 2007). Macaque food provisioning has potentially harmful consequences as well (see, for example, Southwick *et al.* 1976; Sugiyama and Ohsawa 1982; Zhao and Deng 1992).

Hsu *et al.* (2009) studied interactions between Formosan macaques (*Macaca cyclopis*) and tourists at a nature park in Taiwan. Illegally provisioned food significantly increased the duration and frequency of aggressive interactions between the two species. Agonistic behaviors were involved in 16.4% of overall interactions, with adult males of both species as the age/sex class most likely to engage in these behaviors. Human-initiated interactions exceeded macaque-initiated interactions by 2.44:1.

O'Leary and Fa (1993) found some similar patterns when they examined the effects of tourists on the behavior of Barbary macaques (*M. sylvanus*) in Gibraltar. Tourist-initiated

interactions outnumbered macaque-initiated interactions by 3.2:1. Furthermore, macaques adjusted their diurnal activity patterns based on tourist visitation routines and were much more sedentary compared to their unhabituated counterparts, most likely due to frequent food provisioning. More recently, Fuentes (2006a) found significantly more contact than non-contact interactions between Barbary macaques and humans, as well as a high proportion of interactions and food provisioning involving taxi drivers and tour guides.

Fuentes (2006b) also compared human-macaque interactions at sites in Gibraltar and Bali. In Bali, humans interact with long-tailed macaques (*M. fascicularis*) primarily at Hindu temple sites. Tourists are more frequently bitten by long-tailed macaques in Bali than by Barbary macaques in Gibraltar. There are also higher rates of macaque-macaque aggression as well as macaque-human aggression in Bali than on Gibraltar, which Fuentes attributes to both the contexts of interactions and to species-specific differences between *M. sylvanus* and *M. fascicularis*. At both Gibraltar and Bali, adult male macaques are overrepresented in interactions while adult females are underrepresented. Fuentes *et al.* (2007) used qualitative assessment techniques to compare these same two sites. They emphasize the importance of incorporating human variables into assessments of macaque behavior and ecology. Political, cultural, and economic factors at both sites play relevant roles in macaque behaviors and macaque-human interactions. These factors affect the types of interactions that occur, as well as human attitudes toward the macaques and the potential for disease transmission. Fuentes and colleagues underscore the importance of considering the needs of all stakeholders when developing management strategies to maintain macaque ecotourism sites. Indeed, Loudon *et al.* (2006) noted substantial differences in human attitudes toward long-tailed macaques across 11 different Hindu temple sites in Bali. These attitudinal differences are most likely due to varying cultural and economic conditions between the sites. Their results highlight the importance of evaluating each macaque-human interaction site individually before developing management practices.

Additional studies have focused solely on interactions between long-tailed macaques and humans in Bali. Wheatley and Harya Putra (1994) reported that the macaques were positively reinforced for aggressive behaviors toward tourists via food handouts. The speed and intensity of aggressive macaque behaviors were positively correlated with the quantity and quality of available food. Wheatley and Harya Putra also found a positive correlation between the frequency of redirected monkey-macaque aggression and the presence of provisioned food. Patzschke *et al.* (2000) found potentially deleterious consequences related to food-provisioning in long-tailed macaques. Monkeys spent twice as much time near humans and ate five times as much anthropogenic food on days with high versus low levels of food provisioning. Macaques spent less time on the ground and showed more social behaviors when away from the presence of tourists. Fuentes and Gamelr (2005) examined interactions between

tourists and long-tailed macaques, reporting that adult and subadult male macaques were involved in more aggressive interactions with humans than expected by chance, whereas adult females and immatures were involved in fewer. Adult male humans received more aggressive behaviors than expected by chance, and human female children received less. While long-tailed macaques in Bali appear to be afforded some protection through their interactions with humans, these interactions may not be sustainable. Increased tourism, changes in patterns of land use, increased pesticide use, and the potential for disease transmission may eventually lead to a less stable environment for the macaques (Fuentes *et al.* 2005).

In Singapore, long-tailed macaques are significantly more likely to interact with humans if food is present rather than absent (Fuentes *et al.* 2008). Although adult humans typically provision food to the macaques, significantly more children were present at feeding times than at nonfeeding times, suggesting that food provisioning is influenced by children urging their parents to feed the monkeys, or by parents attempting to entertain their children. These results suggest that, similar to other sites, food may be strongly involved in perpetuating macaque-human interactions. Levels of contact interactions and aggression are relatively low between macaques and humans at this site, largely because interactions typically take place along roadsides, where humans throw food from cars. Additionally, unlike at many other interaction sites, Singapore has an education program to minimize contact interactions between macaques and humans and sometimes enforces fines and penalties for doing so (Fuentes *et al.* 2008).

Tibetan macaques (*M. thibetana*) interact with tourists at two sites in China: Mt. Emei and Mt. Huangshan. Tibetan macaques at Mt. Emei often rob visitors of food and other possessions (Zhao and Deng 1992). Visitors are sometimes injured by the macaques, with ten human deaths in a period of eight years as an indirect result of macaque interactions. Zhao and Deng concluded that close interactions were most likely to lead to aggressive encounters, and that visitor behaviors involving food-carrying and submission were most likely to provoke such encounters. More recently, Zhao (2005) assessed data from Mt. Emei and concluded that a combination of classical and operant conditioning procedures have led to aggressive interactions between macaques and humans. The macaques have been classically conditioned to associate tourists with food, and have been shaped through operant conditioning to beg, approach, and use aggression to obtain food. Zhao suggests placing restrictions on food carrying and tourist-macaque interactions at Mt. Emei.

Long-term data from a group of Tibetan macaques at Mt. Huangshan, China, also indicate that ecotourism may negatively impact macaques. Berman *et al.* (2007) reported that the group displayed heightened signs of disturbance, such as increased aggression and infant mortality, as a function of range restriction for tourism purposes. Ruesto (2007) found a significant positive correlation between the frequencies of macaque threats and tourist behaviors directed at macaques. Matheson *et al.* (2006) found that monkeys' threats were

usually directed from adults to juveniles and from juveniles to humans, possibly due to redirection. Self-directed behaviors in this group were positively correlated with tourist density in an area of the home range in close proximity to tourist platforms (Matheson *et al.* 2007). Additionally, grooming bouts among these macaques were significantly more frequent when tourists were present rather than absent. Tourist density showed a positive trend with regard to female grooming, and was positively correlated with self-grooming by adult males (Mack *et al.* 2008).

Although these data demonstrate a relationship between overall frequency of tourist behaviors and macaque threats, the causal relationship between them has remained unclear. The aim of the present study was to clarify the order of events that typically occurs during macaque-human interactions at Mt. Huangshan.

Methods

Subjects and Study Site

Data were collected over 28 sessions from 17–26 August 2006 at Mt. Huangshan, Anhui Province, China (30°07'09"N, 118°09'41"E; elevation 1,841 m). Mt. Huangshan is a tourist site in east-central China (see Fig. 1). The middle and lower elevations of the site support mixed evergreen and deciduous forests that are home to several groups of Tibetan macaques (*Macaca thibetana*).

Yulingkeng A1 (YA1), the group observed for this study, has been studied by Chinese researchers since 1986, and subjected to tourism since 1992 (see Fig. 2). All YA1 subadult and adult macaques (three adult males, five adult females and two subadult males) served as the focus of this study. Infants and juveniles were excluded, since they do not display the full range of species-typical social behaviors under study, and are difficult to identify reliably (Kutsukake and Castles 2001). Data were collected from a tourist viewing platform located in the macaques' home range (see Fig. 3).

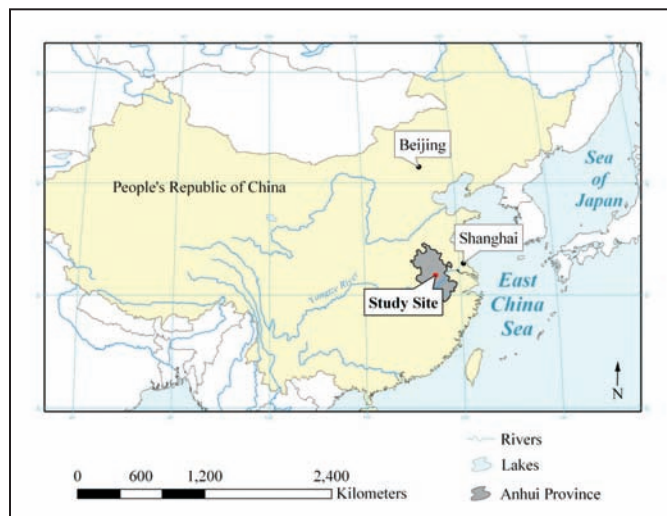


Figure 1. Location of study site. Map by Lucy A. Ruesto, using ArcGIS (v.8).



Figure 2. *Macaca thibetana* adult male, adult female, and infant. Photo by Lucy A. Ruesto.

Table 1. Ethogram of human tourist behaviors.¹

Behavior	Description
Barbed Wire Shake	Tourist shakes the barbed wire that borders the viewing platform railing.
Dangle	Tourist dangles food, body parts, or objects over the viewing platform railing toward macaques.
Flee	Tourist turns and runs away from macaques.
Foot Noise	Tourist stamps feet or kicks wall in observation area.
Hand Noise	Tourist makes noises with one or both hands.
Mimic	Tourist mimics facial expressions and/or body language of a macaque threat, e.g., eyebrow raise, stare, and ground slap.
Mouth Noise	Tourist makes noise with mouth directed toward macaque.
Show Rock	Tourist pretends to throw rock at macaques.
Point	Tourist points at macaques, with arm extending toward macaques' feeding area.
Railing Slap	Tourist slaps rail or post in observation area, which may be done with hands or objects.
Spit	Tourist spits into macaque area.
Throw Food	Tourist drops or throws food item into the macaque area, or directly to a macaque.
Throw Object	Tourist drops or throws non-food item into macaque area (includes rock).
Wave	Tourist waves at macaque. Can be done with hands or objects.

¹Derived from Ruesto (2007) and data collectors' observations in this study.

Procedures

Data collectors achieved 100%, 92% and 95% interobserver reliability for macaque identity, macaque behaviors and tourist behaviors, respectively, prior to beginning data collection. Table 1 shows an ethogram of human behaviors, and macaque behaviors were those described by Berman *et al.* (2004). In each session, data collectors selected a focal animal opportunistically based on their visibility and orientation toward a tourist. Two data collectors recorded the occurrence of any ethogram-defined behaviors, with one data collector speaking the macaque behaviors aloud while the second recorded these and the behavior of the relevant tourist(s) so that the sequence was preserved. A sequence was defined as a string of behaviors in which each behavior occurs within five seconds of the previous behavior. Data collection ceased when the macaques left the area and were no longer visible from the viewing platforms. Data collectors attempted to distribute focal observations equally across macaques; however, the opportunistic nature of data collection meant that some macaques were observed more than others, or contributed more behaviors to sequences. To ensure equal representation under different conditions, observation sessions were distributed across the day, as well as during times of high and low

tourist density. Research methods were approved by the Institutional Animal Care and Use Committee and the Human Subjects Research Committee of Central Washington University.

Results

In all, 3,129 behaviors were recorded; 2,539 (81.1%) were performed by tourists and 590 (18.9%) were performed by macaques. The most commonly observed (macaque and tourist) behaviors overall were *points*, *waves*, and *facial threats*. Of macaque behaviors, 387 (65.6%) were performed by adult females, 151 (25.6%) by subadult males, and 52 (8.8%) by adult males. Overall, 250 behaviors (8.0%) occurred singly and 2,879 (92.0%) occurred in sequences.

A total of 487 sequences were observed. These ranged in length from 2 to 109 behaviors, with two-behavior sequences occurring most frequently ($n = 108$). Of these sequences, 343 (70.4%) involved only human behaviors, 117 (24.4%) involved a combination of human and macaque behaviors, and 27 (5.1%) involved only macaque behaviors. There were significantly more tourist-only sequences than tourist-macaque and macaque-only sequences ($\chi^2 = 326.6$, $p < .005$). Sequences involving both human and macaque behaviors



Figure 3. Macaques and tourists at the provisioning site. Photo by Maureen S. McCarthy.

consisted of significantly more behaviors than tourist-only and macaque-only sequences ($M = 9.56$ behaviors vs. 4.75 and 4.89, respectively; $F(2, 484) = 13.24$, $p < .001$; see Fig. 4). Humans initiated significantly more sequences than did macaques (412 [84.6%] vs. 75 [15.4%]; $z = 15.4$, $p < .005$; see Fig. 5).

The prevalence of tourist behaviors involved in tourist-only sequences and in tourist-macaque sequences were compared. *Flee* ($n = 13$, $z = -2.19$, $p < .05$), *point* ($n = 1,588$, $z = -2.35$, $p < .05$), *show rock* ($n = 15$, $z = -2.34$, $p < .05$), and *railing slap* ($n = 137$, $z = -3.59$, $p < .05$) all occurred more frequently than expected in tourist-macaque sequences. The behaviors following the occurrence of tourist *point* and *railing slap* in sequences were examined. *Flee* and *show rock* were excluded from this analysis due to their low total numbers of occurrences (13 and 15, respectively). For *point*, another *point* (72.6%), *wave* (11.9%), and *railing slap* (2.3%) were the most common tourist behaviors to directly follow. The most frequent macaque behavior to follow a tourist's *point* was a *facial threat* (3.4%). For *railing slap*, another *railing slap* (33.3%), *point* (18.9%), and *wave* (11.7%) were the most common behaviors overall to directly follow. The most fre-

quent macaque behavior to follow a *railing slap* was a *lunge/ground slap* (10.8%).

The most common macaque behaviors were also examined to determine which overall behaviors directly preceded them. For *facial threats*, another *facial threat* most often preceded it (34.2%), followed by a tourist's *point* (18.7%) or a *lunge/ground slap* (14.6%). For *lunge/ground slap*, another *lunge/ground slap* commonly preceded it (33.9%), as well as a *facial threat* (28.7%) or a tourist's *point* (9.2%). For *charge*, a tourist's *point* commonly preceded it (32%), as well as a *facial threat* (22%) or *lunge/ground slap* (18%). Finally, *full grins* were commonly preceded by another *full grin* (30.4%), as well as a *facial threat* or a tourist's *point* (each 17.4%). All other macaque behaviors occurred less than 10 times overall.

Discussion

The most prevalent behaviors by far were those performed by tourists. Not only did they perform the most behaviors, but they also initiated far more sequences than macaques. Despite this, sequences containing both tourist and macaque behaviors were significantly longer than those containing only tourist or macaque behaviors. This suggests that true interactions occurred, not just independent behaviors from each species. Tourist behaviors prompted macaque behaviors and vice versa, thus extending interactions. *Point* and *railing slap* were the tourist behaviors that occurred more than expected in tourist-macaque behavioral sequences. That these behaviors were most commonly followed by another occurrence of the same behavior suggests that tourists repeated behaviors in an attempt to elicit macaque responses. Tourist behaviors such as *show rock* and *throw object* may be considered more intensely threatening to monkeys but occurred much less frequently, possibly because they more effectively elicited frightening macaque responses.

Although *points* and *railing slaps* were most commonly followed by additional tourist behaviors, macaque threats sometimes followed. These macaque threats demonstrate a meaningful pattern based on the human behavior preceding them. A *facial threat* was the most common macaque behavior to follow a tourist's *point*, but a *lunge/ground slap* was the most common macaque behavior to follow a tourist's *railing slap*. Based on its noise component and abruptness, a *railing slap* may be considered a more intense tourist behavior than a *point*, and thus not surprisingly was more commonly followed by a more intense macaque behavior, namely a *lunge/ground slap* as opposed to a *facial threat*. In contrast, a *point*, while occurring very frequently, may be relatively benign and thus result more commonly in a milder response from the macaque, a *facial threat*. However, that a macaque's *charge* was most commonly preceded by a tourist's *point* suggests that the macaques can sometimes be aggressively provoked by *point*.

Data collectors observed anecdotally that some tourist behaviors varied widely in intensity levels while still falling within the operational definitions that had been assigned

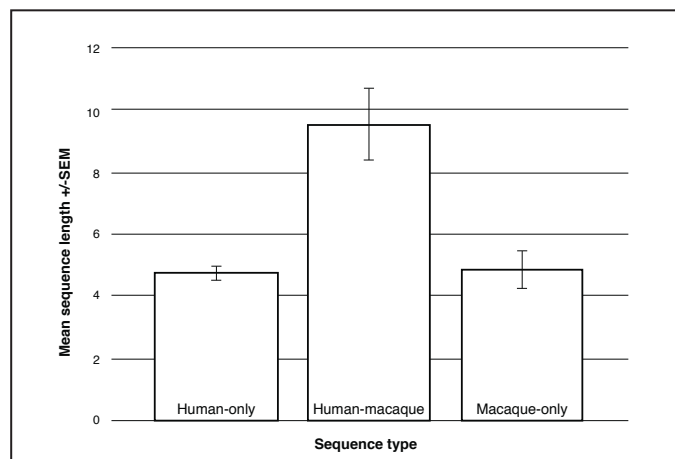


Figure 4. Mean sequence length by sequence type.

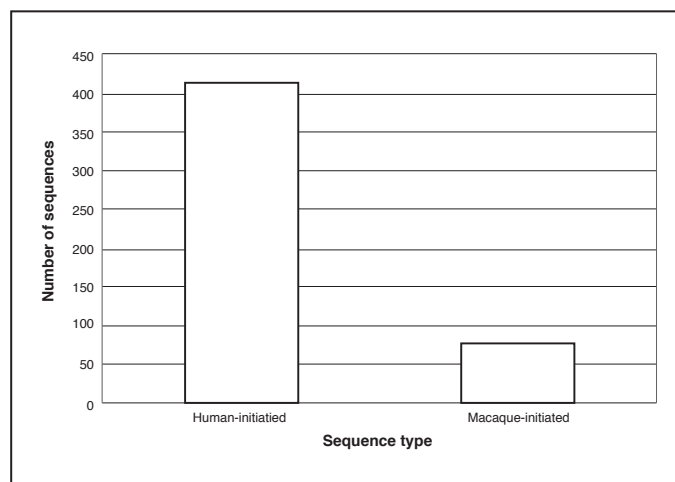


Figure 5. Sequence initiations.

a priori. For example, *points* that broke the plane of the viewing platforms and extended over the railings and into the macaques' feeding area appeared more likely to evoke threats than those that occurred further back on the platforms and away from the feeding area. Behaviors involving an auditory component, such as *hand noise*, *foot noise*, *mouth noise*, and *railing slap* appeared to vary in their likelihood to evoke macaque behaviors based at least partly on the noise level accompanying them. Similarly, Ruesto (2007) reported a significant positive correlation between decibel levels produced by tourists and the frequency of macaque threats.

The current study suggests directions for future research as well as recommendations for tourism management. For example, since *points* and *railing slaps* occur significantly more than expected by chance in human-macaque sequences, and since *points* commonly preceded the most prevalent macaque threats, reduction or elimination of these behaviors could result in a significant reduction in macaque threats. If a reduction in the occurrence of two simple but frequent tourist behaviors could indeed result in a reduction in macaque threats, macaque-tourist interactions could be significantly improved. The long-term consequences of such improvements for macaque well-being and tourist education could potentially be great. In addition, if tourist behavioral adjustments successfully result in the reduction of macaque threats and aggression, this information could be shared with the many other ecotourist sites worldwide where humans and macaques closely interact.

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Back cover: The silky sifaka (*Propithecus candidus*) in Marojejy National Park, northeastern Madagascar, November 2009. This species has been on the list of the world's 25 most endangered primates since 2000. They are hunted and suffer from habitat loss (slash-and-burn agriculture, logging and cutting for fuelwood). The Marojejy National Park and the Anjanaharibe-Sud Special Reserve protect the majority of the remaining populations of this species (see the report by Erik Patel, page 11). Photograph kindly donated by Kevin Schafer/www.kevinschafer.com.

