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ORIGINAL PAPER

The effects of land-use policies on the conservation of Borneo's endemic *Presbytis* monkeys

David A. Ehlers Smith

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Abstract Despite supporting the highest biodiversity on Earth, tropical rainforests are undergoing intensive economic development. In particular, the island of Borneo has lost over 56 % of original lowland forest to resource extraction, fires, and illegal logging. Its rainforests contain 16 primate species, which serve as excellent 'umbrella' taxa for conservation initiatives. The genus *Presbytis* (Colobinae) is well represented across Borneo by four endemic species (*P. chrysomelas*; *P. frontata*; *P. hosei*, and *P. rubicunda*), but remains relatively understudied. Using ecological niche modelling, I calculated the distributions of the 12 Bornean *Presbytis* subspecies; evaluated habitat loss between 2000 and 2010, and examined the current land-use policies across remnant distributions. Subspecies experienced a mean 12.7 % (N = 12 sp.) habitat reduction over the 10 year period. 12.5 % of all habitats were allocated for conversion to oil palm and industrial tree plantations, while logging concessions accounted for a mean 26.3 % across distributions. While the current protected area networks encompassed an average 33.4 % of distributions, most PAs are underfunded, degraded and threatened by logging and mining operations. I therefore recommend priority gazetting of unallocated lands to new PAs within the distribution of *Presbytis chrysomelas* and *Presbytis hosei sabana*, which have experienced the highest forest loss in the last 10 years (22–50 %) and are critically endangered. Logging concessions appear to be at least as effective in maintaining forest cover as PAs and have the economic advantage for effective management, but may have detrimental effects to

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D. A. Ehlers Smith
Orangutan Tropical Peatland Project, Centre for the International Cooperation in Sustainable Management of Tropical Peatlands, Universitas Palangka Raya, Palangka Raya 73112, Central Kalimantan, Indonesia

D. A. Ehlers Smith
Oxford Brookes University, Gipsy Lane, Headington, Oxford OX3 0BP, UK

D. A. Ehlers Smith (✉)
75, Furnace Lane, Nether Heyford, Northamptonshire NN7 3JS, UK
e-mail: dehlersSmith@outrop.com

monkey populations. I recommend an urgent assessment of the effects of selective logging practices on species' persistence, and further recommend population surveys to quantify the populations of critically endangered and data deficient subspecies.

Keywords Borneo · Colobinae · Distributions · Forest loss · Protected areas

Introduction

Tropical rainforests are undergoing intensive economic development (Carlson et al. 2012; Hansen et al. 2008; Page et al. 2002), despite supporting the highest biodiversity on Earth (Pimm and Sugden 1994). The island of Borneo in particular has lost more than 56 % of its original lowland forest, due to resource extraction, fires, and illegal logging (Curran et al. 2004; Geist and Lambin 2002; Langner et al. 2007; Miettinen et al. 2011). Since the 1980s, ~80 % of the forests of Kalimantan (Indonesian Borneo) alone have been allocated as federally-managed industrial timber-logging concessions (Curran et al. 2004). Additionally, in the last 10 years, the rate of forest clearance for oil-palm production has increased by 212 % to account for 31,640 km^2 of Kalimantan's total area; 90 % of such clearances occurred on forested land (Carlson et al. 2012).

Borneo also represents one of the global top 25 biodiversity hotspots under threat (Myers et al. 2000) and is home to 16 primate species, ten of which are endemic (Groves 2001; Munds et al. 2013). Given the high rate of habitat loss, much action has been taken to conserve the Bornean orangutan (*Pongo pygmaeus*), including efforts to stabilize populations and moratoriums on logging throughout known distributions (Soehartono et al. 2007), and comprehensive studies on its ecology (e.g. Morrogh-Bernard et al. 2009; Wich et al. 2008), distribution (Husson et al. 2009) and conservation status (Wich et al. 2012). Such actions in turn serve as useful conservation surrogates for biodiversity within orangutan distributions (Bibby 1998; Margules et al. 2002).

Indeed, in addition to protection for their own sake, the primates, as a large, well-studied (Rowe 1996) and charismatic order (Mittermeier et al. 2012), make excellent 'umbrella' species for conservation initiatives (Soemarna et al. 1995; Kiester et al. 1996; Bibby 1998; Sowa et al. 2007). Primates of the genus *Presbytis* are more extensively distributed than the orangutan across Borneo, and are represented by 4 endemic species: *P. chrysomelas*; *P. frontata*; *P. hosei*, and *P. rubicunda*, but remain relatively understudied (but see Ampang and Zain 2012; Davies et al. 1988; Marshall 2010; Nijman 2010; Ehlers Smith and Ehlers Smith 2013). Locality data for *P. chrysomelas* and *P. frontata* in particular are scarce, and the taxonomic status of the genus is still disputed and in a state of flux (c.f. Groves 2001; Brandon-Jones et al. 2004; Brandon-Jones 2006; Meyer et al. 2011; Table 1).

Basic knowledge of a species' occurrence and distribution is an essential starting point for predicting extinctions as a result of habitat loss, and to subsequently devise strategies to focus conservation efforts (Margules and Pressey 2000; Groves 2003). Three types of species occurrence data exist: point localities, at which a species has been observed; extent of occurrences, as broad geographic ranges within which are all the localities that a species is recorded; and predicted distributions, in which the suitability of environmental conditions within a known extent of occurrence are assessed for the likelihood of species'

Table 1 The taxonomy (after Groves 2001), IUCN Red List Threat Status and habitat requirements (Boitani et al. 2006) of the *Presbytis* monkeys of Borneo

Species	Authority	Status	Habitat[a]	Altitudinal limit (m asl)
P. chrysomelas chrysomelas	Müller 1838	CR A2cd	TWE, PS, FS	500
P. chrysomelas cruciger	Thomas 1892	CR A2cd	TWE, PS, FS	500
P. frontata	Müller 1838	VU A2cd	TWE	2,000
P. hosei canicrus	Miller 1934	EN A4cd	TWE	1,700
P. hosei everetti	Thomas 1892	VU A2cd	TWE	1,700
P. hosei hosei	Thomas 1889	DD	TWE	1,700
P. hosei sabana	Thomas 1893	EN A4cd	TWE	1,700
P. rubicunda carimatae	Miller 1906	DD	TWE, PS	N/A
P. rubicunda chrysea	Davis 1962	DD	TWE, PS	2,000
P. rubicunda ignita	Dollman 1909	LC	TWE, PS	2,000
P. rubicunda rubicunda	Müller 1838	LC	TWE, PS	2,000
P. rubicunda rubida	Lyon 1911	LC	TWE, PS	2,000

[a] *TWE* tropical wet evergreen forests on mineral soils, *PS* peat-swamp forest on regularly flooded soils, *FS* fresh-water swamp forests on regularly flooded soils (Boitani et al. 2006; Miettinen et al. 2011)

occupancy (Corsi et al. 1999; Guisan and Zimmermann 2000; Rondinini et al. 2006). If environmental and habitat requirements are known, ecological niche models can refine extent of occurrence data by omitting areas that are unsuitable (Rondinini et al. 2005, 2006), as these data representations often overestimate species' distributions (commission errors). In species for which an abundance of locality data-points are available, more sophisticated predictive niche modelling techniques are known to perform well, in which computer algorithms predict species' distributions based on a variety of environmental variables at multiple locations (e.g. Maximum Entropy, Phillips et al. 2006).

The high rate of habitat destruction, agricultural expansion and multiple land-use policies on Borneo is impetus for assessing the anthropogenic threats faced by the endemic and scarcely-documented *Presbytis* monkeys. In this study I aim to (1) present the distributions of the 4 species of Bornean *Presbytis* monkeys, and their respective subspecies, based on ecological niche modelling and a consensus of the literature; (2) evaluate the proportion of habitat loss for each subspecies over the 10 year period between 2000 and 2010; (3) calculate the current land-use policies throughout remnant distributions, and investigate the likely effects of these on the conservation status of each subspecies, and (4) make recommendations for conservation strategies to safeguard the persistence of each subspecies.

Methods

Study species

Genus *Presbytis* is a monophyletic taxon comprising at least 11 species restricted to the Greater Sunda Islands and the Malayan Peninsula (Meijaard and Groves 2004; Meyer et al. 2011). The genus is included in the colobine sub-family, which is distinct in the adaptation of its forestomach morphology, which facilitates folivory and granivory (seed eating; Chivers 1994; Kay and Davies 1994). *Presbytis* monkeys are among the most frugivorous

and granivorous colobines, and typically consume between 25 and 65 % fruit parts (Davies et al. 1988; Marshall 2010; Nijman 2010). Body sizes are relatively gracile and range from 5.6 to 8.2 kg (Rowe and Myers 2010). The genus is almost entirely arboreal, and, with the exception of *P. potenziani*, maintains groups of single-adult males and multiple-adult females and their offspring. Extra-group males form all-male bands or range alone (Davies and Oates 1994).

Presbytis chrysomelas

Presbytis chrysomelas was recently elevated from a subspecies of *P. femoralis* (Groves 2001), and now comprises two of its own subspecies (Table 1). It has the most restricted distribution of the *Presbytis* monkeys and inhabits lowland tropical wet evergreen and swamp forests below 500 m above sea level (asl) in groups of 3–13 individuals (Ampang and Zain 2012; Boitani et al. 2006; Nijman et al. 2008a; Table 1). *P. chrysomelas* is considered critically endangered due to its low population size, restricted distribution and the high conversion rate of its habitat (Nijman et al. 2008b; Table 1).

Presbytis frontata

Presbytis frontata is monotypic (Groves 2001; Table 1) and inhabits tropical wet evergreen forests on mineral soils in a broad distribution across the centre of the island at relatively low densities up to an elevation of 2,000 m asl (Meijaard and Nijman 2008; Table 1). It lives in relatively small groups of >6 individuals (Nijman 2001) and is classified as vulnerable due to hunting and habitat loss (Meijaard and Nijman 2008). *Presbytis frontata* is particularly cryptic and under-studied, and is known to freeze upon contact with humans (Nijman and Nekaris 2012).

Presbytis hosei

The taxonomy of *Presbytis hosei* is under dispute and may actually contain up to three separate species (c.f. Nijman and Meijaard 2008a; Nijman 2010), but for simplicity it is treated here as one species comprising four subspecies (Groves 2001; Table 1). *Presbytis hosei* occurs in tropical wet evergreen forests on mineral soils up to 1,700 m asl in the north and east of Borneo. Above this altitude, densities become much lower (Nijman 2010). Average group sizes range from 7 to 10 individuals, at variable densities from >1 to over four groups km^{-2}. The threat assessment for the respective subspecies ranges from vulnerable to endangered, as a result of hunting and habitat loss (Nijman et al. 2008a; Table 1). *Presbytis hosei canicrus* is noteworthy as its status and distribution is currently under review. Lhota et al. (2012) confirmed the species' persistence in the Wehea Forest of the West Kutai district, and Brandon-Jones (1997) established that its range extends west beyond this region. Setiawan et al. (2009) suggested that the species may be absent in the Kutai National Park, its former population stronghold in the south of its range.

Presbytis rubicunda

Presbytis rubicunda represents the most comprehensively-studied of the endemic *Presbytis* monkeys (c.f. Supriatna et al. 1986; Davies et al. 1988; Davies 1991; Marshall 2010; Hanya and Bernard 2012; Ehlers Smith and Ehlers Smith 2013; Ehlers Smith et al. 2013a,

b), contains five subspecies, and is considered the least threatened due to its broad distribution (Nijman and Meijaard 2008b; Table 1). It occurs at a range of densities, dependent on the availability of high-quality foods (Marshall 2010; Ehlers Smith and Ehlers Smith 2013), and typically comprises 3–10 individuals. *Presbytis rubicunda* occupies tropical swamp and wet evergreen forests on mineral soils up to 2,000 m asl, although it is likely populations above 700–800 m asl are at such low densities that they are non-viable (Marshall 2010).

Species' distribution modelling

Given the state of flux that exists within the taxonomy of the genus, historical locality records assigned to the subspecies treated here are of dubious accuracy and practicality. Furthermore, sufficient locality data were lacking for the subspecies to perform advanced predictive niche modelling. Therefore, I used ecological niche modelling on the known extent of occurrence data-sets for each primate subspecies, which represents a consensus of the available locality data.

I used ArcGIS v10 (ESRI 2011) for all modelling. I accessed primate species' extent of occurrence data from "All The World's Primates" (Rowe and Myers 2010), land cover maps of Insular Southeast Asia from 2000 to 2010 produced by the CRISP project (Miettinen et al. 2011), and elevation data from the World Database on Protected Areas Consortium (WDPA 2006). I extracted by attribute the appropriate forest cover classes from the land cover maps of 2000 and 2010 for each subspecies according to its habitat requirements, as listed in the Southeast Asian Mammal Databank (Boitani et al. 2006; Table 1) which represents a consensus of the literature. For each subspecies I merged all habitat classes as appropriate, and then clipped the resultant layer by its recorded altitudinal limit. I then clipped each extent of occurrence layer by its species-specific habitat layer to produce a distribution layer, from which I then erased all rivers that bisected habitats as natural geographic barriers. I projected all data layers into the appropriate WGS 1984 UTM Zones of Borneo to facilitate accurate area calculations. The dissolve and multi-part to single-part functions were used to assign a unique identification to each polygon, and I then calculated the area of each polygon and subsequently deleted those smaller than 10 km^2 from the distribution model as too small to be a viable habitat patch.

Two of the 12 subspecies treated here warrant specific treatment: given the uncertainty of the distributional boundary of *P. h. canicrus*, I performed two boundary scenarios thus: (1) a minimum scenario based on the data provided by Lhota et al. (2012), and (2) a maximum scenario based on the data provided by Brandon-Jones (1997). The interim distributional region between the known boundary of *P. h. everetti* and *P. h. canicrus* is ascribed to an "unconfirmed" subspecies that is thought to exist in the area and may be one or the other subspecies (Nijman 2010). Similarly, taking into account the likelihood that populations of *P. rubicunda* are non-viable above 700–800 m asl (Marshall 2010), I performed two ecological niche model scenarios for each subspecies for whom distributions exceeded 700 m asl thus: (1) the maximum altitudinal limit recorded (2,000 m asl), and (2) the viable altitudinal limit (700 m asl).

Land use projections

To assess the impacts of the current land-use allocations on the distributions of the 12 *Presbytis* subspecies, I used the data-set compiled by Wich et al. (2012) detailing the current major land uses including: (1) oil palm plantations (IOPP); (2) industrial tree

plantations (ITP); (3) logging concessions, and (4) the protected area (PA) networks. A comprehensive overview of these land-use allocations and how the data were compiled can be found in Wich et al. (2012). I overlaid each subspecies' distribution with each land-use data-layer to calculate the area and proportion that each allocation contributed to each overall distribution (Fig. 1).

Industrial oil palm plantation concessions (IOPP)

Oil palm plantation concessions are granted by the Indonesian and Malaysian governments at the local level for the conversion of natural forests and subsequent production of oil palm monocultures. In Malaysia, a minority of concessions are allocated from de-gazetted commercial forest reserves; the majority are issued in 'conversion' or 'production' forests, as is the case for Indonesian concessions. These data were unavailable for the Malaysian state of Sabah, and as such are not included in the analysis of land-use impacts on subspecies' whose ranges (*P. rubicunda chrysea, P. hosei sabana*) are located within that state (Wich et al. 2012).

Industrial tree plantation concessions (ITP)

The conversion of natural forested lands to ITPs is not considered deforestation in the United Nations Framework Convention on Climate Change, as they are legally defined as 'forest' (Sasaki and Putz 2009; Wich et al. 2012). In Indonesia, ITP are granted by the Ministry of Forestry and converted on lands classified as 'production forests'; in Malaysia the Forestry Department grants concessions on the equivalent 'commercial forest reserves' (Wich et al. 2012).

Logging concessions in natural forests

Logging concession licenses permit companies to extract natural timber products from the rainforests, although mandates of sustainable resource extraction prohibit deforestation through clear-felling (Wich et al. 2012).

Protected areas (PAs)

The gazetting of natural habitats into national parks; nature and wildlife sanctuaries; game and virgin jungle reserves; protection forests, and recreational parks prohibits logging and degradation of PAs (Wich et al. 2012). However, in reality, deforestation occurs in a large proportion of PAs on Borneo (Nellemann et al. 2007; Gaveau et al. 2013).

Results

Taken at the specific level, *P. rubicunda* was distributed across the largest range of the *Presbytis* monkeys found on Borneo (>278,000 km^2; Fig. 2), followed by *P. frontata* (>160,000 km^2) and *P. hosei* (~130,000 km^2). *Presbytis chrysomelas* had the most restricted distribution (~21,000 km^2). *Presbytis rubicunda carimatae*, found only on the adjacent island of Karimata, had the smallest distribution at the subspecific level (130 km^2). Of the mainland

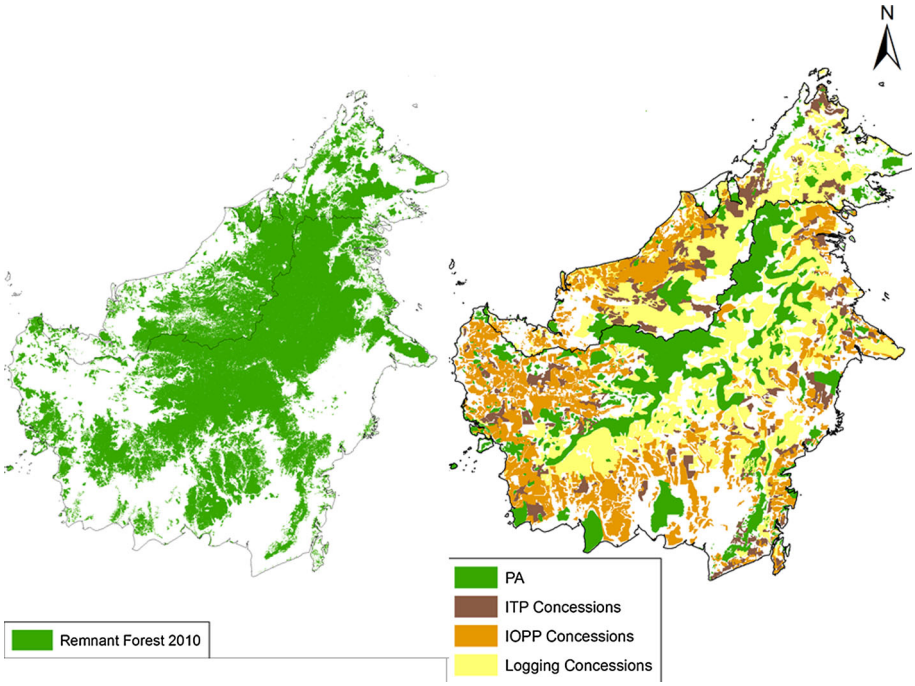

Fig. 1 Forest cover remaining on Borneo in 2010 (Miettinen et al. 2011) and the current land-use allocations, including Protected Areas (PA); Industrial Tree Plantations (ITP); Industrial Oil Palm Plantations (IOPP), and Logging Concessions (Wich et al. 2012)

subspecies, *P. chrysomelas cruciger* and *P. hosei hosei* occupied the smallest area (3,780 and 5,790 km^2, respectively; Fig. 3). *Presbytis r. rubicunda* (>98,000 km^2), *P. r. rubida* (>92,000 km^2) and *P. r. ignita* (~66,000 km^2) had the largest distributions (Table 2; Fig. 3).

Each of the 12 subspecies lost natural habitat through forest conversions between 2000 and 2010 (range 4.6–53.7 %) with the exception of *P. r. carimatae*, whose range is entirely gazetted as a PA (Table 2; Fig. 4). A third of the subspecies lost over 10 % of their natural habitat over the 10 year period. *Presbytis chrysomelas cruciger* experienced the highest rate of habitat loss (>50 %) habitat since 2000. The subspecies located in Sabah (*P. r. chrysea*, *P. h. sabana*) also lost substantial proportions of forested habitat in the 10 year period (21.1 % and 22.8 %, respectively; Table 2).

For most subspecies, logging concessions represented the largest allocation of land-use throughout distributions, followed by PAs (Table 2). Unallocated land accounted for >10 % of all allocations in 10 subspecies, and in the case of *P. h. hosei* as much as 50 % of land within its distribution was unallocated. For the majority of subspecies, ITPs were a minor allocation throughout distributions. In Central Kalimantan and Sarawak, IOPPs also comprised a substantial proportion of land use, but were not assessable for Sabah (Table 2).

The overall distributions of *P. rubicunda* under 700 m asl were substantially lower in most cases, resulting in a general increase in the proportion of economic land-use allocations within distributions, and a complete decrease across all subspecies in PAs (with the exception of *P. r. carimatea*, for whom the entire distribution is protected), as much of the

Table 2 The area of occupancy of all *Presbytis* ssp. on Borneo as derived from land cover maps from 2000 and 2010 (Miettinen et al. 2011), and the current proportions of land-use allocations on 2010 distributions (Wich et al. 2012)

Species	Area of occupancy 2000	Area of occupancy 2010	Occupancy reduction (%)	Current IOPP concessions	Current ITP concessions	Current logging concessions	Current PAs (%)	Unallocated land
P. c. chrysomelas	18,860	17,190	8.9	1,540 (**9.0 %**)	190 (**1.1 %**)	3,630 (**21.1 %**)	6,890 (**40.1**)	4,940 (**28.7 %**)
P. c. cruciger	8,160	3,780	53.7	1,030 (**27.3 %**)	100 (**2.7 %**)	470 (**12.4 %**)	1,050 (**27.7**)	1,130 (**29.9 %**)
P. chrysomelas	27,020	20,970	22.4	2,570 (**12.2 %**)	290 (**1.4 %**)	4,100 (**19.6 %**)	7,940 (**37.9**)	6,070 (**28.9 %**)
P. frontata	171,220	160,280	6.4	5,310 (**3.3 %**)	8,570 (**5.4 %**)	59,880 (**37.3 %**)	55,250 (**34.5**)	31,270 (**19.5 %**)
P. h. canicrus MIN	8,600	7,200	16.3	490 (**6.8 %**)	400 (**5.6 %**)	3,680 (**51.1 %**)	1,100 (**15.3**)	1,530 (**21.2 %**)
P. h. canicrus MAX	12,710	11,340	11.8	580 (**5.1 %**)	800 (**7.1 %**)	6,120 (**54.0 %**)	1,760 (**15.5**)	2,080 (**18.3 %**)
P. h. everetti	59,340	55,570	6.4	2,480 (**4.4 %**)	6,000 (**10.8 %**)	23,010 (**41.4 %**)	16,490 (**29.7**)	7,590 (**13.7 %**)
P. h. hosei	6,240	5,790	7.2	230 (**4.0 %**)	710 (**12.3 %**)	460 (**7.9 %**)	1,450 (**25.0**)	2,940 (**50.8 %**)
P. h. sabana	28,380	21,920	22.8	830 (**3.8 %**)	1,770 (**8.1 %**)	13,250 (**60.5 %**)	4,100 (**18.7**)	1,970 (**8.9 %**)
P. hosei	142,760	129,820	9.1	5,500 (**4.3 %**)	9,410 (**7.2 %**)	55,810 (**43.0 %**)	36,970 (**28.5**)	22,130 (**17 %**)
P. r. carimatae	90	130	0	0	0	0	130 (**100**)	0
P.r. chrysea	26,710	21,080	21.1	N/A	1,770 (**8.4 %**)	13,530 (**64.2 %**)	3,660 (**17.4**)	2,100 (**9.9 %**)
P. r. ignita	69,150	65,980	4.6	2,780 (**4.3 %**)	6,080 (**9.2 %**)	22,460 (**34.0 %**)	20,340 (**30.8**)	14,320 (**21.7 %**)
P. r. rubicunda	106,580	98,980	7.1	4,130 (**4.2 %**)	2,120 (**2.1 %**)	40,410 (**40.8 %**)	32,440 (**32.8**)	19,880 (**20.1 %**)
P. r. rubida	105,710	92,210	12.8	11,980 (**13.0 %**)	4,200 (**4.6 %**)	26,510 (**28.7 %**)	26,570 (**28.8**)	22,950 (**24.9 %**)
P. rubicunda	308,240	278,380	9.7	18,890 (**6.8 %**)	14,170 (**5.1 %**)	102,910 (**36.9 %**)	83,140 (**29.9**)	59,270 (**21.3 %**)

N. B. The total figures for *P. hosei* include the area of occupancy ascribed to the unconfirmed species (Fig. 3)

Numbers in bold refer to the percentage of respective land-use policy on the total distribution

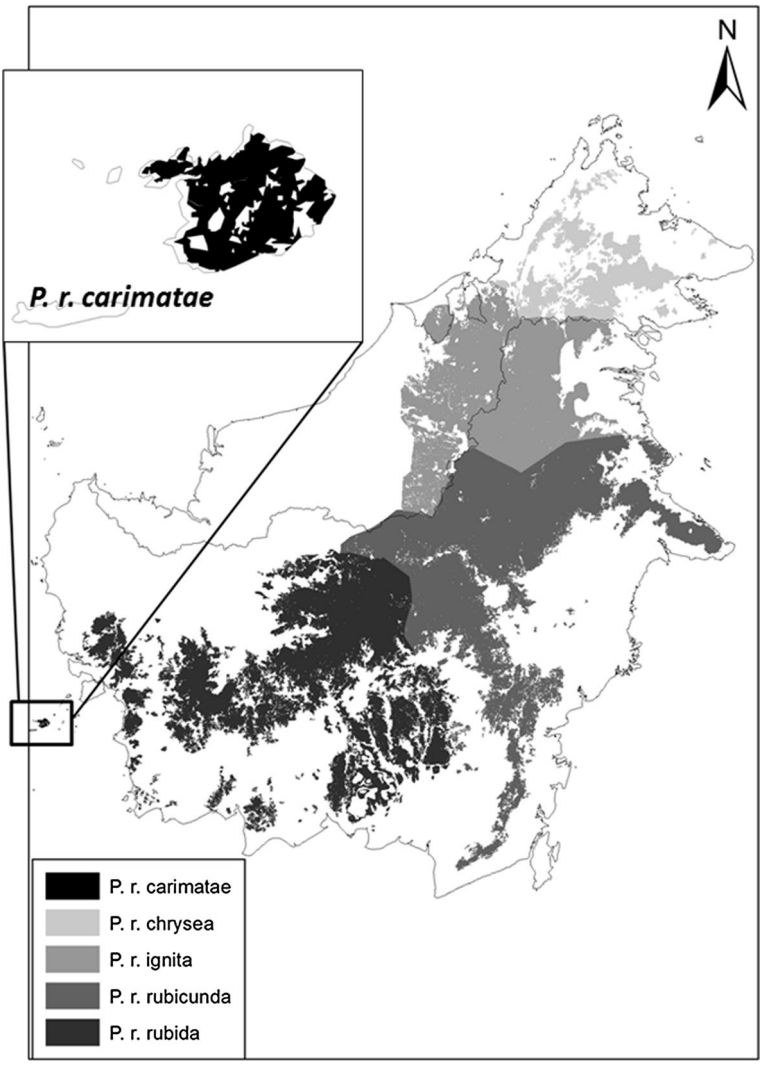

Fig. 2 Distribution of all *Presbytis rubicunda* subspecies

PA network spans the central mountainous region (Table 3; Fig. 1). The maximum scenario for modelling the distribution of *P. h. canicrus* increased the area by $\sim$4,000 km^2.

Discussion

Distribution model caveats

The distributional modelling presented here represented a consensus of the known locality data and environmental and habitat requirements of the *Presbytis* monkeys on Borneo, and is the most comprehensive description of their distributions to date. Given the lack of locality data and the under-studied nature of the subject species, it is likely that both

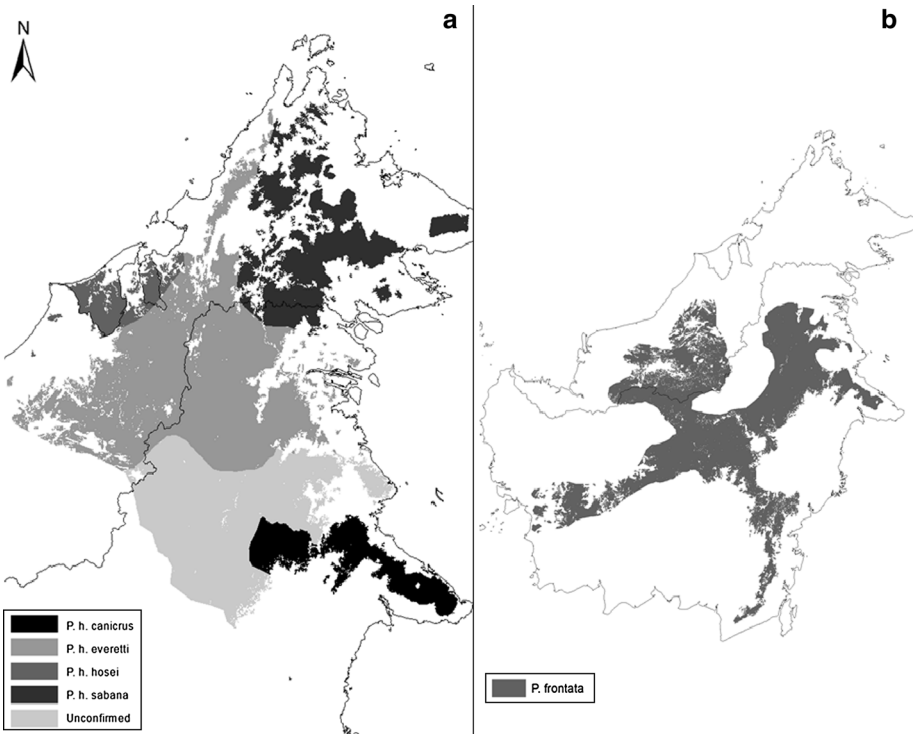

Fig. 3 Distribution of (**a**) all *Presbytis hosei* subspecies including an unconfirmed subspecies between the known ranges of *P. hosei everetti* and *P. hosei canicrus*, based on the maximum scenario (Brandon-Jones 1997) and (**b**) Distribution of *P. frontata*

commission and omission errors featured in the model (Rondinini et al. 2006). Commission errors may have occurred within areas of occupancy that have not been surveyed, despite the removal of habitat known to be unsuitable. For example, large Bornean mammals exhibit irregular distributional patterns with large gaps in apparently suitable habitats, as a result of mutual exclusion through intra-genus competition and hunting pressures (Meijaard et al. 2005). Similarly, it is possible omission errors occurred in unsurveyed regions that lie beyond species' known geographical boundaries. However, given their understudied nature and the rapid habitat destruction occurring throughout their ranges, such an assessment of the anthropogenic threats facing Bornean *Presbytis* monkeys is urgently required and potential errors in the modelling process may be justified to this end.

Land-use impacts on the distribution of Bornean *Presbytis* monkeys

Oil palm plantations and ITPs have replaced vast tracts of lowland forest and fragmented remnant forest blocks (Carlson et al. 2012). Given their arboreal nature (Davies and Oates 1994), *Presbytis* monkeys cannot persist in deforested and converted habitats, and are unlikely to be able to cross denuded patches to disperse. Thus, land conversions for agricultural expansion likely pose the largest threat to the persistence of *Presbytis* monkeys.

As clear-felling within logging concessions is prohibited, it is possible that with effective management, concessions can contribute to species' persistence and appear to be

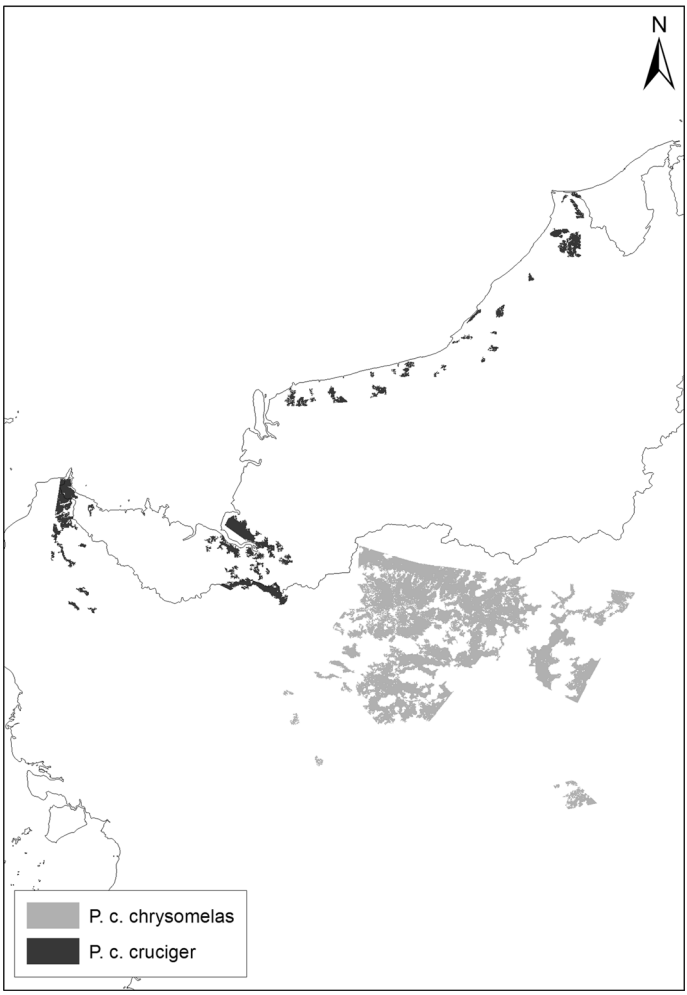

Fig. 4 Distribution of all *Presbytis chrysomelas* subspecies

as successful as PAs in preventing total forest clearance (Putz et al. 2012; Wich et al. 2012; Gaveau et al. 2013). However, the actual response of Bornean *Presbytis* monkeys to the effects of logging within concessions is largely uncertain. For example, *P. rubicunda* has been variously reported as "neutral" (Bennett and Dahaban 1995) and intolerant to logging (Meijaard et al. 2008), with declines in population density as a result (Johns 1992; Blouch 1997). The tallest trees with the largest DBHs, which are targeted for selective logging practices, are important within the ecology of *P. rubicunda* and are used for feeding (Ehlers Smith et al. 2013a), predator avoidance (Nijman and Nekaris 2012) and sleeping sites (DA Ehlers Smith, unpublished data).

Johns and Skorupa (1987) predicted that a primate species' ability to persist in disturbed habitats increases with its degree of folivory, although the most important factor in resistance to disturbance appears to be the ability to exploit leaves when, as a result of logging activities, fruits are scarce (Johns 1986; Meijaard et al. 2005). In this way, *Presbytis* monkeys have an

Table 3 Viable area of occupancies and land-use allocations within distributions of *Presbytis rubicunda* ssp. below 700 m asl

	Area km^2 <700 m asl[a] (%)	Current IOPP concessions	Current IPT concessions (%)	Current logging concessions (%)	Current PAs (%)	Unallocated land (%)
P. r. chrysea	15,360 (**72.7**)	N/A	1,220 (**7.9**)	10,560 (**68.8**)	2,170 (**14.1**)	1,390 (**9.1**)
P. r. ignita	41,080 (**62.3**)	2,720 (**6.6 %**)	4,180 (**10.2**)	17,380 (**42.3**)	7,560 (**18.4**)	9,240 (**22.5**)
P. r. rubicunda	74,890 (**75.7**)	3,900 (**5.2 %**)	2,110 (**2.8**)	36,560 (**48.8**)	16,830 (**22.5**)	15,490 (**20.7**)
P. r. rubida	86,540 (**93.9**)	11,980 (**13.8 %**)	4,200 (**4.9**)	25,990 (**30.0**)	21,640 (**25.0**)	22,730 (**26.3**)

[a] Viable area of occupancy presented as a percentage of the maximum area of occupancy

Numbers in bold refer to the percentage of respective land-use policy on the total distribution

adaptive advantage. However, infant mortality due to starvation, lactational stress and abandonment often follows logging practices (Meijaard et al. 2005). Indeed, population density of *P. h. canicrus* appears to decline after logging events (Nijman 2000), with a possible time-lag between the onset of logging and population declines (Howell 2003). Meijaard et al. (2005) also noted that logging often results in an increase in hunting, which has a dramatic effect on *P. hosei* populations in Borneo's interior where traditional hunting practices persist (Nijman 2005). No data exist for the responses of *P. frontata* and *P. chrysomelas* to logging, but Meijaard et al. (2005) concluded that *Presbytis* ssp. would benefit from a more wildlife conscientious concession management policy (Fig. 5).

The long-term viability of habitats may be more secure if designated as PAs, as they are less likely to be de-gazetted than other land-use allocations (Wich et al. 2012; Gaveau et al. 2013). However, while current PA allocations also account for substantial areas within each distribution (28–38 %, Table 2), up to 80 % of all established PAs in Indonesia are degraded (Nellemann et al. 2007). Indeed, clearance of forests in PAs occurred at a similar rate as that in logging concessions (1.2 and 1.5 %, respectively) during the 10 year period (Gaveau et al. 2013), and severe logging occurs in as much as half of all PAs, in particular the Gunung Palung; Tanjung Puting; Danau Sentarum, and Kutai National Parks (Curran et al. 2004; Ministry of Forestry 2006). While PAs in Malaysia appear to be subject to lower rates of degradation and deforestation, they are subject to small-scale deforestation across the region where insufficient management resources are available (Wich et al. 2012).

While not a land-use policy *per se*, it is also worth discussing the detrimental effect that hunting has on *Presbytis* monkeys, as all species on Borneo are hunted for their meat and bezoar stones (visceral excretions of high economic value, used in traditional Asian medicines). For example, Nijman (2005) documented declines of up to 80 % in *P. hosei* in the Kayan Mentarang National Park following hunting for bezoar stones, and highlighted that although the protected forest remained in pristine condition, a lack of protection enforcement can result in drastic declines. Such hunting pressures can have profound influences on species' distributions, with local extinctions occurring within even remote and protected environments (Meijaard et al. 2005).

Conservation implications

At the species level, each of the four endemic *Presbytis* monkeys lost substantial areas of natural habitat over the 10 year period between 2000 and 2010. Given the current land-use

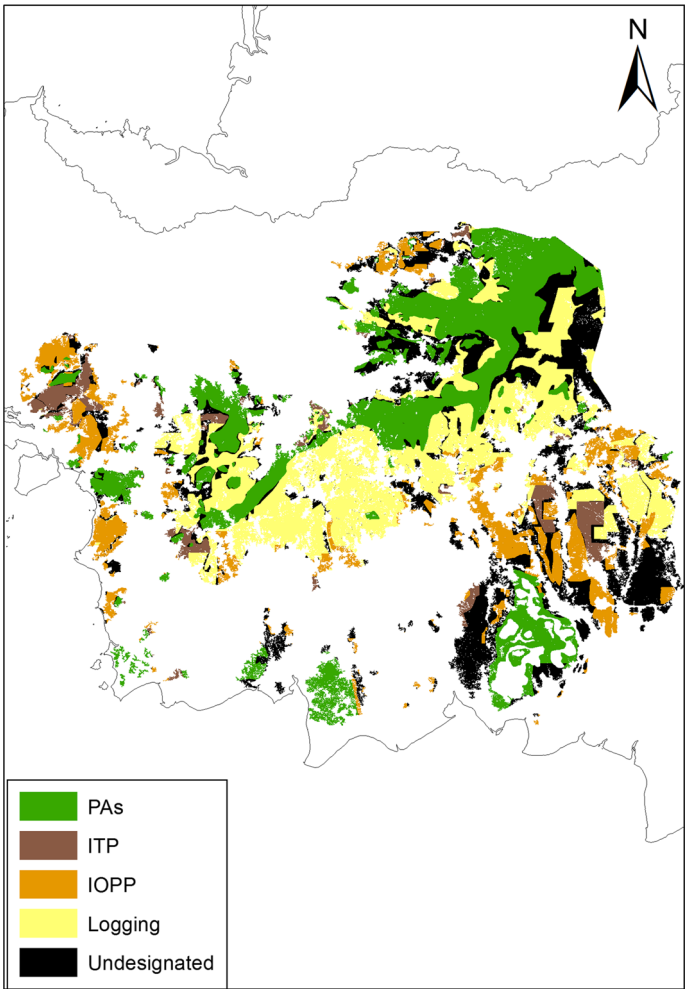

Fig. 5 An example of the land-use allocations across the distribution of *P. rubicunda rubida*

allocations, this trend will continue into the future. Indeed, if all economic allocations are exploited, between 45 and 65 % of remaining habitat will be under concession, and a minimum of 10 % will be destroyed outright (Table 2; Fig. 1).

Presbytis frontata, as a monotypic species with a broad distribution across the centre of the island where the largest concentration of PAs are situated, experienced the smallest habitat reduction in the 10 year period, and is the least affected by IOPPs (given the requirements of low elevation and slope for oil palm cultivation [Wakker 2004]). However, logging concessions accounted for over one-third of its remnant habitat (Table 2). At the subspecific level, only the habitat size of *P. r. carimatae* maintained stasis, suggesting efficient management of the PA of Karimata Island is in operation. However, populations of *P. r. carimatae* have not been conducted in 20 years (Yanuar et al. 1993) and the subspecies is considered data deficient on the IUCN Threat List (Nijman et al. 2008a), suggesting a resurvey should be considered a priority. Similarly, the uncertainty surrounding the area of occupancy of *P. h.*

canicrus is impetus for further surveys to establish the subspecies' distribution patterns. The minimum modelling scenario predicts an area of occupancy of 7,200 km^2, while the maximum predicts 11,300 km^2, which would increase if the unconfirmed species that occupies the area toward the northern border with *P. h. everetti* is in fact *P. h. canicrus*. Indeed, the subspecies occurs in the south-eastern portions of one of the largest, unfragmented forest-blocks remaining on Borneo (>55,000 km^2) with few major geographic boundaries.

The largest habitat declines occurred in the subspecies with the most restricted distributions, particularly those in Sabah (Table 2). The most extreme case of distribution reduction occurred in the critically endangered *P. c. cruciger*, which lost over 50 % of its habitat in 10 years, while *Presbytis hosei sabana* and *P. r. chrysea* each lost over 20 % of their habitat in the same period, suggesting rapid population declines. Current economic land allocations are likely to have a particularly strong negative impact on the already threatened habitat of the critically endangered *P. c. chrysea*, as a further 30 % is due to be converted for OIPP and TIPs. Likewise, the huge logging concession allocations within the distributions of *P. h. sabana* and *P. r. chrysea* (60–65 % of total distributions; Table 2) may have negative impacts on their restricted populations (Meijaard et al. 2005; Howell 2003). Data for IOPPs in Sabah are unavailable, but given the large percentages of IOPP allocations in Sarawak (Table 2) and the large reduction in habitat over the last 10 years that has occurred in the Sabahan *Presbytis* subspecies, it is not unreasonable to assume that habitat conversion for current IOPPs is likely to be high, to the detriment of *P. h. sabana* and *P. r. chyrsea* populations.

The current PA network accounts for over 20 % of subspecies' distributions in all but those that occur in Sabah. However, data show that the majority of PAs are too degraded and under-resourced to provide effective protection (Nellemann et al. 2007; Gaveau et al. 2013). Furthermore, the location of the majority of PA allocations is concentrated in the highlands, where in some cases, population densities are lower (*P. rubicunda*) or distributions do not occur (*P. chrysomelas*; Figs. 1, 2 and 4). These findings are consistent with the conclusions of Wich et al. (2012) that the current PA network is not optimally located and thus is likely insufficient for the long-term persistence of Bornean *Presbytis* monkeys.

Instead, huge proportions of each subspecies' distributions are currently not designated for economic allocation (Table 2). While the gazetting of these areas would, of course, increase the PA network (for example, ~30 % of the remnant 3,800 km^2 of habitat for *P. c. chrysea* is unallocated; if gazetted, the PA network of this critically endangered primate would increase to approximately 58 %), this does not guarantee species protection and persistence, due to the ineffective way PAs are currently managed (Nellemann et al. 2007). It is, in fact, possible that increasing the number of PAs within an already insufficiently funded PA network may be diluting, counterproductive and ultimately detrimental to species conservation. However, it is particularly clear that urgent action is required for the critically endangered species; the designation of a small PA on the unallocated lands to safeguard their persistence is likely to be more beneficial than no action at all. While logging concessions are better able to generate revenue than PAs, and appear at least as effective in preventing forest loss (Gaveau et al. 2013), the actual effects of logging on *Presbytis* populations living within concessions remain unclear, and may well have a negative effect (Nijman 2000; Howell 2003; Meijaard et al. 2005, 2008).

Conclusions and recommendations

The threat to Borneo's endemic *Presbytis* monkeys through habitat destruction, disturbance and degradation is severe and sustained, and the trend is set to continue given current

land-use allocations (Wich et al. 2012). Between 2000 and 2010, the *Presbytis* subspecies of Borneo experienced a mean 12.7 % (N = 12; mean of each value for both scenarios of *P. h. canicrus* contributes to grand mean) habitat reduction, and 12.5 % of all habitats were allocated for conversion to IOPP and ITPs. While the current protected area networks encompassed an average 33.4 % of distributions, the majority of PAs are degraded and threatened by logging and mining operations (Nellemann et al. 2007). I therefore recommend priority gazetting of unallocated lands to PAs within the distribution of critically endangered *Presbytis chrysomelas* and *P. h. sabana*, which have experienced the highest forest loss in the last 10 years (22–50 %). Logging concessions, which may have detrimental effects to *Presbytis* monkey populations (Meijaard et al. 2005), accounted for a mean 26.3 % across distributions but appear to be at least as effective in maintaining forest cover as PAs (Gaveau et al. 2013). I therefore recommend an urgent assessment of the effects of logging practices on species' persistence, as concessions have an economic advantage for effective management over PAs. I further recommend population surveys to quantify the populations of the critically endangered *P. c. chrysea* and the data deficient *P. r. carimatae* and *P. r. chrysea*, and further surveys to establish the distributional boundaries of *P. h. canicrus* and *P. h. everetti*.

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