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Large Mammal Use of Linear Remnant Forests in an Industrial Pulpwood Plantation in Sumatra, Indonesia

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Abstract

Riparian forests are often the last remaining areas of natural vegetation in agricultural and plantation forestry landscapes. Covering millions of hectares of land in Indonesia, industrial pulpwood plantations have rapidly replaced native forests. Our study aimed to better understand the conservation importance of linear remnants of riparian forest by examining their use by larger (>1 kg) mammal species. Our study site was located within an extensive acacia (*Acacia mangium*) plantation adjoining Tesso Nilo National Park in Sumatra, Indonesia. Camera traps were used to detect mammals at 57 sites to assess the effects of corridor design and land cover covariates and species behavioral traits on mammal habitat use of four linear riparian forests. We recorded 17 species (including one International Union for Conservation of Nature [IUCN] Critically Endangered, two Endangered, and four Vulnerable) in riparian forests inside the plantation, including the Sumatran tiger (*Panthera tigris sumatrae*), Malay tapir (*Tapirus indicus*), and sun bear (*Helarctos malayanus*). Some threatened species were only detected in the park buffer zone. Species varied in their responses to riparian forests, but distance to the national park, remnant width, and percent forest cover around the camera site were common predictors of remnant use. Many mammal species used riparian forests regardless of whether they were surrounded by intact acacia forests or recently cleared land. Our results indicate that linear remnant riparian forests ≤ 200 m in width can facilitate local (< 4 km) movements of many large mammal species in Sumatra, but wider riparian remnants would likely be more effective at promoting mammal movements over longer distances.

Keywords

acacia, corridor, Indonesia, mammal, plantation, tropical forest

Introduction

Production landscapes threaten tropical ecosystems in Indonesia through deforestation, inadequate governance, and poor management of knock-on effects associated with development (Holmes, 2002; McCarthy & Zen, 2010; Miettinen, Shi, & Liew, 2011; Murdiyarso, Dewi, Lawrence, & Seymour, 2011; Paoli et al., 2013). Indonesia has suffered large environmental losses with the conversion of natural forests to production landscapes, especially oil palm and pulpwood plantations (Abood, Lee, Burivalova, Garcia-Ulloa, & Koh, 2015; Fitzherbert et al., 2008; Obidzinski & Dermawan, 2012). Against this backdrop, conservation strategies that incorporate production landscapes have become increasingly popular drawing attention to factors

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affecting their conservation value (e.g., William F Laurance et al., 2010; Wilson et al., 2010; Yaap, Struebig, Paoli, & Koh, 2010).

Riparian forests are afforded legal protection in Indonesia (Republic of Indonesia, 2011) and often constitute the last remnants of native forest in industrial production landscapes, such as wood pulp and oil palm plantations. When in close proximity to larger blocks of native forest, remnant linear strips of riparian forest can potentially serve as corridors for forest-dependent species, facilitating access to forest fragments embedded in a plantation matrix and providing connectivity across the broader landscape (McShea et al., 2009; Nasi, Koponen, Poulsen, Buitenzorgy, & Rusmantoro, 2008).

Pulpwood plantations are rapidly expanding in Indonesia and have replaced extensive areas of natural forest (Abood et al., 2015; Obidzinski & Dermawan, 2012). Covering millions of hectares of land—estimated at 4.9 million ha in 2010 and with national targets to triple planted areas to 14.7 million ha by 2030 (Obidzinski & Dermawan, 2012)—wildlife-friendly pulpwood plantations could play an important role in conserving biodiversity. These plantations (predominantly *Acacia* and *Eucalyptus* spp.) are often adjacent to protected areas and large blocks of native forest, especially in Sumatra (*Last Chance to Save*, 2010; World Wide Fund for Nature [WWF], 2006). By law, industrial plantations are required to maintain a network of riparian forests of 50 to 100 m width on either side of rivers (Republic of Indonesia, 2011), but in practice, these riparian buffers are highly variable based on company interpretations of various laws (Nasi et al., 2008) and different levels of illegal forest encroachment. Wider buffers tend to be associated with unplantable (steep) and seasonally flooded forests.

Empirical studies on the use of biological corridors (including linear remnant forests) have largely focused on temperate regions (de Lima & Gascon, 1999; S. G. Laurance & Laurance, 1999; Lees & Peres, 2008). Decades of corridor research suggest that a number of factors can influence the functionality of corridors, including ecology of the target species, corridor width and length, matrix permeability, habitat quality in the corridor, level of connectivity (i.e., presence of gaps), presence of alternate pathways and nodes (i.e., resting spots along a corridor), anthropogenic disturbances, overall ecosystem connectivity, and, importantly, political will for implementation (Beier, Majka, & Spencer, 2008; Bennett, 2003; J. Fischer, Lindenmayer, & Manning, 2006; Hilty, Lidicker Jr, & Merenlender, 2006; Jain, Chong, Chua, & Clements, 2014; S. G. W. Laurance, 2004; D. B. Lindenmayer & Nix, 1993; United States Department of Agriculture [USDA], 2004).

Studies on connectivity through linear forest remnants have been undertaken in fragmented agricultural and pasture landscapes in the American tropics (Barlow et al., 2010; Ibarra-Macias, Robinson, & Gaines, 2011; Lees & Peres, 2008), but few such studies have focused on large mammals or riparian forests, especially in tropical Asia. To date, only two studies have assessed the use of remnant forests by large mammals in Southeast Asian plantations. In Sumatra, Nasi et al. (2008) identified a need for direct connectivity of riparian remnants (with no gaps) to allow movements of primates and underscored the importance of habitat quality in the remnants. In Malaysian Borneo, McShea et al. (2009) found that forest type (secondary forest versus acacia plantation) and proximity to secondary forest affected remnant occupancy for seven large mammal species.

This study aims to better understand the use of linear remnant riparian forests by large mammals in an acacia (*Acacia mangium*) plantation in Sumatra, Indonesia. In addition to determining the species composition of mammals using linear remnants, we also investigate how remnant use is influenced by corridor-design covariates such as (a) remnant length and width, (b) remnant connectivity, (c) distance to a core forest habitat (Tesso Nilo National Park [TNNP]), as well as (d) surrounding land cover (including the presence and age of the surrounding acacia plantation). We hypothesize that (1) wider and shorter remnants, (2) connected remnants, (3) sites located closer to the national park, and (4) remnant sites with more native forest or older acacia plantations are more likely to be used by larger mammals. Based on our findings, we provide management recommendations to improve the function of riparian remnants as corridors in Sumatran production landscapes.

Methods

Study Area

This study was conducted from July 2011 to January 2012 in lowland tropical rainforest embedded within and abutting an *Acacia mangium* wood fiber plantation in Riau, Sumatra, Indonesia (0°18'–0°24'S, 101°52'–102°0'E; Figure 1). The plantation borders TNNP, which is likely to harbor the complete array of medium- and large-sized mammals native to Riau's lowland rainforests (excluding flooded forests; [IUCN], 2016; Nasi et al., 2008; ProForest, 2006). Located just south of the equator, the site has a mean annual rainfall of 2,600 mm with a drier period in July (averaging ~120 mm) and the wettest period in November (averaging ~300 mm). Temperature is relatively consistent throughout the year, with a mean high of 31°C and a mean low of 23°C.

At the time of survey, the plantation was dominated by acacia stands of varying ages (<1 to 8 years old) and a network of riparian forests ranging from 80 to 1,000 m in width. Some of the riparian forests connect directly with

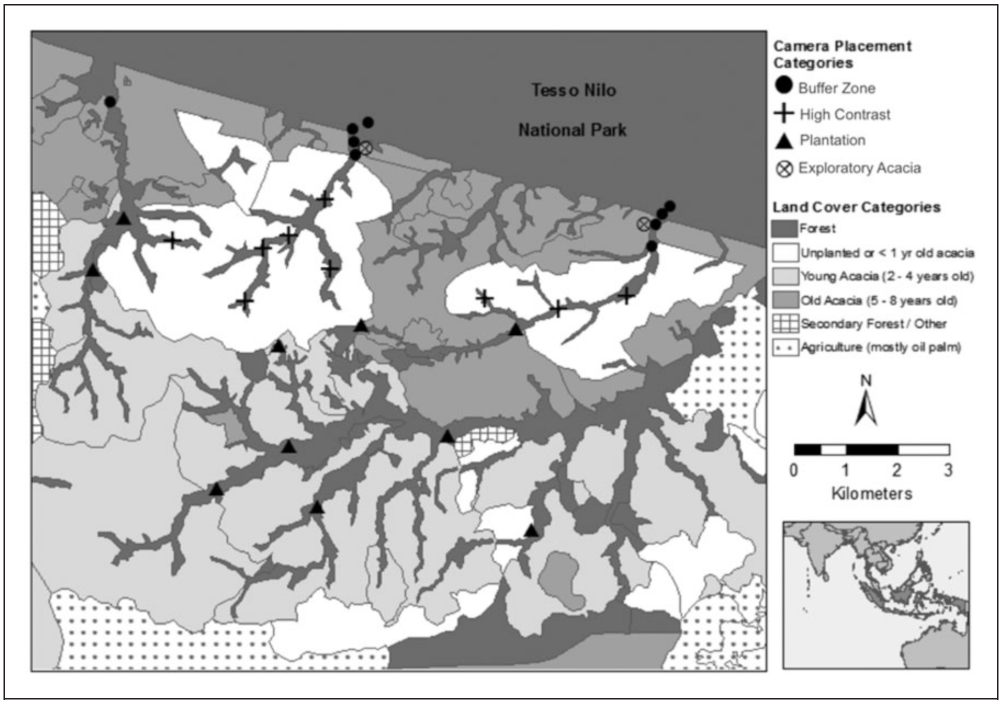

Figure 1. Survey area located in Riau Province, Sumatra, Indonesia.

native forest in TNNP at the northern border of the plantation. Toward the southern and eastern borders of the plantation, riparian forests exist as islands within the acacia matrix, disappearing where the landscape shifts to oil palm and rubber plantations (Figure 1). To the west, the plantation is contiguous with another acacia plantation that is similar in layout to the plantation we surveyed. All riparian forests in the study area had breaks in forest cover where management roads (typically 15–20 m wide) traversed the linear remnants, though many of these roads supported vegetation themselves and were impassible to vehicles in older acacia stands. Smaller areas of conservation forest set-asides, typically unplantable and seasonally flooded forests, and patches of contested land with regenerating forest were also present in the landscape.

The acacia plantation was undergoing its first harvest during the year the survey took place, leaving some riparian forests surrounded by a deforested landscape of bare soil (Figure 2(a)) or newly planted seedlings (Figure 2(b) and 2(c)). We refer to these as ‘‘high-contrast remnants’’ because they lack the acacia-tree matrix that larger mammal species may use and have a sharply contrasting edge along the forest-plantation transition. Track surveys in areas surrounding the high-contrast remnants revealed very few signs of mammal use, but these remnants were not truly isolated because they maintain connectivity to the national park and acacia matrix at one or both ends (Figure 1) and some mammal species may occasionally cross these expanses of bare soil and newly planted seedlings.

Survey Design and Camera Trapping Protocol

We deployed 20 camera traps (Reconyx Hyperfire HC 500, Wisconsin, USA) to detect larger mammals in riparian forests near TNNP and the adjacent acacia plantation (Figure 1). We camera-trapped 53 sites over three trapping rounds along a distance gradient in four riparian forests; three of which were directly connected to TNNP. The linear remnant forests sampled ranged from 80 to 530 m (mean = 207 m, $SD = 112$ m) in width, with seasonally inundated riparian areas being up to 850 m wide in one remnant. We deployed an additional four cameras in the acacia matrix. Each site contained a pair of cameras placed ~50 m apart. For each pair, we randomly selected one camera for treatment with scent lure (Ross Carman, Magna Glan, New Milford, PA). We attached all cameras to trees, ~40 cm above the ground, along animal trails. The cameras were set and left unchecked for three consecutive periods ranging from 7 to 12 weeks. We cataloged all camera images and considered consecutive detections of the same species to be ‘‘notionally independent’’ if there was >30 minutes between detections. We combined lesser (*Tragulus kanchil*) and greater (*T. napu*) mouse deer detections, as these two species could not be consistently discriminated.

Figure 2. Photos of the *Acacia mangium* plantation surveyed showing the barren landscape surrounding "high-contrast" remnants which includes (a) recently harvested areas and (b) recently planted areas. Aerial photographs of the plantation showing (c) a "high-contrast" remnant in newly planted acacia and (d) a linear remnant embedded in 8-year-old acacia plantation (the most mature acacia in the study site) at the point where it adjoins Tesso Nilo National Park.

Corridor-Design Covariates

Using ArcGIS (Version 10; Environmental Systems Research Institute, Inc., Redlands, CA), we calculated remnant width by averaging remnant width at the camera site (100 m upstream and downstream). We arbitrarily assigned a width of 700 m (more than 100 m wider than the widest linear remnant in our analysis) to cameras placed in native forest near TNNP (i.e., remnants not surrounded by acacia or barren areas). We only calculated remnant length for high-contrast remnant sites, as these sites are most representative of true corridors. We also used Euclidean distance, which is highly correlated with distance via riparian corridors (Spearman correlation, $r = 0.97$), to measure distance to TNNP from each camera. Given the paucity of animal signs, we assumed that individual animals did not traverse the bare or newly planted land surrounding the high-contrast remnants to reach the sample sites, but rather traveled along the linear remnants. The distance that a species traveled along high-contrast remnants was measured from the point where (1) the land cover on either side of the remnant became denuded or (2) recently planted to the furthest site in the remnant where the species was detected. On the basis of their location, we assigned each camera site to one of three remnant categories (Figure 1), each of which had similar sampling intensity: (a) "high-contrast" remnant ($n = 17$; Figure 2(a) to (c)), (b) "buffer-zone" remnant located within 1 km of TNNP ($n = 17$; Figure 2(d)), or (c) "plantation" remnant, located > 2 km from the national park ($n = 19$; Figure 1).

Land-Cover Covariates

Using ArcGIS, we assigned categories of land cover surrounding each camera based on plantation-company planting maps that were verified by ground truthing and satellite imagery (Landsat 7 images from 24 August 2011 and 30 December 2011) recorded during the study in mid-late 2011: (a) native forest, (b) older acacia (5–8 years old), (c) younger acacia (2–4 years old), or (d) barren land (bare soil or newly planted with acacia seedlings), represented by the percent area of each in a 1-km radius buffer area around each camera. We excluded barren land from our data analyses because of its strong negative correlation with forest cover (Spearman correlation, $r = -0.66$).

Data Analysis

We compared mammal species diversity among remnant categories using sample-based rarefaction curves with 95% confidence intervals, constructed using the Chao 1 abundance estimator using the *iNEXT* package (Colwell, 2006; Hsieh, Ma, & Chao, 2013) in R 3.1.0 (R Development Core Team, 2014). Due to the high sensitivity of species richness estimates to sample size, we standardized accumulation curves by the total number of individuals sampled within each linear remnant category (Gotelli & Colwell, 2001). We conducted an ordination of sites based on their species composition using the Bray–Curtis index, and then compared community composition among remnant categories using nonmetric multidimensional scaling in R. We also conducted a permutational multivariate analysis of variance using distance matrices to assess effects of landscape covariates.

We elucidated important corridor-design and land-cover correlates (i.e., connectivity with national park, distance to national park, remnant width, extent and age of acacia, extent of native forest; Table 1) of species richness using linear mixed-effect models (LMM). We used all detection data and included ‘‘remnant’’ as a random factor to account for nonindependence of cameras located within the same remnant and with bait as a fixed factor. We also included offsets for the number of nights a camera was active. To avoid model over fitting due to the limited size of the data set, we included no more than one landscape covariate per 10 samples in a single model and no more than 20 models were run in a model set (Field, Miles, & Field, 2012). We built models representing all possible combinations of covariates, while also keeping the number of covariates in the models ≤ 2 and not combining land-cover covariates or intercorrelated covariates ($r > 0.6$) in the same model. We selected the best-fitting models based on Akaike's Information Criterion (AIC; Akaike, 1974), with all models $\Delta AIC < 2$ considered useful for inference (Burnham & Anderson, 2002). We built LMMs using the lme4 package (Bates, Maechler, Bolker, & Walker, 2014) in R.

We investigated how the same set of corridor-design and land-cover covariates (see earlier) affected habitat use of individual species (Table 1). We created single-season occupancy models (D. I. MacKenzie et al., 2002) in the program PRESENCE v.6.9 (Proteus Wildlife Research Consultants, New Zealand; http://www.proteus.co.nz) to estimate the probability of occupancy (*psi*) and detection (p) of a species. When sampling takes place in the absence of a closed sampling period (individuals can move in and out of the study site) and sample units are not based on the home-range size of a species, occupancy rates resulting from PRESENCE models can be interpreted as habitat use (D. MacKenzie, Royle, Brown, & Nichols, 2004). We partitioned detection histories into 2-week sample periods (the length of time that best suited our data) and used to analyze eight species that had 40 or more detections in the 2-week data set (20% detection rate or higher).

We used a two-step approach for habitat-use modeling (McClure, Rolek, & Hill, 2012; Olson et al., 2005). First, we modeled sampling covariates (Table 1) using single-covariate models to identify the most influential covariate of detection probability while holding *psi* constant at the

Table 1. Site and Sampling Covariates Used to Respectively Model Mammal Habitat Use and Detection Probability for Eight Species in the Study Site.

Abbreviation	Name	Description
Site covariates		
AcOld	Percent older acacia (planted between 2004 and 2007) in a 1-km radius from the sample site	Numerical
AcYoung	Percent younger acacia (planted between 2009 and 2010) in a 1-km radius from the sample site	Numerical
Forest	Percent forest in a 1-km radius from the sample site	Numerical
Width	Corridor width (average of width at sampling point and 100 m up and down stream)	Numerical
DistMain	Distance to core habitat (Tesso Nilo National Park)	Numerical
ConnMain	Direct connectivity with core habitat (Tesso Nilo National Park)	Categorical (Yes, No)
Sampling covariates		
Bait	Bait used (Magna Glan)	Categorical (Yes, No)
Setup	Camera position	Categorical (High, Low, Good)
IsoCorr	High-contrast remnant (surrounded by bare land or acacia planted <1 year prior to sampling)	Categorical (Yes, No)
Season	Wet (October to January) or dry (July to September) season	Categorical (Wet, Dry)
Corridor	Corridor sampled	Categorical (1 to 4)

intercept (Table A1). Second, we included the top *p* model with all combinations of selected corridor-design and land-cover covariates (*psi* models) to identify the most important predictors of habitat use (Table A2). We built the same set of models for species richness (Table A3). In all models, detection-probability covariates included bait (present or absent), camera setup (lower or higher), sites located in high-contrast remnants (yes or no), season (dry = July–August; wet = October–January), and remnant (four sampled) as random effects (Table 1).

We used weight averaged occupancy rates calculated from PRESENCE models for each species to test the independence of our sample sites using Moran's I test for spatial autocorrelation in the Spatial Toolbox of ArcGIS (Version 10; Environmental Systems Research Institute, Inc., Redlands, CA). The resulting values range from 1 (displaying a complete clustering of detections) to −1 (showing a negative autocorrelation).

Results

In 3,337 trap days, we recorded 19 mammal species in 895 separate camera detections (Table 2). This constitutes about half of the terrestrial and semiterrestrial larger mammal species we considered likely to be present in our study area. The pig-tailed macaque, sun bear, red muntjac, and Malay tapir were the most frequently detected species, both in overall detections and the proportion of cameras that detected them (Table 2). Carnivores and the Sunda pangolin were among the least frequently detected species (Table 2).

High-Contrast Remnants

Thirteen of the 19 (68%) mammal species detected in the study were detected in high-contrast remnants, suggesting that mammals are able to use riparian forest remnants of 80 to 320 m width (mean 137 m, *SD* 45 m) surrounded by barren land (Table 2). The tapir, sun bear, pig-tailed macaque, red muntjac, marbled cat, and wild pig were detected at the sites farthest from the national park, up to 3.75 km into one of the high-contrast remnants. Detections were relatively well spread throughout the length of the remnants showing no correlation between detection frequency and distance into the remnant, with the exception of the tapir which has increasing detections

Table 2. Species Detected, IUCN Red List Category, Species Diet, and Detection Frequency Statistics.

						Detections per 100 trap days			
Common name	Scientific name	IUCN	Diet	No. of detections	Proportion of sites ($n = 57$)	Plantation ($n = 19$)	Buffer zone TNNP ($n = 17$)	High-contrast remnant ($n = 17$)	Acacia ($n = 4$)
Pig-tailed macaque	*Macaca nemestrina*	VU	H	185	0.89	3.52	6.47	7.49	3.38
Sun bear	*Helarctos malayanus*	VU	O	143	0.68	3.24	4.64	6.51	0
Red muntjac	*Muntiacus muntjak*	LC	H	106	0.58	1.90	3.90	3.44	4.14
Mouse deer	*Tragulus* spp.	LC	H	99	0.28	0.86	4.98	3.69	0
Malay tapir	*Tapirus indicus*	EN	H	98	0.54	2.38	4.06	1.23	5.26
Malay porcupine	*Hystrix brachyura*	LC	O	85	0.47	2.00	3.73	1.97	1.13
Wild pig	*Sus scrofa*	LC	O	70	0.60	2.00	1.24	3.69	1.5
Malay civet	*Viverra tangalunga*	LC	O	49	0.40	2.66	0.58	1.72	0
Long-tailed porcupine	*Trichys fasciculata*	LC	O	21	0.07	0.00	1.58	0.25	0
Marbled cat	*Pardofelis marmorata*	NT	C	9	0.12	0.00	0.08	0.98	0
Sambar deer	*Rusa unicolor*	VU	H	8	0.12	0.38	0.33	0.00	0
Short-tailed mongoose	*Herpestes brachyurus*	NT	C	7	0.11	0.10	0.08	0.61	0
Common palm civet	*Paradoxurus hermaphroditus*	LC	O	4	0.07	0.10	0.17	0.00	0.38
Leopard cat	*Prionailurus bengalensis*	LC	C	3	0.04	0.00	0.17	0.00	0.38
Sumatran tiger	*Panthera tigris sumatrae*	CR	C	3	0.05	0.10	0.17	0.00	0
Yellow-throated marten	*Martes flavigula*	LC	O	2	0.04	0.00	0.08	0.12	0
Clouded leopard	*Neofelis diardi*	VU	C	1	0.02	0.00	0.08	0.00	0
Binturong	*Arctictis binturong*	VU	O	1	0.02	0.00	0.08	0.00	0
Sunda pangolin	*Manis javanica*	CR	I	1	0.02	0.00	0.00	0.12	0

Note. International Union for the Conservation of Nature (IUCN) Red List Categories: CR = Critically Endangered; EN = Endangered; VU = Vulnerable; LC = Least Concern; C = carnivore, O = omnivore, H = herbivore, I = insectivore.

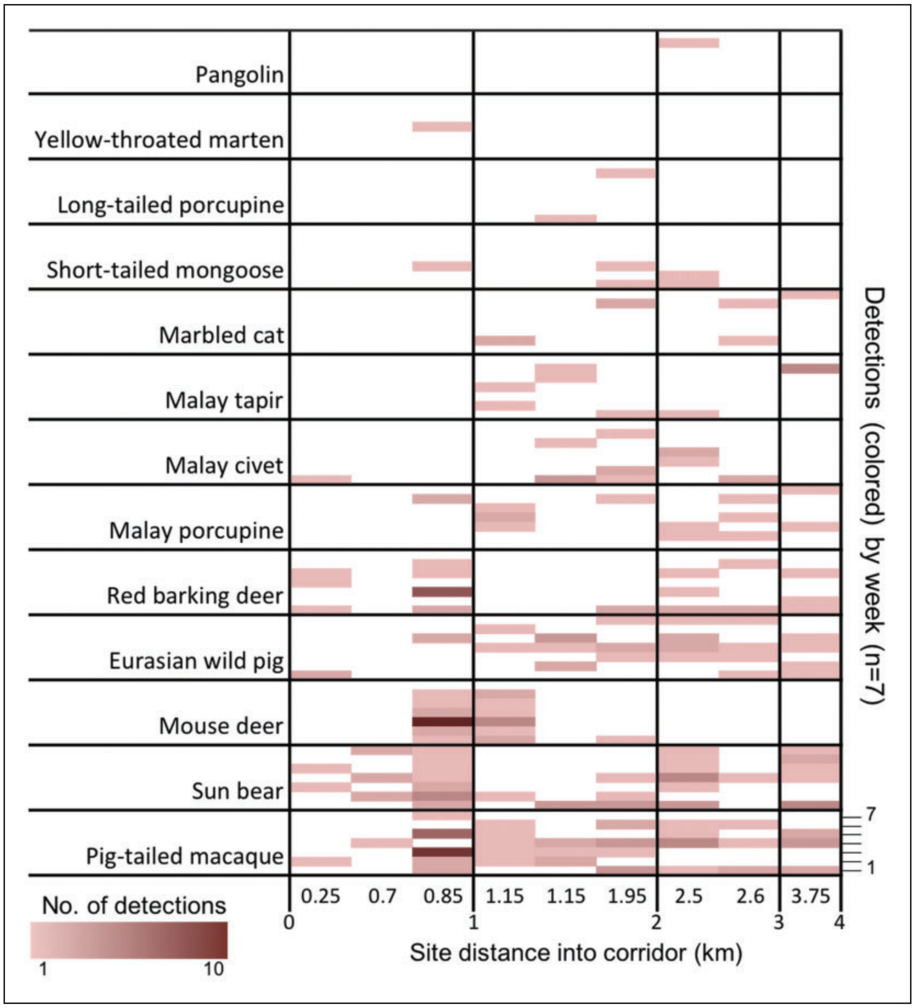

Figure 3. Species detections in "high-contrast" riparian forest remnants (linear forest remnants roughly 100 to 200 m wide surrounded either side by denuded land or recently planted *Acacia mangium*). Each distance sampled (represented by two cameras spaced c. 50 m apart) is displayed along the x-axis and detection data along the y-axis. Shaded cells indicate that a detection occurred that week; the shade gradient shows a single detection in the lightest shade and the maximum detections recorded ($n = 10$) in the darkest shade. Detections are considered independent if a 30-min gap exists between the last photo of a series and the first photo of a subsequent series.

at greater distances into high-contrast remnants ($r = 0.61$, $p < 0.05$; Figure 3).

The other six species not detected in high-contrast remnants were detected infrequently elsewhere: the Sunda clouded leopard and binturong were only detected once in the park buffer zone; the Sumatran tiger was detected three times by our cameras (in two of the three connected remnants) although tiger tracks were seen throughout the plantation over the course of the study, often along dirt transit roads and well-used human paths; and the leopard cat, sambar deer, and common palm civet were detected ≤ 8 times. We detected the Asian elephant regularly on transit roads in the plantation through tracks, dung, and company–employee sightings, but never within high-contrast remnants.

Rarefaction curves showed that mammal species diversity was largely similar in high-contrast and plantation remnants (Figure 4). Observed and extrapolated species diversity in the national park buffer-zone remnants was higher than that found in high-contrast remnants and plantation sites, although the 95% CIs overlapped, suggesting this difference was nonsignificant. Nonmetric multidimensional scaling suggests that differences in mammal community composition among the three remnant categories were not large (Figure 5), although species richness was significantly different (pseudo F value $= 2.445$; $p = 0.002$; permutational multivariate analysis of variance with 1,000 randomizations; Figure 5).

Effects of Corridor-Design and Land-Cover Covariates

Our analysis showed that distance to national park was the strongest predictor of species richness

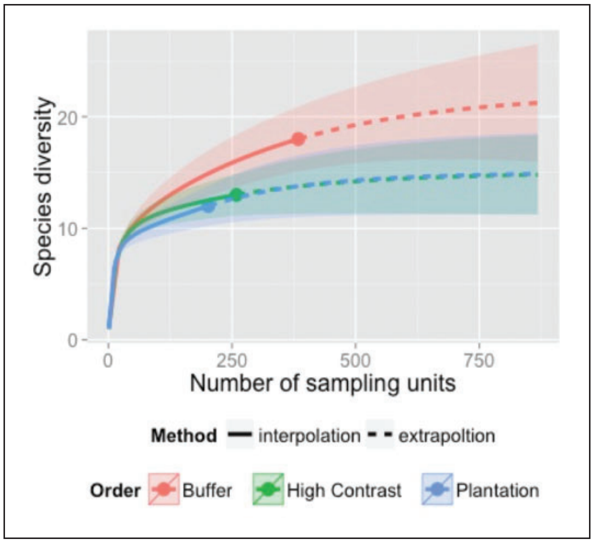

Figure 4. Observed (solid line) and extrapolated (dotted line) species diversity constructed using sample-based rarefaction curves for Tesso Nilo National Park buffer zone (red), sites within the plantation (blue), and "high-contrast" remnants (green). The x-axis is scaled to show extrapolations up to the same number of individuals sampled in each habitat category. Shading represents the 95% CI for each habitat category.

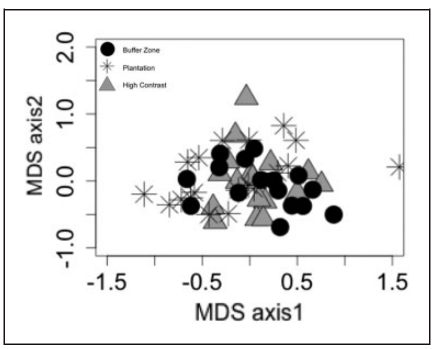

Figure 5. Multidimensional scaling graph suggesting minor differences in species composition between landscape categories.

(online Appendix S1). This covariate was in the AIC-best LMM model, although its 95% CI included zero. However, Arnold (2010) suggested using covariates for inference if the 85% CI excluded zero. Therefore, we interpret our results as weak evidence for an effect of distance from the national park.

Of the eight species analyzed using occupancy models, detection probabilities for seven were affected by sampling covariates (online Appendix S2). The use of bait had a strong positive influence on Malay civet detection ($\beta = 2.26$, SE $= 0.59$). Cameras set relatively high had a positive effect on red muntjac detection ($\beta = 1.18$, SE $= 0.45$) and a negative effect on porcupine detection ($\beta = -1.06$, SE $= 0.49$). High-contrast remnants had a negative influence on tapir detection ($\beta = -1.45$, SE $= 0.55$) and a positive influence on wild pig and pig-tailed macaque detection ($\beta = 1.02$, SE $= 0.39$; $\beta = 0.86$, SE $= 0.34$, respectively). Remnants (four sampled localities) influenced mouse deer detection probability, with lowest detection in the plantation remnant ($\beta = -2.95$, SE $= 1.09$) and the highest in the western and central remnants ($\beta = 3.71$, SE $= 1.22$; $\beta = 3.45$, SE $= 1.34$, respectively). Season did not appear to affect detection probability for any of the species modeled. There was no significant spatial autocorrelation (Moran's I ranging from 0.01 to −0.11) for the eight species modeled, except the mouse deer which was clustered (Moran's I $= 0.21$), having been detected many times at few cameras. Z-scores for all except the mouse deer were between −1.96 and 1.96, indicating that the data were not significantly autocorrelated within a 95% confidence level.

Habitat use of six out of eight species appeared to be affected by corridor-design and land-cover covariates (Table 3 and online Appendix S3). Remnant width and distance to the national park were the most common corridor-design covariates in the top models for each species. Tapir and Malay civet had increased habitat use with wider remnants, whereas sun bear showed an opposite trend. Tapir and red muntjac habitat use increased with increasing distance from the national park, but this relationship was opposite for the Malay civet. Direct connectivity to the national park only influenced habitat use of the Malay porcupine, which increased use with more direct connectivity. None of the covariates analyzed in the models explained habitat use of the pig-tailed macaque or wild pig (Table 3).

Land-cover covariates influenced five of the eight species. Tapir and mouse deer had a positive association with forest cover, but sun bear had a negative association. Malay porcupine had a strong negative association with older acacia, while the red muntjac had a negative association with young acacia (Table 4). Although not included in the occupancy modeling, tapirs were regularly detected by the four exploratory cameras placed in old acacia stand.

Discussion

Corridor Length and Width

The importance of corridor width and length has been little studied in the tropics. Our study suggests that many larger mammal species in Sumatra are willing to use linear remnants ranging from 80 to 530 m in width (with most remnants being 100–200 m in width), traveling at least 3.75 km along these remnants away from core areas of native forest. Our findings are broadly similar to studies of Australian arboreal mammals, which suggested that remnant rainforest corridors of at least 200 m

Table 3. Top Logistic Models for Predicting Habitat Use of Eight Mammal Species Based on Riparian Corridor Features in an *Acacia* Plantation Landscape.

Species	Est. naïve *psi*	AIC	ΔAIC	AIC wgt	No.Par.	(−2LL)	β	SE
Tapir p(IsoCorr)	0.52							
psi(forest+DistMain)		207.88	0.00	0.2470	5	197.88	4.10	2.08
psi(DistMain+width)		207.98	0.10	0.2350	5	197.98	1.201	0.91
Sun bear p(1)	0.73							
psi(width)		248.44	0.00	0.1884	3	242.44	−0.813	0.43
psi(forest)		249.77	1.33	0.0969	3	243.77	−3.206	2.08
Pig-tailed macaque p(IsoCorr)	0.87							
psi(1)								
Wild pig model p(IsoCorr)	0.80							
psi(1)								
Mouse deer p(corridor)	0.30							
psi(forest)		144.94	0.00	0.1649	6	132.94	0.6390	0.45
Malay porcupine p(setup)	0.48							
psi(AcOld+ConnMain)		191.37	0.00	0.3916	5	181.37	−6.972; 2.773	3.39; 1.50
Red muntjac p(setup)	0.56							
psi(DistMain)		217.69	0.00	0.3484	4	209.69	15.198	18.22
psi(AcYoung)		219.61	1.92	0.1334	4	211.61	−2.701	1.78
Malay civet p(bait)	0.44							
psi(DistMain+width)		148.09	0.00	0.5131	5	138.09		

Note. AIC = Akaike's Information Criterion.

Table 4. Occupancy Model Beta Coefficients (β) and SEs Showing the Strength (slope) and Direction of Influence of Each Habitat Use Covariate on the Species Analyzed.

	Model occupancy covariates					
Species	Forest	Old Acacia	Young Acacia	Distance to national park	Connectivity to national park	Remnant width
Tapir	4.09			1.20		0.86
Sun bear	−3.21					−0.81
Pig-tailed macaque						
Wild pig						
Mouse deer	0.64					
Malay porcupine		−6.97			2.77	
Red muntjac			−2.70	15.20		
Malay civet				−9.22		10.43

in width were desirable (S. G. Laurance & Laurance, 1999). In Amazonia, it was also suggested that remnants of ∼400 m were desirable for mammals (Lees & Peres, 2008). Our results also fall roughly within recommended corridor widths of 30 to 500 m for temperate forests (R. A. Fischer & Craig Fischenich, 2000).

Even high-contrast remnants, surrounded by a relatively hostile matrix of recently cleared or replanted land, facilitated movement of threatened species and tiger prey species. The Sumatran tiger is of particular importance as a Critically Endangered species central to conservation efforts in Sumatra. Four of the eight IUCN-listed threatened species detected in the landscape (pig-tailed macaque, sun bear, tapir, and pangolin; Table 1) and all tiger-prey species (pig-tailed macaque, tapir, sambar deer, mouse deer, wild pig, red muntjac, and Malay porcupine; O'Brien, Kinnaird, & Wibisono, 2003), except the sambar deer, were detected in high-contrast remnants. The four threatened species not detected in high-contrast remnants (clouded leopard, binturong, tiger, and sambar deer) were detected infrequently by our cameras. The Sumatran tiger and sambar deer are known to use acacia plantations (McShea et al., 2009; Sunarto et al., 2012). Tigers are likely opting to travel along areas with better forest and acacia cover (Sunarto et al., 2012) and on larger trails than those sampled (Karanth & Sunquist, 2000). The species detected furthest (3.75 km) along the high-contrast remnants (tapir, sun bear, pig-tailed macaque, red muntjac, marbled cat, wild pig) have relatively large home ranges, which may explain their willingness to travel further from native forest.

Remnant width was an important predictor of habitat use for only three of the eight species used in occupancy modeling (all detected in high-contrast remnants), with the tapir and Malay civet favoring wider remnants and the sun bear favoring narrower remnants. Linkie et al. (2013) found that tapir occupancy in regions such as Sumatra increased in areas with a lower human disturbance, a situation more likely to be found in wider remnants. Our results indicated that tapirs showed greater habitat use with increasing forest cover (a correlate of remnant width). As the Malay tapir is an important target species for conservation, corridor design in landscapes with this species should focus on creating wider corridors and access to additional forest habitat to accommodate their needs.

Distance and Connectivity to Core Habitat

A number of tropical corridor studies have documented species-specific responses to use of core habitat compared with corridors and a negative response to reduced corridor connectivity (W. F. Laurance, Laurance, & Hilbert, 2008; Lees & Peres, 2008; Nasi et al., 2008; Parren, de Leede, & Bongers, 2002). We found a similar pattern with distance to core habitat being an important covariate for three species (tapir, red muntjac, and Malay civet, although the direction of the relationship differed among species), but less importance on direct connectivity to core habitat. Comparisons among the buffer zone, high-contrast remnants, and plantation-remnant categories showed that the national-park buffer-zone sites (closest to core habitat) were the most species-rich; although mammal community composition among the three categories was similar.

Among the species we studied, only the Malay porcupine showed evidence of requiring corridors directly connected to the national park as an important corridor-design covariate. We consider it likely that other species will avoid moving far into an acacia matrix (e.g., sun bear; McShea et al., 2009) and will also likely require a relatively well-connected network of corridors to move throughout plantation landscapes. Species dependence on direct connectivity was probably less important in this study due to the relatively high permeability of the acacia matrix (McShea et al., 2009) and the terrestrial nature of most of the species we studied. Some species, such as the clouded leopard, are unlikely to move far from core habitat forest, regardless of the level of remnant connectivity.

Land Cover

Overall, species use of the acacia plantation we surveyed was relatively high compared with detection rates in plantations in Malaysian Borneo (McShea et al., 2009). This is likely a result of our study site being connected to Tesso Nilo National Park, providing quality source habitat. It may also be a result of individual animals exploring a newly evolving landscape, displaying greater movement rates than would exist as the system approaches equilibrium. Severe poaching and the illegal planting of oil palm in and around TNNP might also have prompted some animals to use the commercial plantation, where signs of hunting activity were much more limited.

The extent of forest cover surrounding a sample site appears to be less important than we initially hypothesized, with only the tapir and mouse deer showing a positive association with increased forest cover. Tapir preference for forest remnants deep inside the plantation and forested areas in the park buffer zone reflects the known willingness of tapir to use degraded and edge habitat (Maddox, Priatna, Gemita, & Salampessy, 2007; O'Brien et al., 2003), while generally preferring forest over plantations (Maddox et al., 2007). High levels of tapir activity in the plantation, including the use of old acacia stands, may be a result of reduced habitat in TNNP, as well as the proximity of our survey sites near water, in lowland forest, and the apparent absence of tapir hunting in Sumatra (Linkie et al., 2013).

Our hypothesis that older acacia stands would be favored over younger stands was supported only for the red muntjac. The Malay porcupine, however, had a negative association with plantation age.

Implications for Conservation

Our study suggests that linear riparian remnants can have utility as habitat and potential movement corridors for many larger mammal species in Sumatra, at least for localized movements extending up to a few kilometers in length. Our corridors of remnant native riparian forest mostly ranged from 100 to 200 m in width. We believe this is a reasonable minimum width for riparian buffers to serve as movement corridors for large mammals in Sumatra. Small breaks in connectivity (e.g., service roads) did not appear to be an impediment for most large, terrestrial mammals, though wider breaks in connectivity were more important for some species.

Our study is the first to assess the habitat and landscape factors that influence the use of linear remnants by the Malay tapir. We found that that tapir use of linear remnants increases with remnant width and availability of native forest within the remnant. We also found that tapir venture deep into acacia plantations, travelling up to 3.75 km along high-contrast linear remnants, using remnants with greater intensity as they travel farther from core habitat.

The design and management of corridors for mammals in plantation-dominated landscapes require consideration of many factors affecting their suitability. Edge effects could reduce the quality of riparian corridors, especially during harvest rotations when the plantation is temporarily denuded and remnant corridors are more exposed to wind, microclimatic stresses, and additional environmental and anthropogenic pressures. In neighboring plantations that had experienced multiple harvesting rotations, riparian-forest quality was severely degraded compared with our study area. The impact of biophysical stresses was likely worsened by illegal logging, which we also observed in our study area in corridors where the surrounding acacia had recently been harvested. To maintain habitat quality and corridor functioning in the long term, the widest possible riparian corridors are recommended to counter edge effects and improve the likelihood of recovery from illegal logging.

Habitat quality and permeability of the land cover surrounding linear remnants should also be considered. Although many mammal species in our study showed a willingness to use forest remnants, the presence of an adjacent acacia matrix may be helpful to enlarge effective habitat for some species of conservation concern. Research into optimal spatial and temporal harvesting rotations that encourage corridor use by large mammals and other native wildlife could improve biodiversity outcomes of plantation management (D. Lindenmayer, Franklin, & Fischer, 2006). Based on our current knowledge on corridor use and connectivity, harvesting regimes should ensure that plantation areas do not rely solely on long, high-contrast riparian corridors to connect large mammals to core habitat.

Finally, depending on the type of plantation, governance, and ownership, connectivity and corridor design issues are often considered postdevelopment or in already fragmented landscapes. In situations where riparian buffers are degraded, patchy, or no longer present, reestablishing buffers of 100 to 200 m of native vegetation is likely to provide passage for many large mammals. Corridors at this width or wider, even when kilometers in length, could play an important role in maintaining landscape connectivity.

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