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THE ENVIRONMENTAL AND SOCIAL IMPACTS OF ROADS IN SOUTHEAST ASIA

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To Jackie, Clements, Sheema…

…and to the Leopard that crossed the road.

STATEMENT OF CONTRIBUTION

Chapters 2-5 in this thesis are manuscripts that are in preparation for submission or manuscripts that have been published. Several researchers have made contributions to these chapters and it is necessary to recognise their contributions. The entire thesis was proofread by WF Laurance and S Abdul Aziz.

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ABSTRACT

The expansion of road networks shows no signs of abating, especially in developing countries where economic growth is rapid and opportunities for natural resource exploitation are plentiful. When a road is built, there will invariably be environmental and social impacts. Among tropical regions, however, these impacts are probably least studied in Southeast Asia.

When studying the environmental impacts of roads, mammals are one of the ideal animal groups to focus on due to their sensitivity to disturbance. In Southeast Asia, there is an urgent need to address the environmental impacts of roads on mammals, especially when predicted extinction rates of mammals are relatively high. As such, I interviewed 36 relevant experts to identify roads that are contributing the most to habitat conversion and illegal hunting of mammals in 7 Southeast Asian countries. We have now identified 16 existing and eight planned roads - these collectively threaten 21% of the 117 endangered terrestrial mammals in those countries. Using various techniques, I demonstrated how existing roads contribute to forest conversion and illegal hunting and trade of wildlife. Such empirical evidence can also be used to inform decision-makers and support efforts to mitigate threats from existing and proposed roads to endangered mammals. Finally, I highlighted key lessons and propose mitigation measures to limit road impacts within the region.

Roads that warrant urgent conservation attention must be prioritised because conservation resources are limited. One way would be to focus mitigation measures on roads cutting through forests with mammal species whose populations are at ‘tipping points’. To address this, I developed the *Species’ Ability to Forestall Extinction* (SAFE) index, which incorporates a benchmark population target for long-term species persistence. I found that the SAFE index better predicts the widely used IUCN Red List threat categories than do previous measures such as percentage range loss. I argue that a combined approach – IUCN threat categories together with the SAFE index – is more informative and provides a good proxy for

gauging the relative “safety” of a species from extinction. Finally, I show how the SAFE index can be used to prioritise roads in Southeast Asia that warrant urgent conservation attention based on their passage through habitats with the most number of mammal species whose populations are at ‘tipping points’.

There is a paucity of information on the social impacts of roads in Southeast Asia. In order to address this, I interviewed 169 indigenous people (known as the Orang Asli) living in a biodiversity-rich forest complex bisected by a highway in northern Peninsular Malaysia. My surveys revealed that the majority if respondents supported the presence of the highway and construction of additional roads to their village. Overall, respondents perceive that the highway has a net positive impact on their livelihoods, despite low actual use of the highway for livelihood activities including hunting. Therefore, under circumstances where roads need to be opposed, conservation planners and practitioners may find it difficult to garner support from indigenous people who already have direct access to a previously constructed road, and desire greater access to markets, health clinics and jobs. Before a road is built, forest-dependent indigenous peoples should ideally be consulted to better understand how their socio-economic needs can be met without negatively impacting biodiversity.

In habitats fragmented by roads, underpasses are one possible mitigation measure to facilitate animal crossings. However, the role of underpasses as crossing structures for mammals as yet to be quantified in Southeast Asia. I investigated this for 20 underpasses at two fragmented habitat linkages in Peninsular Malaysia. Camera trap surveys in forests around the underpasses revealed that despite the effects of fragmentation, both linkages are still of high conservation importance for native mammals. For seven focal large mammal species, fragmentation had some degree of effect on the forest use of every focal species. The Clouded Leopard (*Neofelis nebulosa*) was the most sensitive species to fragmentation, with its forest use declining with increasing proximity to the road and reservoir, and less intact forest cover. Not only has fragmentation affected forest use of large mammals around all 20 underpasses, it has also affected the efficiency at which underpasses are used as crossing structures. Overall, these underpasses appear to be effective crossing structures for only two herbivore species, Asian Elephant (*Elephas maximus*) and Serow (*Capricornis sumatraensis*). Individual underpass-use efficiencies have been sub-optimal for all focal species except Serow. For five species, the presence of underpasses at the end of trails did not have an effect on increasing trail use – this questions the ability of underpasses to mitigate road impacts on animal crossings. Conservation planners and practitioners must recognise that it may be unrealistic to expect underpasses to be effective crossing structures for all large mammal species and ecological guilds. At each linkage, management interventions to minimise the negative effects of forest fragmentation around the underpasses should be adopted to improve their efficiency of use by large mammals.

This thesis augments the body of knowledge on the environmental and social impacts of roads in Southeast Asia. While this thesis provides strategies on how to mitigate the negative impacts of roads in this region, the real challenge lies with implementing these strategies on the ground. As an example of how conservation research can be translated into action, I report how my lobbying efforts in the State of Terengganu, Peninuslar Malaysia, have prompted the state government to: (1) implement a state-wide ban on the legal hunting of Flying Foxes (*Pteropus* spp.) that I found threatened by roadside hunting; and (2) issue a moratorium on infrastructure development along a road cutting through a habitat linkage that is important for mammal conservation.

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Chapter 1: General Introduction

Roads are proliferating across the planet at unprecedented rates. Road development poses a particularly severe challenge to conservation initiatives in developing countries, where increasing road densities are linked with economic growth and habitat degradation (Wilkie et al. 2000). For instance, between 2005 and 2010, the percentage of total roads that were paved in developing countries within East Asia soared from 16% to 51% (World Bank 2013).

When a road is built, environmental and social impacts are expected to follow. In developing countries, the impacts of roads on the environment are generally negative, contributing to deforestation, unregulated human colonization and unsustainable hunting (Laurance et al. 2009). The impacts of roads on people, however, have usually been regarded as positive. Better rural transportation in developing countries is often regarded as the major factor that improves livelihoods through better access to markets, increased social mobility, migration and greater economic opportunities (Adam et al. 2011).

When examining the environmental impacts of a road, mammals are one of many ideal taxonomic groups to focus on due to their sensitivity to disturbance. A meta-analysis on 234 mammal species showed that the negative impacts on mammalian population densities generally extend over distances of up to 5 km from infrastructure such as roads (Benítez-López et al. 2010). Fahrig and Rytwinski (2009) also found that roads have a net negative effect on animal abundance, and large-bodied mammals are especially susceptible.

In terms of research conducted on the impacts of roads on mammals and biodiversity in general, there appears to be a geographic bias. According to Taylor and Goldingray (2011), less than 25% of 244 published studies of road impacts on biodiversity were on tropical species. Within the tropics, negative impacts of roads on mammals have been mainly documented from the Amazon (Nepstad et al. 2001), Central Africa (Laurance et al. 2006, Laurance 2007; Blake et al. 2008) and northeast Queensland (Goosem 2000; 2001; Goosem et al. 2001). Studies explicitly investigating the impacts of roads on mammals in Southeast Asian are surprisingly scarce, although the region has the greatest deforestation rates in the tropics (Sodhi et al. 2004). Using a hierarchically-nested combination of keywords (Appendix 1), I found that out of 533 road-specific biodiversity studies in the BIOSIS Previews® database, only one (Austin et al. 2007a) explicitly investigated the impacts of roads on mammals in this region.

In Southeast Asia, between 9–36% of lowland forest mammal species are predicted to be extinct by 2100, especially if deforestation rates continue at 1.6% y^{-1} (Wilcove et al. 2013). As such, there is an urgent need to mitigate the negative impacts of roads on mammals in this region. To do this, conservation planners and practitioners need to know where and how roads are facilitating high rates of forest conversion and illegal hunting of mammals in their respective countries. Therefore, in my second chapter, I ask: **Where and how are roads endangering forest mammals in Southeast Asia?** To address this I solicit opinions from relevant experts involved in mammal research to locate existing and planned roads that are contributing to habitat conversion and illegal hunting of mammals in the region. Also, I use species distribution models, satellite imagery, mammal- and hunting-sign surveys and social interviews to empirically demonstrate how certain roads contribute to habitat conversion and illegal hunting and trade of mammals.

Once these specific roads are known, it would be ideal to prioritise those that warrant urgent implementation of mitigation measures as conservation resources are limited. One possible method would be to select roads that cut through forests with the most number of mammal species whose populations are at ‘tipping points’. Arguably the most widely used barometer of a mammal species’ threatened status is the IUCN Red List (International Union for the Conservation of Nature 2013), which classifies species at high risk of global extinction through an explicit, objective, and semi-quantitative framework. However, IUCN threat categories do not reflect the distance of a given species or population from extinction; for example, categories such as “Endangered” might not be easily differentiated from “Vulnerable” conceptually. Therefore, in my third chapter I ask: **Can we measure species' distance from extinction?** This is achieved with a new index that measures species' or population's distance from an arbitrary, but risk-averse minimum viable population (MVP) size required for long-term persistence and evolutionary potential (Traill et al. 2010).

However, conservation planners and practitioners concerned about the environmental impacts of roads often overlook the social impacts of roads. Positive social impacts arising from road expansion include the poverty alleviation, particularly in rural areas (Jones 2006). Yet roads sometimes do not confer sizable benefits on local people in Southeast Asia. For example, surveys in Lao People's Democratic Republic PDR revealed that the poorest rural residents ranked ‘the value of roads/access to markets’ only 8th out of 12 potential measures that can help improve their income levels (Government of Laos 2000), in part due to the poor not being able to afford supplies, such as market goods, vehicles and petrol, brought by roads (Robichaud et al. 2001). Roads may also cause social and health problems. Increases in cases of HIV/AIDS resulting from rising prostitution (Skeldon 2000) have been reported among people living near roads in Indonesia. In more extreme scenarios, local communities have had to relocate because of road development. For instance, the Asian Development Bank-financed Northern Economic Corridor Project, which links Lao (PDR) to China, necessitated the relocation of more than 90 ethnic minority villages (Cleetus 2005).

In general, there is a paucity of information on the extent to which roads have affected local livelihoods, and the degree to which they are supported by indigenous people. Peninsular Malaysia, which has more than 90,000 km of roads crisscrossing its biodiversity-rich forests (e.g. Olson and Dinerstein 2002), is a suitable location to study the impacts of roads on the livelihoods of indigenous people known as the Orang Asli (which means ‘original people’ in the Malay language). In the interests of biodiversity conservation, it is also important to elucidate the influence of roads on their hunting practices. This is because roads have been blamed for transforming indigenous people from semi-nomadic hunters into commercial traders (Suárez et al. 2009). Therefore, in my fourth chapter, I ask: **How do roads affect the livelihoods of indigenous people and what are the demographic determinants of their support for roads?** I achieve this by conducting interviews with indigenous people living in an important mammal habitat bisected by a road in northern Peninsular Malaysia.

Habitat corridors or linkages are regarded as a key conservation strategy to address forest fragmentation (Noss 1987; Saunders & Hobbs 1991). A linkage is defined (see Bennett 1998, 2003) as a habitat configuration that is not necessarily linear or contiguous that enhances the movement of animals or the continuity of ecological processes throughout the landscape. To date, empirical evidence suggests that at least some linkages can provide adequate connectivity between isolated habitats to maintain population viability (Beier & Noss 1998). By facilitating faunal movement (Harris 1984) and immigration (Harris & Scheck 1991) between fragmented habitats, linkages can help maintain gene flow and minimise deleterious effects arising from inbreeding depression (Harris 1984) and demographic stochasticity (Merriam 1991). For mammals, examples of linkages apparently facilitating population connectivity have been documented in both temperate (Mech & Hallett 2001; Hilty & Merelender 2004) and tropical regions (Laurance & Laurance 1999; Nasi et al. 2008; Caro et al. 2009).

In Peninsular Malaysia, the federal government has developed a plan to restore habitat connectivity between four fragmented forest complexes via a network of 17 primary forested linkages (Fig. 1) – known as the Central Forest Spine Master Plan for Ecological Linkages (Department of Town and Country Planning & Department of Forestry 2012). However, all but two of the 17 linkages in Peninsular Malaysia have been bisected by paved roads and most have become fragmented by logging and conversion to monoculture plantations. Two linkages have even been affected by the creation of artificial reservoirs for hydroelectric dams. As such, the importance of these linkages for the conservation of mammals remains uncertain.

At two of the 17 linkages (PL 7 and 8; Fig.1), underpasses have been integrated into the roads that bisect them, mostly to surmount topographical obstacles such as streams or large gullies. However, three of these underpasses were intentionally built by the government to facilitate animal passage (Kawanishi et al. 2011; Laurance & Clements 2010). To date, the effectiveness of underpasses as crossing structures for mammals has been evaluated in North America (Clevenger & Waltho 2000; McDonald & St-Clair 2004; Ng et al. 2004; Clevenger & Waltho 2005; McCollister & van Manen 2010; Gagnon et al. 2011), Europe (Mata et al. 2005; Mata et al. 2008), Australia (Goosem et al. 2001) and East Asia (Pan et al. 2009), but never before in Malaysia, or even within Southeast Asia.

Underpass use does not, however, imply that the structure has *mitigated* the impacts of the road. Negative impacts of roads on mammals include impediment of movement (thereby decreasing habitat accessibility and gene flow; Lesbarrères & Fahrig 2012), mortality (Colón 2002) and behavioural avoidance due to vehicle traffic (Vidya & Thuppil 2010; Gubbi et al. 2012; Brehme et al. 2013), habitat degradation (Roger et al. 2011) and hunting pressure (Blake et al. 2008). Therefore, there is a need to investigate whether underpasses have been able to ameliorate possible road impacts on mammals.

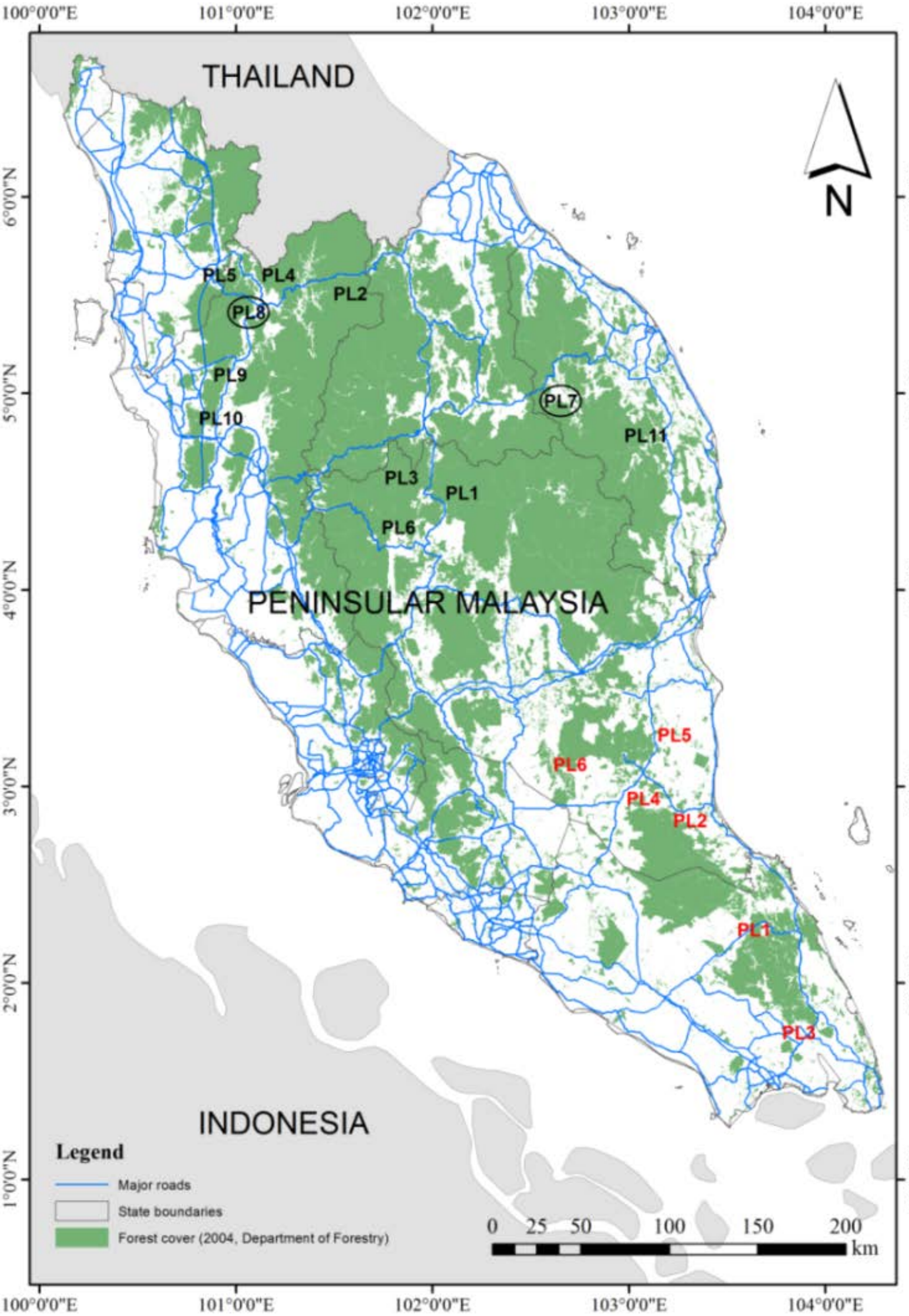

Fig. 1. *Location of 17 Primary Linkages (PL) in the northern (black labels) and southern (red labels) parts of Peninsular Malaysia to help restore habitat connectivity between fragmented forest complexes known as the Central Forest Spine (green areas). Circled linkages - Linkages 7 and 8 - are fragmented by roads that have 20 underpasses integrated into them. The role of underpasses as crossing structures for mammals is evaluated in Chapter 5.*

Therefore, in my fifth chapter, I ask four specific and related questions: **What is the conservation importance of two fragmented habitat linkages for native mammals in Peninsular Malaysia? Can all 20 underpasses serve as effective crossing structures for large mammals? Which individual underpasses are efficiently used by large mammals? Can underpasses actually mitigate the impacts of the road?** I answer these questions by deploying camera traps in forests and at underpasses in two fragmented linkages to obtain detection/non-detection data of mammals.

Ultimately, this thesis will generate new knowledge and provide valuable lessons for conservation planners and practitioners working in areas where roads impact important wildlife habitats and indigenous communities. Most importantly, the strategies recommended at the end of each chapter and in my concluding chapter can help limit the negative impacts of roads in Southeast Asia and beyond.

End of chapter 1

Chapter 2: Where and how are roads endangering forest mammals in Southeast Asia?

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INTRODUCTION

Habitat loss and unsustainable hunting are two major drivers of biodiversity declines, particularly for terrestrial mammals in tropical forests (Brooks et al. 2000; Linkie et al. 2003; Chapron et al. 2008). The expansion of roads through forests can be a precursor to both of these threats (Gaveau et al. 2009; Suárez et al. 2009; Peh et al. 2011), and is increasingly seen as a severe environmental challenge (Laurance et al. 2001; Blake et al. 2007; Laurance & Balmford 2013).

In Southeast Asia, rates of forest conversion for agriculture (Koh & Wilcove 2008) and tree plantations (Aziz et al. 2010) remain high, and hunting levels for bushmeat and traditional medicine can reach unsustainable levels (Bennett & Robinson 2008; Bennett 2011). If measures to mitigate the impacts of roads on biodiversity are to be successfully implemented in this region, conservation planners and practitioners must first know which roads are facilitating high rates of forest conversion and illegal hunting in their respective countries. The next step would be to gather empirical evidence on threats from these roads, which can be used to support efforts to mitigate threats from existing and proposed roads to endangered species.

Here, we use three eclectic lines of evidence to evaluate the impacts of roads on forests and hunting in Southeast Asia, with a particular focus on endangered mammals and their habitats. First, we asked experts involved in mammal research and conservation to identify roads that currently or potentially threaten endangered species through forest conversion and illegal hunting. Second, we gathered evidence from journals and grey literature to corroborate the threats from each road and presence of endangered species around them. Third, we developed detailed case studies based on species distribution models, satellite imagery and sign surveys to illustrate how roads (1) cut through important mammal habitats, (2) have led to intensified forest conversion, and (3) contribute to illegal hunting and wildlife trade. Based on these case findings, we highlight key lessons regarding road proliferation in Southeast Asia, and propose mitigation strategies to minimise the negative impacts of existing and proposed roads on the region's endangered mammal species.

METHODS

Location of existing and planned roads contributing to forest conversion and illegal hunting

Expert interviews have increasingly been used to gain insight into contemporary biodiversity threats (e.g. Laurance et al. 2012). Ideally, people working on the ground should provide the best available information about roads threatening endangered mammals in the region. We emailed short questionnaires to a list of experts in mammal research and/or conservation from relevant scientific institutes/universities, environmental NGOs and wildlife departments in the following countries (and sub-regions) - Cambodia, Lao PDR, Indonesia (Irian Jaya, Java, Sulawesi, Sumatra, Kalimantan), Malaysia (Peninsular Malaysia and Malaysian Borneo), Myanmar, Philippines, Thailand and Vietnam. At least one expert from each country and sub-region was contacted. To maximise response rates from busy experts, we limited each opinion to a maximum of three roads believed to contribute to forest conversion and illegal hunting/trade in each region, including road names and threatened mammal habitats. Several experts who did not respond in writing were subsequently interviewed by telephone. To minimise observer and organisation bias, we only highlighted roads named by at least two respondents with different affiliations. We relaxed our criteria for countries where there is a paucity of publicly available information on threats to mammals, such as Myanmar. Respondents also identified proposed roads in their country, but this information was included without bias reduction because the roads may not have been sufficiently publicised for corroboration by different experts. The information was eventually returned to country experts for final verification. Lastly, we corroborated expert claims of roads affecting endangered mammals with information from journals and grey literature. As a precautionary measure to prevent political repercussions, the names of experts who identified these roads will not be revealed unless permission is given.

We acknowledge three caveats here. First, the list of roads identified by experts is not exhaustive for Southeast Asia, especially when respondents are limited in number – there could certainly be more roads that were not captured by our interviews. Second, the list of roads for each country does not represent the most threatening roads in terms of impact on endangered mammals in reality, but are merely prominent *examples* based on their in-country experience. Third, roads may only be proximate drivers of forest conversion and hunting some scenarios, while government decisions to implement resource extraction activities that require the construction of new roads, such as granting logging or mining concessions or creating hydroelectric dams, may be the ultimate drivers.

Do roads bisect important mammal habitats?

Expert claims of roads cutting through important mammal habitats should ideally be supported by empirical evidence. If presence-only data for a particular species are available around roads, we recommend the use of species distribution models to illustrate the degree to which habitats around the road are important or highly suitable for the species. In areas where roads have yet to be built, this method can also be used to investigate whether a planned road would cut through important habitats for a particular species. Here, we provide a case study using presence-only data on the endangered Asian Tapir (*Tapirus indicus*) in Peninsular Malaysia to assess whether three roads identified by experts (Table 1) pass through important habitats for this species. We used Maximum Entropy modeling, a machine-learning method that models the probability of occurrence from presence-only data as a function of environmental variables using randomly selecting background pixels (known as pseudo-absences; Phillips et al. 2006). Even with limited datasets, this method can be used to predict the geographic distributions of species with reasonable accuracy (Phillips et al. 2006; Pearson et al. 2007; Wilting et al. 2010). We used a MaxEnt-predicted distribution model for the Asian Tapir from Clements et al. (2012), which was created with (1) a large dataset that was spatially and temporally representative of tapir occurrence in Peninsular Malaysia (1,261 occurrence points recorded between 1999 and 2011); (2) a small suite of biologically-meaningful variables to avoid model over-fitting (19 bioclimatic [Hijmans et al. 2005], elevation, soil [Food and Agriculture Organization et al. 2009] and 2007 land cover layers [Miettinen et al. 2008]); and (3) a grid to account for spatial bias in tapir occurrence points (see Appendix 2 for instructions to create the bias grid). In the MaxEnt software, (version 3.3.3a; Computer Sciences Department, Princeton University 2004), default settings were applied, except that 10-fold cross-validation (Elith et al. 2011) was used and the bias grid was included. Model performance was measured by the area under the receiver-operating characteristic curve (Phillips et al. 2006), which describes the ability of the model to discriminate presence from background points (Elith et al. 2010). Areas with a logistic value ≥ 0.45 were considered to be important tapir habitats. This value approximates to 0.5, which has been used by previous MaxEnt studies to indicate suitable habitats (Elith et al. 2011). Given that conservation resources are limited, it is justifiable to consider habitats that have at least a 50% chance of a species being present as important.

Predictions by MaxEnt models have certain weaknesses. They do not account for imperfect detections (e.g. Karanth et al. 2009), and the indices produced by MaxEnt are not directly related to probability of occurrence, which is a more informative measure of the importance of habitat for a species (Royle et al. 2012). When resources are available for a more in-depth quantification of important mammal habitats, detection/non-detection surveys can be conducted under an occupancy framework (MacKenzie et al. 2006) to generate occupancy maps or habitat-use-intensity maps that account for imperfect detection.

For the next case study, we used data from camera-trapping surveys (10,502 trap nights) in Chapter 5 to generate forest-use-intensity maps for two endangered mammal species that had sufficient data, the Asian Elephant (*Elephas maximus*) and Asian Tapir. The data were collected between April 2011 and March 2012 from forests on either side of State Road 156, a road identified by one of the experts in Peninsular Malaysia. Two survey blocks (see Chapter 5 for rationale) along the road were each stratified into 21 cells (2 x 2 km). Within each cell, a camera trap was first deployed in the upper-left sub-cell (1 x 1 km). After one sampling period (~60 trap-days), that camera was rotated to the upper-right sub-cell. This rotation occurred two more times, to the bottom-left and bottom-right sub-cells, until the entire cell was surveyed in a 'Z' shape manner after four sampling occasions.

Using a likelihood-based approach (Mackenzie et al. 2002; Mackenzie et al. 2005), we estimated forest use ($\hat{\psi}$) by these two species using detection/non-detection data from 158 sub-cells. Species detection histories (H) were constructed over four temporal sampling occasions (15 trap nights each) to facilitate calculation of detection probabilities (p) to account for imperfect detection. Within each detection history, '1' indicated the detection of a species by a camera trap within it, '0' indicated the non-detection of a species by a camera trap within it, and '-' indicated that that no detections were obtained from that sub-cell on that particular occasion. For example, a detection history for sub-cell i (H_i) consisting of four sampling occasions of '1001' would represent species detection on the 1st occasion and 4th occasion, and non-detection on the 2nd and 3rd occasion over a single season; the probability of recording history H_i would be,

$$\Pr(H_i = 1001) = \psi_i \, [p_{i1} (1 - p_{i2}) (1 - p_{i3}) \, p_{i4}] \qquad \text{(eqn.1)}$$

where ψ_i is the probability that sub-cell *i* is occupied and $p_{i\,j}$ is the probability of detecting the species at sub-cell *i* during sampling duration *j* (= 1, 2, 3 and 4), conditional upon the species being present.

To explicitly account for variation in detection probability (*p*), two sampling covariates were modelled for both linkages: (1) number of trap nights that cameras were operational during each sampling occasion; and (2) daily rainfall (recorded from nearest weather stations installed by the Department of Meteorology). We also modelled the effect of four site covariates that could hypothetically affect forest use of both species: 1) distance to State Road 156; 2) distance to nearest plantation; 3) distance to reservoir edge; and 4) forest cover type as a proxy of logging intensity (a binary variable; 1 - relatively intact lowland forest vs. 2 – disturbed lowland forest based on a 2010 land cover layer derived from MODIS 250-m resolution satellite images; Miettinen et al. 2012). Because our forest-use maps were at 1-km^2 sub-cell resolution, all measurements for each covariate were made using the centroid of each sub-cell as a reference instead of the camera trap location. After testing for collinearity among continuous and categorical covariates using the *hetcor* function implemented in the polycor library in R statistical environment 3.0.0 (R Development Core Team 2013), we retained covariates with coefficients <|0.5| for model construction. All continuous covariates were normalized to z-scores prior to modeling.

To obtain forest-use estimates for the two species that account for imperfect detection, we adopted a two-step process under the single-species, single-season occupancy framework in PRESENCE v5.3 software (Hines 2006). First, detection probability (*p*) was modelled where the parameter was assumed constant or allowed to vary with individual or additively combined sampling covariates, with all site covariates included in each model (MacKenzie 2006). Second, the influence of covariates on forest use (ψ) was modelled where the parameter was assumed constant or allowed to vary with individual or additively combined covariates, while maintaining the top-ranked model for detection probability derived from the first step. Models were ranked using Akaike's Information Criterion (AIC) corrected for small sample size and evaluated for goodness-of-fit against 999 simulated bootstrap datasets (MacKenzie & Bailey 2004). For each species, the top-ranked model was used to map forest-use intensities at a 1 km^2 sub-cell resolution. Four levels of forest-use intensities based on natural breaks were defined using the Spatial Join function in ArcGIS v10 (ESRI, Redlands). Finally, we calculated the mean forest-use estimates of the Asian Elephant and Asian Tapir affected by the path of State Road 156.

Does forest conversion intensify following road construction?

Satellite images are useful for detecting forest conversion around a road, especially from freely available and regularly acquired Landsat satellite imagery, which has global coverage, medium spatial resolution (30-80 m) and large historical archives (Wulder et al. 2011). Despite missing data from persistent cloud cover over some tropical forests and faults in the Scan-Line Corrector of Landsat-7, methods are available to ensure Landsat composites are comparable over considerable temporal scales (Wijedasa et al. 2012). Using a 2009 cloud-free Landsat 5 (TM) image (Path/Row: 127/52; United States Geological Survey; glovis.usgs.gov), we produced a false-colour composite for one road (Provincial Road 76) identified by experts in Cambodia, which bisects the Snoul Wildlife Reserve (12° 5'26.98"N; 106°39'40.83"E). With this technique, this road can be differentiated from vegetation and bare or built-up areas.

Time-series satellite imagery can provide more detailed information on the impacts of roads on forest cover. Once images are classified, an intensity analysis (Aldwaik & Pontius Jr. 2012) can reveal: (1) differences in annual rates of overall land category change before and after road construction; (2) the variation in intensity of gross primary forest, mosaic and bare or built-up area gains and losses during each phase; (3) whether primary forests were avoided or targeted by transitions to bare or built-up areas during each phase; and (4) whether forest conversion occurred closer to or further from roads. To obtain this information for the same road, we classified land cover in georeferenced and orthorectified cloud-free Landsat 4, 5 (TM) and Landsat 7 (ETM+) images at 30-m resolution. Analyses were run at three time intervals: when the road (1) was absent (1990), (2) was recently completed (2001), and (3) had existed for several years (2009). Inputs for land cover classification included the first three layers of a Tasseled-cap transformation (Kauth & Thomas 1976) and spectral bands 1-5 and 7. Data layers were processed using an unsupervised classification (ISODATA) algorithm with a maximum class of 200, 50 maximum iterations with a convergence threshold of 0.95. Using both the original satellite data and Google Earth images as auxiliary references, and information on the forest types present in Snoul (Walston et al. unpublished report), classified data was manually defined and merged into 5 land-cover categories: 1) primary forest; 2) mosaic (i.e. secondary forest/regrowth/scrub); 3) bare or built-up areas; 4) other (i.e. riparian/swamps); and 5) water bodies. Next, cross-tabulation matrices analyzed the intensity of land category change for two time intervals (1990-2001 [Y_t] and 2001-2009 [Y_{t+1}]). First, we analyzed the variation in size of annual rate of change in each time interval (Y_t, Y_{t+1}), comparing observed rates (S_t) to a uniform rate (U) that would exist if annual changes were distributed uniformly across the entire time duration:

$$S_t = \frac{\text{area of change during interval } [Y_t, Y_{t+1}]/\text{area of Snoul}}{\text{duration of interval } [Y_t, Y_{t+1}]} \times 100\% \qquad \text{(eqn. 2)}$$

$$U = \frac{\text{area of change during all intervals/area of Snoul}}{\text{duration of all intervals}} \times 100\% \qquad \text{(eqn. 3)}$$

At the category level, we examined land categories that were relatively dormant or active during land category conversions by comparing the observed intensities of gross gains (G_{tj}) and losses (L_{ti}) for each category with a uniform intensity (S_t) of annual change that would exist if the change during each interval was distributed uniformly across the entire spatial extent.

$$G_{tj} = \frac{\text{area of gross gain of category } j \text{ during}[Y_t,Y_{t+1}]/\text{duration of } [Y_t,Y_{t+1}]}{\text{area of category } j \text{ at time } Y_{t+1}} \times 100\% \quad \text{(eqn. 4)}$$

$$L_{ti} = \frac{\text{area of gross loss of category } i \text{ during}[Y_t,Y_{t+1}]/\text{duration of } [Y_t,Y_{t+1}]}{\text{area of category } i \text{ at time } Y_{t+1}} \times 100\% \quad \text{(eqn. 5)}$$

At the transition level, we calculated whether primary forests or mosaics (i.e. secondary forests/regrowth) were more likely to transition to bare or built-up areas by comparing the observed intensity of each transition (R_{tin}) with a uniform intensity (W_{tn}) that would exist if the change during each interval were distributed uniformly among the available categories.

$$R_{tin} = \frac{\text{area of transition from } i \text{ to } n \text{ during}[Y_t,Y_{t+1}]/\text{duration of } [Y_t,Y_{t+1}]}{\text{area of category } i \text{ at time } Y_{t+1}} \times 100\% \quad \text{(eqn. 6)}$$

$$W_{tn} = \frac{\text{area of gross gain of category } n \text{ during}[Y_t,Y_{t+1}]/\text{duration of } [Y_t,Y_{t+1}]}{\text{area of category } i \text{ at time } Y_{t+1}} \times 100\% \quad \text{(eqn. 7)}$$

Finally, we created kernel density plots to examine whether transitions of primary forest and mosaic to bare or built-up areas occurred close to or further from the road. Kernel density plots are more effective than histograms for examining the distribution of continuous variables such as distance from road, mainly because kernel estimates converge more quickly to true underlying densities (Scott 1979). Land cover classification was carried out using ENVI 4.8 (ITT, Boulder), cross-tabulation matrices were created in IDRISI Selva (Clark Labs, Worcester) and GIS analyses were performed in ArcMap 10.0 (ESRI, Redlands).

Do roads contribute to illegal hunting and wildlife trade?

When collected in a systematic manner, signs of camps and snares targeting mammals can be used to provide empirical evidence of roads contributing to illegal hunting. When the intensity of forest use by mammals targeted by poachers is high along a road, we expect that hunting signs increase with increasing proximity to the road, in part due to the ease of access and convenience of transferring hunted animal products to vehicles along roads. In this case study, we surveyed for hunting signs in forests on either side of State Road 156 in Peninsular Malaysia (Table 1), along which forests are intensely used by two endangered mammal species (see Results). Three temporal replicates of sign surveys were carried out on foot using a cell-based approach and over the dry season (May - Oct 2011). Surveys in each cell covered three habitat types (animal trail, ridge or old logging road) where detection probability of large mammals and hunting signs are likely to be high. Among the three temporal replicates, route overlaps were minimised to achieve spatial independence and greater coverage within each cell. We created kernel density plots (as for forest conversion analyses) to ascertain, in relation to the road, the distribution of hunting signs detected over 131 notionally independent survey routes.

Roads have also been implicated in the illegal trade of mammals and other wildlife. In Vietnam, for example, roads are said to have increased local demand for bushmeat in once remote areas (Long & Hoang 2007), and now serve as trafficking routes to international wildlife markets (Shepherd et al. 2007). Myanmar has also been recognised as a major illegal source of animal parts to consumer and re-export markets in China and Thailand (Martin & Redford 2000; World Bank 2005). With help from the Wildlife Conservation Society (WCS) Myanmar programme, we mapped trading routes in the country, mainly utilising information from hunting and market surveys, interviews with villagers, police and township officials, and field survey data.

RESULTS

Existing roads contributing to forest conversion and illegal hunting

Thirty-six of 45 respondents returned opinions on 16 existing roads covering 10 sub-regions in seven SE Asian countries (Table 1). Images of each road from Google Earth were compiled (Appendix 2), except for Myanmar where data on specific roads were insufficient.

Roads from the Philippines and several Indonesian regions (Java, Irian Jaya and Sulawesi) were not highlighted because of insufficient feedback from experts. A total of 25 endangered mammal species (IUCN categories EN and CR) have been reported to occur in the vicinity of roads identified by our experts – this is around 21% of the total number (117) of endangered terrestrial mammal species known to occur in the represented countries (Table 1). In view of their potential threats, 8 proposed road construction or upgrading projects need to be halted, if not re-routed (Table 2).

***Table 1**. Summary of 16 existing roads contributing to forest conversion of mammal habitats and hunting of endangered mammals according to 36 experts from seven Southeast Asian countries.*

Country (% response)	**Existing road (network)**	**Threatened habitats**	**Endangered mammals recorded (historically and currently) in habitats [citation]**
Cambodia (4/4)	National Highway 4	Kirirom and Bokor NP	Asian Elephant, Banteng, Eld's Deer, Tiger, Pileated Gibbon [1]
	Provincial Road Network 76-141	Eastern Plains Landscape*	Asian Elephant, Banteng, Black-shanked Douc Langur, Eld's Deer, Tiger, Yellow-cheeked Crested Gibbon [2]
	National Road 48	Cardamom Mountains^	Asian Elephant, Dhole, Pileated Gibbon, Tiger [3]
Indonesia			
Kalimantan (5/5)	Bontang-Sangata Road	Kutai NP	Banteng, Bornean Orangutan, Bornean Gibbon [4]
	Balikpapan-Samarinda Road	Bukit Soeharto RF	Bornean Gibbon, Sunda Otter Civet [5]
	Logging road networks	Priority sites for Orangutan conservation#	Banteng, Bornean Orangutan [6]
Sumatra (7/8)	Sanggi-Bengkunat/Krui Liwa Roads	Bukit Barisan Selatan NP	Agile Gibbon, Asian Elephant, Asian Tapir, Siamang, Sumatran Rhino, Tiger [7]
	Blangkejeren-Kutacane Road	Gunung Leuser NP	Asian Elephant, Sumatran Orangutan, Sumatran Rhino, Tiger [8]
	Logging road networks	Tiger conservation landscapes†	Asian Elephant, Sumatran Orangutan, Tiger [9]
Lao PDR (3/3)	Route 9	Phou Xang He and Dong Phou Vieng NBCAs	Asian Elephant, Douc Langur, Giant Muntjac, Tiger [10]
	Route Network 12-1E-8	Nakai-Nam Theun NBCA	Asian Elephant, Dhole, Douc Langur, Giant Muntjac, Yellow-cheeked Crested Gibbon, Saola, Tiger [11]
	Route Network 17A-3	Nam Ha NBCA	Asian Elephant, Black-crested Gibbon, Dhole, Tiger [12]
Malaysia			
East (5/7)	Kalabakan-Sapulut Road	FRs in Tawau and Pensiangan Districts	Asian Elephant, Sumatran Rhino [13]
	Logging road networks	FRs, Kelabit highlands	Banteng, Bornean Gibbon, Sumatran Rhino [14]
	Access roads for dams	Murum, Danum and Pileran Valleys	Bornean Gibbon [15]
Peninsular (7/9)	Federal Route 4	Royal Belum State Park, Temengor FR	Asian Elephant, Asian Tapir, Siamang, Sunda Pangolin, Tiger, White-handed Gibbon [16]
	Federal Route 8	Tamana Negara NP, Titiwangsa Main Range	Asian Elephant, Asian Tapir, Dhole, Siamang, Sunda Pangolin, Tiger, White-handed Gibbon [17]
	State Route T156	Tembat, Petuang and Hulu Telemong FRs	Asian Elephant, Asian Tapir, Dhole, Sunda Pangolin, Tiger, White-handed Gibbon [18]

Myanmar (1/3)	Wildlife trade route network	All mammal habitats in Myanmar	See Results
	Roads in E, W and NW sector	Alaungdaw Kathapa NP	Asian Elephant, Banteng, Dhole, Tiger [19]
	Ledo road	Hukaung Valley WS	Tiger [20]
Vietnam (3/3)	Ho Chi Minh Highway	Protected areas§	Asian Elephant, Delacour's Langur, Northern White-cheeked Gibbon, Red-shanked Douc, Saola, [21]
	Roads in banteng habitats	Ea So, Yok Don and Krong Trai NR, Vinh Cuu NP	Banteng [22]
	Roads in	Cat Tien NP	Asian Elephant, Javan Rhino (hunted to extinction during time of writing) [23]

* Mondulkiri PF, Seima BCA, Lumphat, Snoul, Phnum Prech and Phnum Namlier WS

^ Phnum Samkos and Phnum Aural WS, Central Cardamom PF

Gunung Palung, Danau Sentarum/Bentung Kerihun, Tanjung Puting, Belantikan, Gunung Gajah/Berau/Kelai, Sebangau

† Kerinci Seblat NP, Tesso Nilo and Bukit Tigapuluh landscapes, Bukit Rambang Baling, Kuala Kampar-Kerumutan, Rimbo Panti-Batang Gadu, proHUsed Senepis-Buluhala Tiger National Park

§ Cuc Phuong and Phong Nha-Ke Bang NP, Vu Quang NR

[1] Protected Areas Development (2004); Http 1

[2] Walston et al, unpublished report. A wildlife survey of Southern Mondulkiri Province, Cambodia

[3] Daltry & Momberg (2000)

[4] Wich et al. (2008); Setiawan et al. (2009); MONGABAY.COM (2009)

[5] Yasuma (1994); Oka et al. (2000)

[6] Oranutan Conservation Services Program (2007); Wich et al. (2008)

[7] O’Brien & Kinnaird (1996)

[8] Singleton et al. (2004)

[9] Dinerstein et al. (2006); Eyes on the Forest (2008)

[10] Cleetus (2005)

[11] Timmins & Evans (1996); Timmins & Duckworth (2004)

[12] Tizard et al. (1997); Johnson et al. (2005)

[13] Unet (2009); Ambu et al., unpublished report. Asian Elephant Action Plan Sabah (Malaysia). Sabah Wildlife Department, Kota Kinabalu, Malaysia.

[14] Tajuddin Abdullah et al. (1999)

[15] Then (2009)

[16] Rayan et al. (2012a)

[17] Kawanishi & Sunquist (2004)

[18] Chapter 3

[19] Wildlife Conservation Soceity (2002); Lynam et al. (2009)

[20] Rabinowitz (2004)

[21] Eve et al. (2000); Reuters (2001);

[22] Pedrono et al. (2009)

[23] Polet & Ling (2004); Brook et al. (2012)

NOTE: BCA = Biodiversity Conservation Area; FR = Forest Reserve; PA = Protected Area; PF = Protection Forest; NBCA = National Biodiversity Conservation Area; NP = National Park; NS = Nature Reserve; RF = Recreation Forest; WS = Wildlife Sanctuary

Table 2. Summary of 8 planned road construction or improvement projects that can potentially contribute to forest conversion of mammal habitats and hunting of endangered mammals according to experts from five Southeast Asian countries.

Country	**Planned road construction or upgrading project**	**Threatened habitats**	**Endangered mammals in habitats (and citation)**
Cambodia	Expansion of National Road 48	Phnum Samkos and Phnum Aural Wildlife Sanctuaries, Central Cardamom Protection Forest	Asian Elephant, Dhole, Pileated Gibbon, Tiger [1]
	Expansion of logging road to link National Road 48 and Samkos	Phnum Samkos and Phnum Aural Wildlife Sanctuaries, Central Cardamom Protection Forest	Asian Elephant, Dhole, Pileated Gibbon, Tiger [2]
Indonesia			
Kalimantan	Kalimantan Border Oil Palm Mega-Project	Bentung Kerihun National Park	Bornean Orangutan [3]
	Balang Island Bridge Project	Sungai Wain Protection Forest	Bornean Gibbon, Bornean Orangutan, Bay Cat [4]
Sumatra	Ladia Galaska Scheme	Gunung Leuser National Park	Sumatran Orangutan [5]
Lao PDR	Upgrading of Route 18	Xe Pian National Biodiversity Conservation Area	NA
Malaysia			
Peninsular	Kuala Lumpur Outer Ring Road	Selangor State Park	Asian Tapir [6]
Myanmar	Upgrading of Dawei-Myeik-Kyawthaung Highway	Contributes to Thailand-Myanmar-China smuggling route	Mammals targeted by wildlife trade
Vietnam	Road in northern section of Mondulkiri Protection Forest	Mondulkiri Protection Forest	[7]

[1] Asian Development Bank 2005

[2] Sovan 2008

[3] Wakker 2006

[4] Hance 2010

[5] Gaveau et al. 2009

[6] Http: 2

[7] International Organization for Migration 2009

Roads cut through important mammal habitats

The MaxEnt generated Asian Tapir habitat-suitability map had a mean (SD) AUC score of 0.76 ± 0.02 (Clements et al. 2012). Models with AUC scores above 0.75 are considered potentially useful (Phillips & Dudík 2008). Based on the mean (± SD) logistic value of pixels that roads passed through, all three roads identified by experts in Peninsular Malaysia cut through important habitats (logistic value ≥ 0.45) for the Asian Tapir (Fig. 2): Federal Route 4 (0.50 ± 0.13); Federal Route 8 (0.49 ± 0.08); and State Route T156 (0.51 ± 0.04).

Forest-use intensity maps show that State Route T156 passes through forests that are intensely used by the Asian Elephant ($\hat{\psi}$ ± SE = 0.61 ± 0.11; Fig. 3) and Asian Tapir ($\hat{\psi}$ ± SE = 0.75 ± 0.07; Fig. 4).

According to our logistic regression models, which did not exhibit evidence of over-dispersion, distance to State Route T156 did not have an effect on the habitat use by the two species (Table 3). Therefore, our results suggest that even though the road cuts through forests that are intensely used by these two endangered species, it does not appear to have any negative effect on their habitat use.

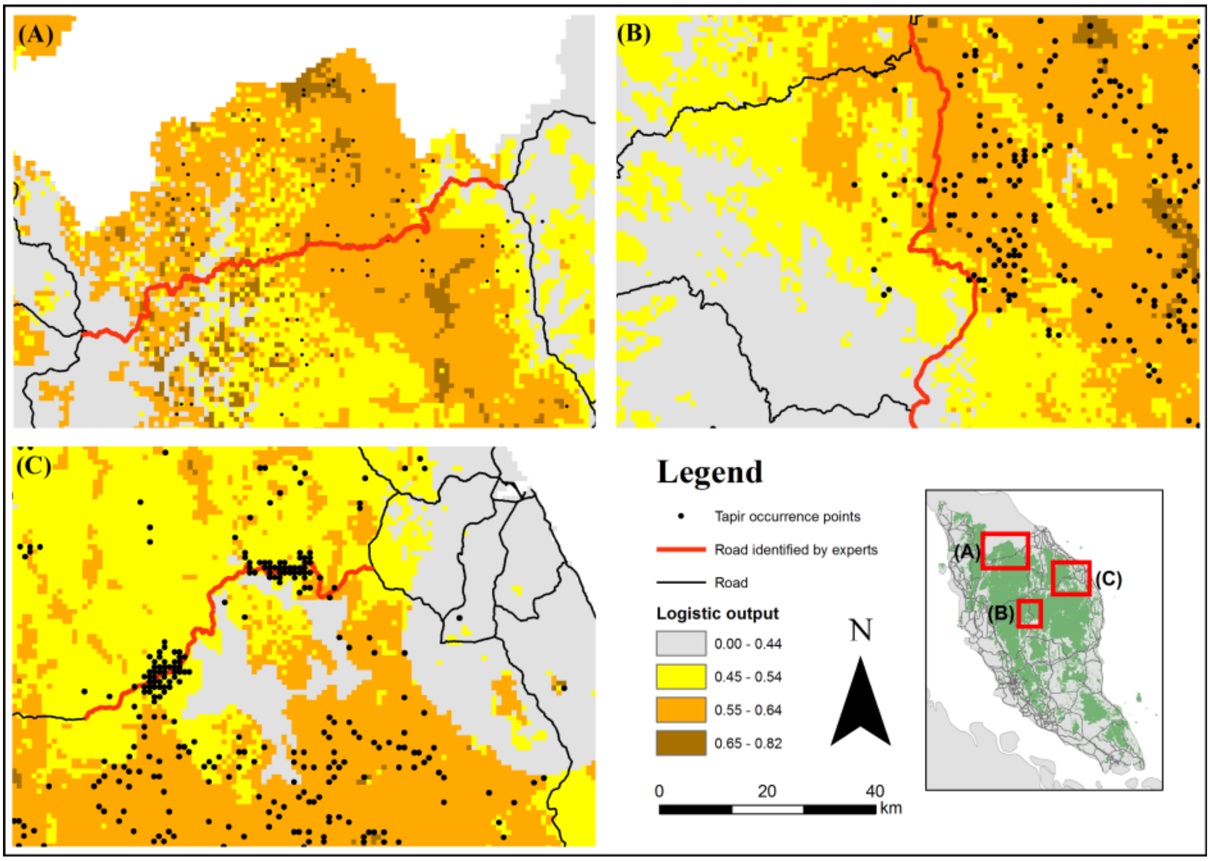

Fig. 2. *Habitat suitability map for the endangered Asian Tapir* (Tapirus indicus) *generated by Maximum Entropy modelling showing how three roads identified by experts in Peninsular Malaysia, (A) Federal Route 4, (B) Federal Route 8 and (C) State Route T156, cut through important habitats (pixels with logistic value ≥ 0.45) for this species. Mean (± SD) logistic value of pixels that were passed through by all three roads are: Federal Route 4 (0.50 ± 0.13); Federal Route 8 (0.49 ± 0.08); and State Route T156 (0.51 ± 0.04). Note: 1) other roads nearby cut through unimportant habitats (logistic value < 0.45) for this species; 2) clustering of presence-only points in State Route T156 is due to intensive sampling (see Chapter 3), but this sampling bias has been accounted for through the use of a bias grid.*

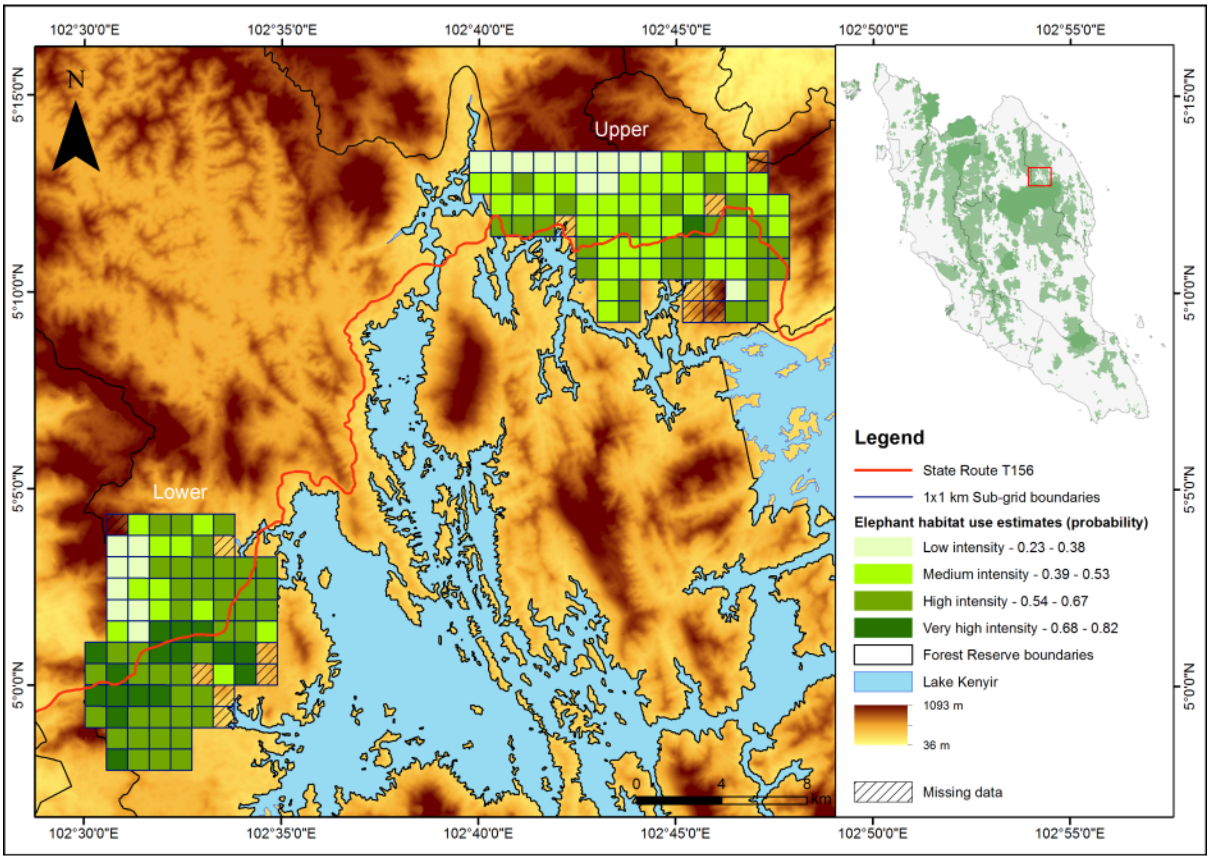

Fig. 3. *Forest-use intensity map for the endangered Asian Elephant* (Elephas maximus), *illustrating whether forests intensely used by this species are bisected by State Route T156 in the State of Terengganu, Peninsular Malaysia. Maps were generated using detection/non-detection data from camera traps analyzed in a likelihood-based occupancy framework. This analysis shows that the road passes through forests used intensively by this species* ($\hat{\psi} \pm SE = 0.61 \pm 0.11$).

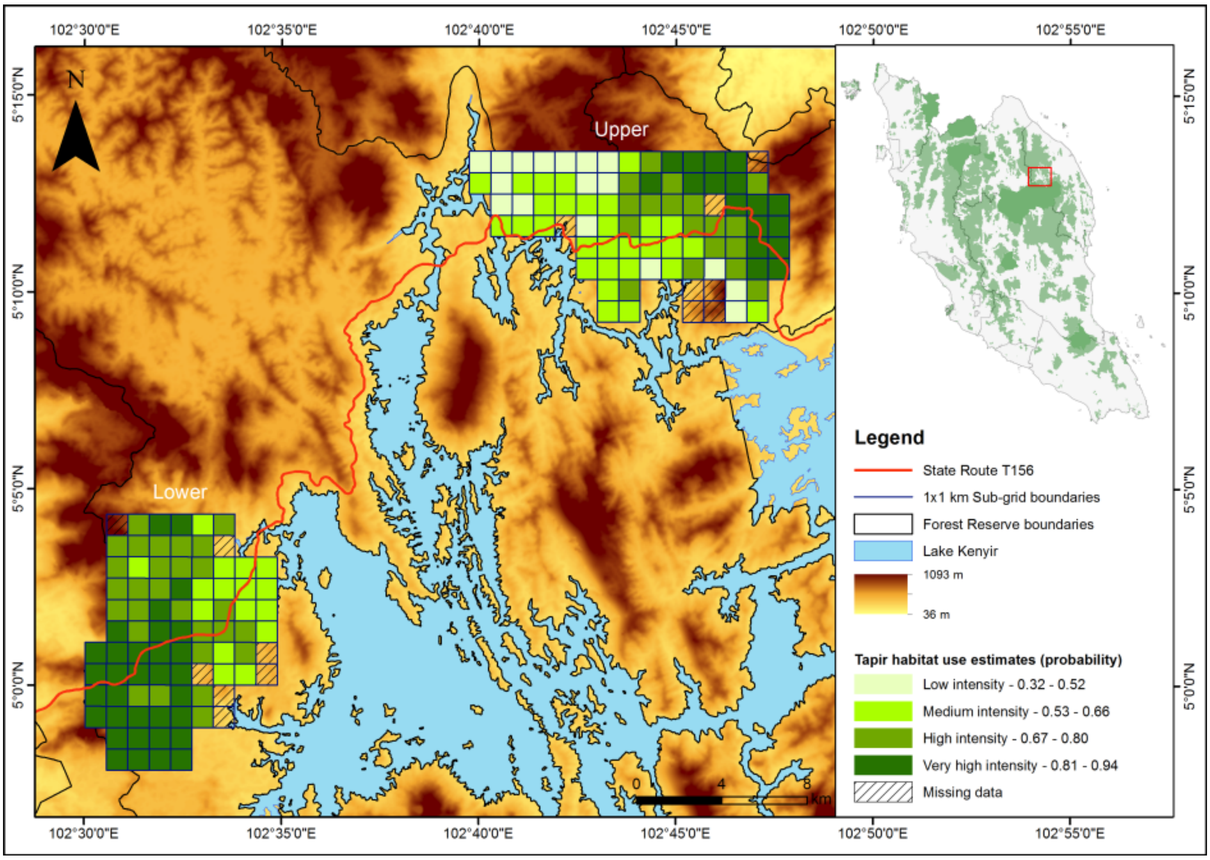

Fig. 4. *Forest-use intensity map for the endangered Asian Tapir* (Tapirus indicus), *illustrating whether forests intensely used by this species are passed through by State Route T156 in the State of Terengganu, Peninsular Malaysia. Maps were generated using detection/non-detection data from camera traps analyzed in a likelihood-based occupancy framework. Our results show that the road passes through forests used intensively by this species ($\hat{\psi} \pm SE = 0.75 \pm 0.07$).*

Table 3. *Logistic regression models examining the effect of four site covariates on endangered Asian Elephant* (Elephas maximus) *and Asian Tapir* (Tapirus indicus) *habitat use (ψ), and three sampling covariates affecting their detection probability (p), based on camera-trap data from forests along State Route T156 in Terengganu, Peninsular Malaysia. Only candidate models with ΔAICc < 2 are shown.*

Candidate models	**AICc**	**ΔAICc**	**wAICc**	**k**	**DE**	**%DE**	$\hat{c}$	**ER**
Asian Elephant								
ψ(.),*p*(blk+trap+rain)	510.52	0.00	0.31	6	497.96	0.00	0.52	1.84
ψ(lake),*p*(blk+trap+rain)	511.74	1.22	0.17	7	496.99	0.19	0.49	
ψ(plan),*p*(blk+trap+rain)	512.49	1.97	0.11	7	497.74	0.04	0.47	
Asian Tapir								
ψ(resv),*p*(trap+rain)	822.17	0.00	0.25	5	811.78	0.45	0.89	1.36
ψ(resv+plan),*p*(trap+rain)	822.79	0.62	0.18	6	810.23	0.64	0.93	
ψ(.),*p*(trap+rain)	823.73	1.56	0.11	4	815.47	0.00	0.92	
ψ(resv+road),*p*(trap+rain)	823.81	1.64	0.11	6	811.25	0.52	0.86	

*Note: Site covariates included in each model are: 1) road = distance to edge of State Route T156; 2) plan = distance to nearest plantation edge; 3) resv = distance to reservoir edge; and 4) fors = forest cover type (as a proxy of logging intensity). AIC_c = Akaike's Information Criterion corrected for small sample size; ΔAIC_c = difference in AIC_c for each model from the most parsimonious model; $wAIC_c$ = AIC_c weight, k = number of parameters; DE = deviance; % DE = % deviance explained in the response variable by the model under consideration; $\hat{c}$ = overdispersion factor. Sampling covariates included in each model are: 1) trap = no. of trap nights that cameras were operational during each sampling occasion; and 2) rain = daily rainfall. * indicates species that had models evaluated based on quasi likelihood Akaike's Information Criterion corrected for small sample sizes ($QAIC_c$) due to evidence of overdispersion.*

Forest conversion can intensify following road construction

For our case study in Snoul Wildlife Reserve, a false colour composite of a Landsat image allowed us to differentiate vegetation from roads and bare or built-up areas (Fig. 5). Furthermore, the 'fish-bone' pattern of arterial roads spawning from the larger Provincial Road 76 was evident, which is typically observed in certain landscapes, such as forest-colonisation projects in the Amazon, where numerous lateral roads are facilitating forest conversion away from a main arterial road.

Based on our calculations using classified Landsat imagery for three different years in Snoul Wildlife Reserve (Fig. 6), the observed gross gain in bare or built-up areas and gross loss of primary forest was greater in the later time interval (2001-2009; during most of the road's existence) than the earlier time interval (1990-2001; mostly during absence of the road). Indeed, the intensity analysis revealed that the annual rate of land category change in Snoul Wildlife Reserve was faster in the later interval than the earlier interval (Fig. 7).

The intensity analysis provided three lines of evidence that forest degradation and loss intensified following road construction. First, in terms of gains at the category level, gains in mosaics and bare or built-up areas were more intense in the later interval than the earlier interval. Second, transitions to mosaics did not target primary forests in the earlier interval (Table 4A), but targeted primary forests in the later interval (Table 4B). Third, transitions to bare or built-up areas, which targeted mosaics in both time intervals, occurred at a much lower intensity in the earlier interval (Table 4C) than the later interval (Table 4D).

Kernel density plots indicate that the road through Snoul Wildlife Reserve appears to be driving forest conversion. For example, most of the transitions of primary forest (Fig. 8A) and mosaics (Fig. 8B) to bare or built-up areas occurred closer to the road. To corroborate expert claims of other roads facilitating forest conversion, we summarised information from journals and grey literature in Table 5.

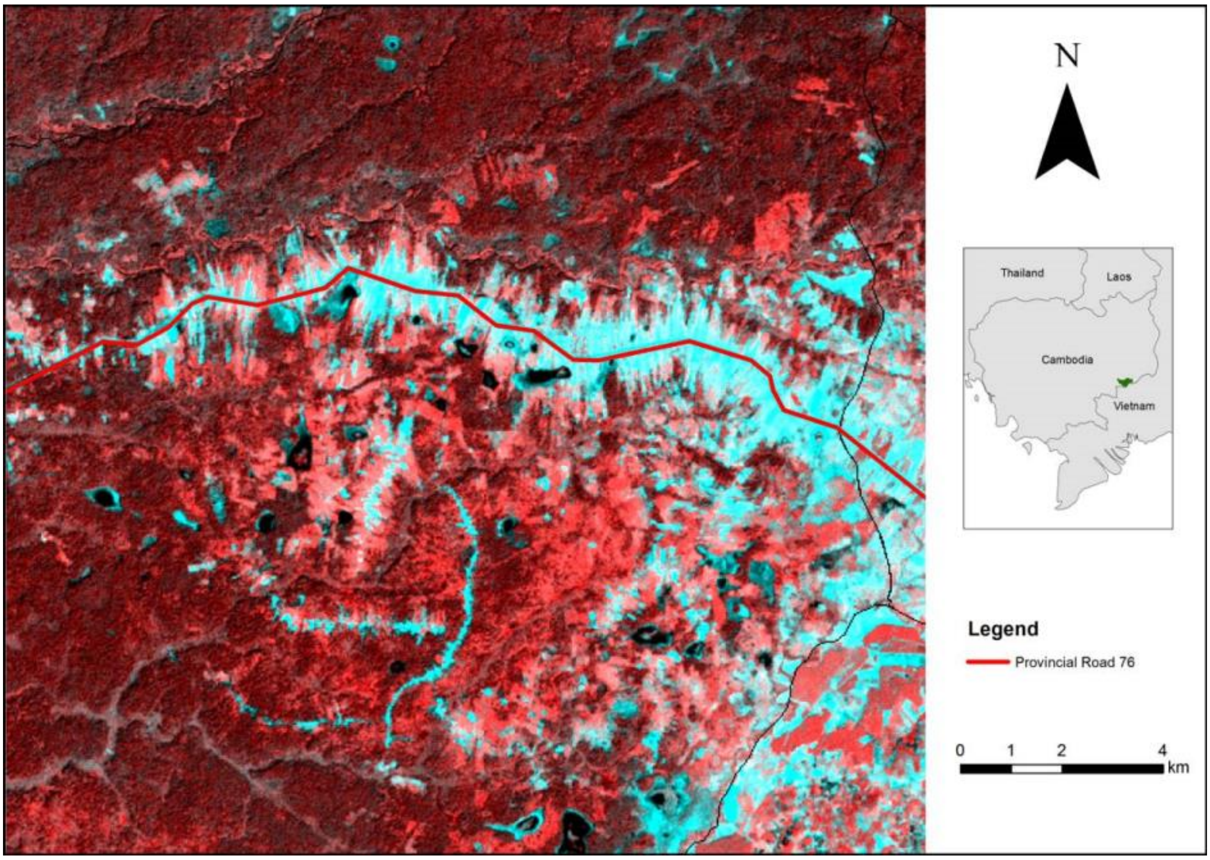

Fig. 5. *A false colour composite of a 2009 Landsat image 5 (TM) depicting a 'fish-bone' pattern of arterial roads spawning from the larger Provincial Road 76 bisecting the Snoul Wildlife Reserve, Cambodia. Landsat Scene Path/Row: 127/52. Acquisition date: 09/12/2009.*

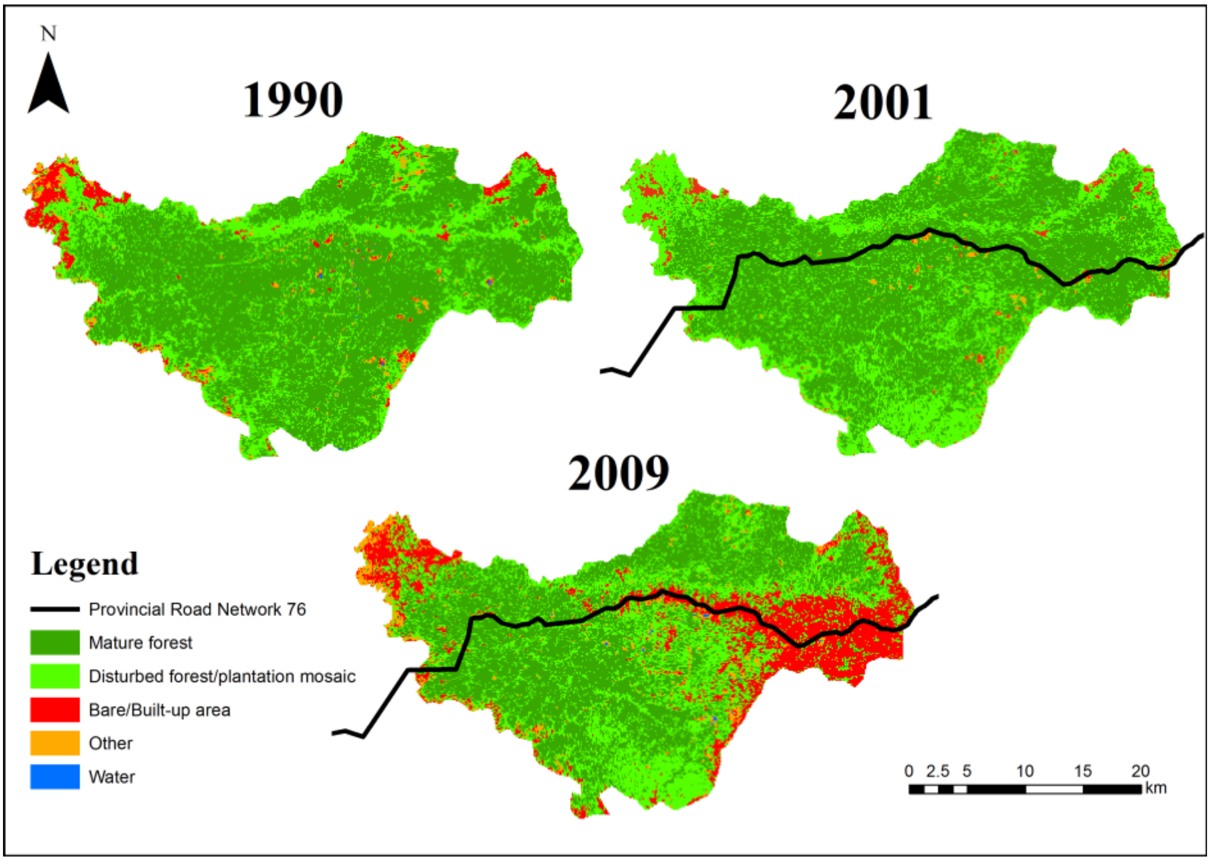

Fig. 6. *Land cover maps of Snoul Wildlife Reserve in Cambodia for three time points when the road was (1) absent (1990), (2) recently completed (2001), and (3) had existed for some time (2009). Observed gross gain in bare or built-up areas and gross loss of primary forests was greater in the later interval (2001-2009) than the earlier interval (1990-2001). Landsat Scene Path/Row: 127/52. Acquisition dates for Landsat 4, 5 (TM) and Landsat 7 (ETM+) images: 27/01/1990; 15/04/2001; and 09/12/2009. Inputs for land cover classification included the first three layers of a Tasseled-cap transformation (Kauth & Thomas 1976) and spectral bands 1-5 and 7. Data layers were processed using an unsupervised classification (ISODATA) algorithm with a maximum class of 200, 50 maximum iterations with a convergence threshold of 0.95. Accuracy analysis was only conducted for the classified image from 2010 using the original Landsat 5 image and a Landsat 7 image from a comparable time period. The overall accuracy of the 2010 image was relatively high at 84.8%. The confusion matrix is provided in Appendix 3.*

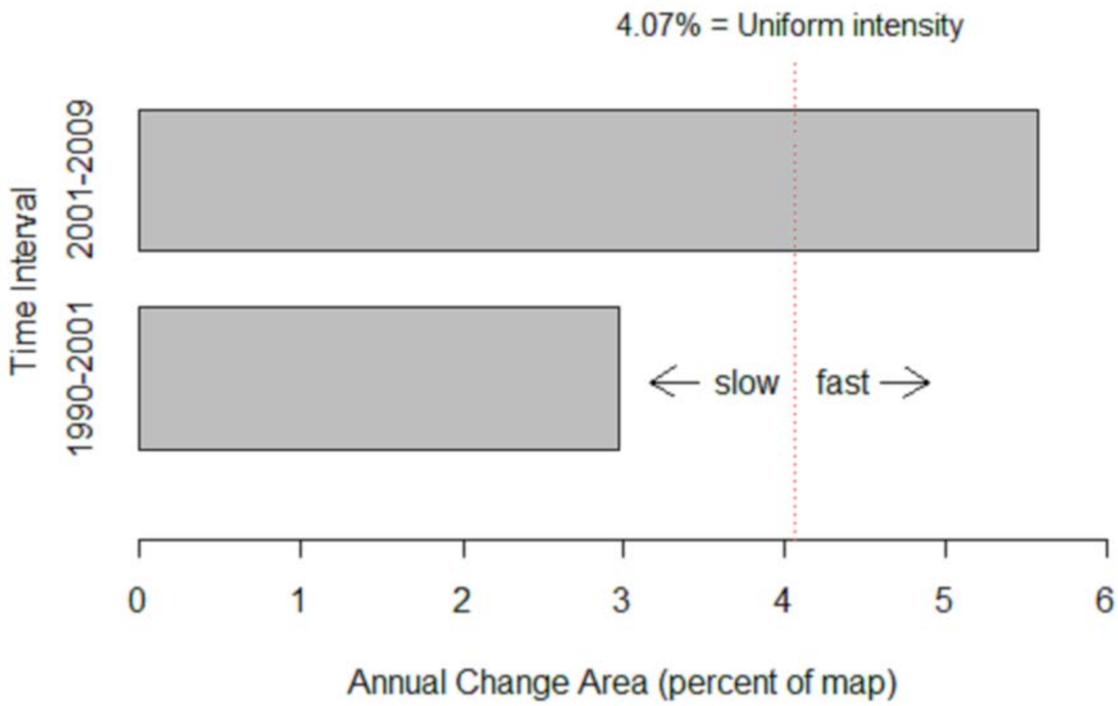

Fig. 7*. Time intensity analysis of land category change in Snoul Wildlife Reserve, Cambodia. Bars show intensity of annual area of change within each time interval: 1) 1990-2001 (mostly during the road's absence) and; 2) 2001-2009 (during most of the road's existence).*

Table 4. *Summary statistics for transition of land categories to mosaic in (A) earlier interval and (B) later interval, and transition of categories to bare or built-up areas in (C) earlier interval and later (D) interval in Snoul Wildlife Reserve, Cambodia. Each row respectively gives: a) category name, b) area of transition in terms of cell counts, c) intensity of transition per gross gain, d) uniform distribution of transitions across the area that is possible for that transition, given the empirical gross gain for mosaic or bare or built-up areas, e) uniform annual transition, f) annual number of pixels of hypothesized error, g) commission or omission intensity in t map and h) hypothesized error as percent of t map.*

(A)	1990 to 2001							
	transitions TO Mosaic							
FROM Category	Observed transition	Intensity of transition	Uniform distribution	Hypothesized annual transition	Annual # of pixels of hypothesized error	Commission intensity	Ommission intensity	Error as % of map1990
Primary forest	835	2.06	2.40	1020	185	0.00	14.80	3.31
Bare or Built-up	150	5.90	2.40	29	121	11.38	0.00	3.31
Other	78	5.91	2.40	15	63	5.91	0.00	3.31
Water	2	3.54	2.40	1	1	0.09	0.00	3.31

(B)	2001 to 2009							
	transitions TO Mosaic							
FROM Category	Observed transition	Intensity of transition	Uniform distribution	Hypothesized annual transition	Annual # of pixels of hypothesized error	Commission intensity	Ommission intensity	Error as % of map1990
Primary forest	1268	3.31	3.20	1214	54	4.21	0.00	0.71
Bare or Built-up	17	1.06	3.20	61	45	0.00	3.34	0.71
Other	8	1.71	3.20	18	9	0.00	0.73	0.71
Water	0	0.00	3.20	0	0	0.00	0.01	0.71

(C)	1990 to 2001							
	transitions TO Bare or Built-up							
FROM Category	Observed transition	Intensity of transition	Uniform distribution	Hypothesized annual transition	Annual # of pixels of hypothesized error	Commission intensity	Ommission intensity	Error as % of map1990
Primary forest	27	0.07	0.12	49	22	0.00	23.34	0.39
Mosaic	24	0.14	0.12	21	3	4.43	0.00	0.39
Other	19	1.41	0.12	1	17	24.17	0.00	0.39
Water	1	2.31	0.12	0	1	1.84	0.00	0.39
(D)	2001 to 2009							
	transitions TO Bare or Built-up							
FROM Category	Observed transition	Intensity of transition	Uniform distribution	Hypothesized annual transition	Annual # of pixels of hypothesized error	Commission intensity	Ommission intensity	Error as % of map1990
Primary forest	771	2.01	2.20	859	89	0.00	6.29	1.19
Mosaic	543	2.56	2.20	451	92	6.95	0.00	1.19
Other	8	1.69	2.20	11	3	0.00	0.22	1.19
Water	0	0.00	2.20	0	0	0.00	0.01	1.19

Note: Coloured cells present the observed intensity in terms of the percent of the category at t *map of each interval, the area of transition within the interval, and the omission or commission errors in* t *map: 1) green cells: indicates that the category targets for that transition; 2) red cells: indicates that the category avoids for that transition; 3)pink: omission error; 4) dark gray: commission error; and 5) black: transition from category to category mosaic or bare or built-up areas.*

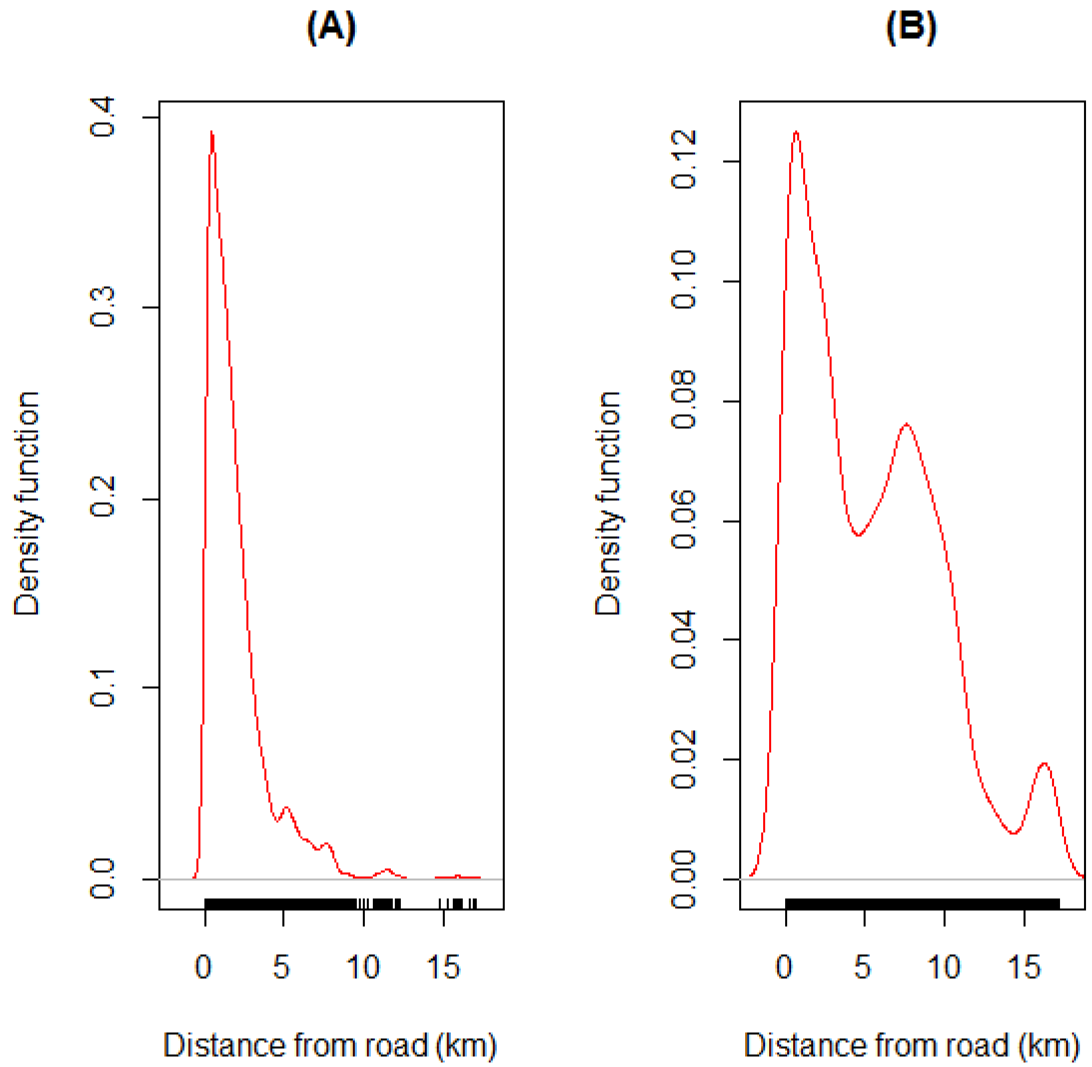

Fig. 8. *Kernel density plots of transitions of (A) primary forest and (B) mosaic categories to bare or built-up areas in relation to distance from Provincial Road 76 bisecting the Snoul Wildlife Reserve, Cambodia.*

Table 5. *Supporting evidence from publications and grey literature corroborating expert claims that roads contribute to forest conversion of habitats where endangered mammals occur in Southeast Asia.*

Country	**Name of road/road network**	**Habitats where endangered mammals occur**	**Supporting evidence for threats from publications and grey literature**
Cambodia	National Highway 4	Kirirom and Bokor NP	Roadside forests vulnerable to illegal logging for firewood, charcoal and timber (Food and Agricultural Organization 1998)
	Provincial Road Network 76-141	Eastern Plains Landscape*	Illegal timber stockpiles found along road bisecting Snoul (Société Générale de Surveillance 2005)
			Villagers paid to drag logs harvested from Snoul to the road side (Global Witness 1999)
	National Road 48	Cardamom Mountains^	Road has intensified illegal logging in neighbouring protected areas (Asian Development Bank 2005)
Indonesia			
Kalimantan	Bontang-Sangata Road	Kutai NP	Road has spawned arterial roads that were utilized by both industrial timber companies and illegal loggers (Jepson et al. 2002)
			Road has intensified forest conversion to plantations (Vayda & Sahur 1996; World Bank 1998)
			Forests along road now dominated by coal mines and oil palm plantations (Setiawan et al. 2009)
	Balikpapan-Samarinda Road	Bukit Soeharto RF	Park has degraded due to road and expected to lose 100% of original forest cover by 2013 (Harris et al. 2008)
	Logging road networks	Priority sites for Orangutan conservation#	Park buffer zones near logging roads suffered higher deforestation rates than those next to paved roads (Curran et al. 2004)
Sumatra	Logging road networks	Tiger conservation landscapes†	49,020 km of logging roads has led to extensive forest loss and degradation (Gaveau et al. 2009)
			Forests along logging roads prone to clearing and forest conversion by villagers for farmlands (Linkie et al. 2004)
Lao PDR	Route Network 17A-3	Nam Ha NBCA	Road has accelerated forest conversion for teak, rubber and sticky rice cultivation (Butler 2009)
Malaysia			
East	Kalabakan-Sapulut Road	FRs in Tawau and Pensiangan Districts	Road has contributed to overland illegal timber traffic out of East Kalimantan (Obidzinski et al. 2007)
	Logging road networks	FRs, Kelabit highlands	300-km logging road by Samling Corporation has threatened to accelerate deforestation (Then 2008)
	Access roads for dams	Murum, Danum and Pileran Valleys	Access road to Murum dam site has resulted in timber extraction from roadside forests (Then 2009)
Vietnam	Ho Chi Minh Highway	Protected areas§	Highway has led to loss of habitat in protected areas beside it (Gray 2009)
	Roads in banteng habitats	Ea So, Yok Don and Krong Trai NR, Vinh Cuu NP	Roads has encouraged human settlement and elevated incidences of logging (Nguyen 2009; Pedrono et al. 2009)

* Mondulkiri PF, Seima BCA, Lumphat, Snoul, Phnum Prech and Phnum Namlier WS

^ Phnum Samkos and Phnum Aural WS, Central Cardamom PF

Gunung Palung, Danau Sentarum/Bentung Kerihun, Tanjung Puting, Belantikan, Gunung Gajah/Berau/Kelai, Sebangau

† Kerinci Seblat NP, Tesso Nilo and Bukit Tigapuluh landscapes, Bukit Rambang Baling, Kuala Kampar-Kerumutan, Rimbo Panti-Batang Gadu, proHUsed Senepis-Buluhala Tiger National Park

§ Cuc Phuong and Phong Nha-Ke Bang NP, Vu Quang NR

NOTE: BCA = Biodiversity Conservation Area; FR = Forest Reserve; PA = Protected Area; PF = Protection Forest; NBCA = National Biodiversity Conservation Area; NP = National Park; NS = Nature Reserve; RF = Recreation Forest; WS = Wildlife Sanctuary

Roads contribute to illegal hunting and trade of wildlife

For our case study at State Route T156, our indirect sign surveys recorded a total of 125 encroachment camps and 131 snares in the forests on either side of the road. All hunting signs were likely to be foreign (based on the language of discarded cigarette boxes, tree markings and personal encounters) and hence, were illegal. Kernel density plots revealed that detections of camps (Fig. 9A) and snares (9B) were higher nearer to the road than further from it. In total, we recorded at least 43 access trails lead from the road leading into the forest.

Information from the WCS Myanmar programme verified that road networks are facilitating illegal trade of mammals at a national level. Specifically, routes from sources to trade centres, and trade centres to borders, were identified. At the Thai-Myanmar border, parts of at least 187 bears and 1158 felids were recorded between 1999 and 2006 at border markets such as Three Pagoda Pass and Tachilek (Fig. 10; Zaw 2005; Shepherd & Nijman 2008). Improved road links across the border and upgraded highways, such as those connecting Mandalay, Lashio and Muse cities (Fig. 10), have increased access by traders to lucrative border markets in China (Shepherd & Nijman 2007). Because of the poaching and harvesting of prey species for trade, the country's tiger population has been depleted to less than 150 individuals (Lynam et al. 1999; Lynam 2003). Suppressing illegal trade in tigers and their prey species are now key priorities for recovering the species in Myanmar (Lynam et al. 2006). To corroborate expert claims of other roads facilitating illegal hunting and trade, we summarised information from publications and grey literature in Table 6.

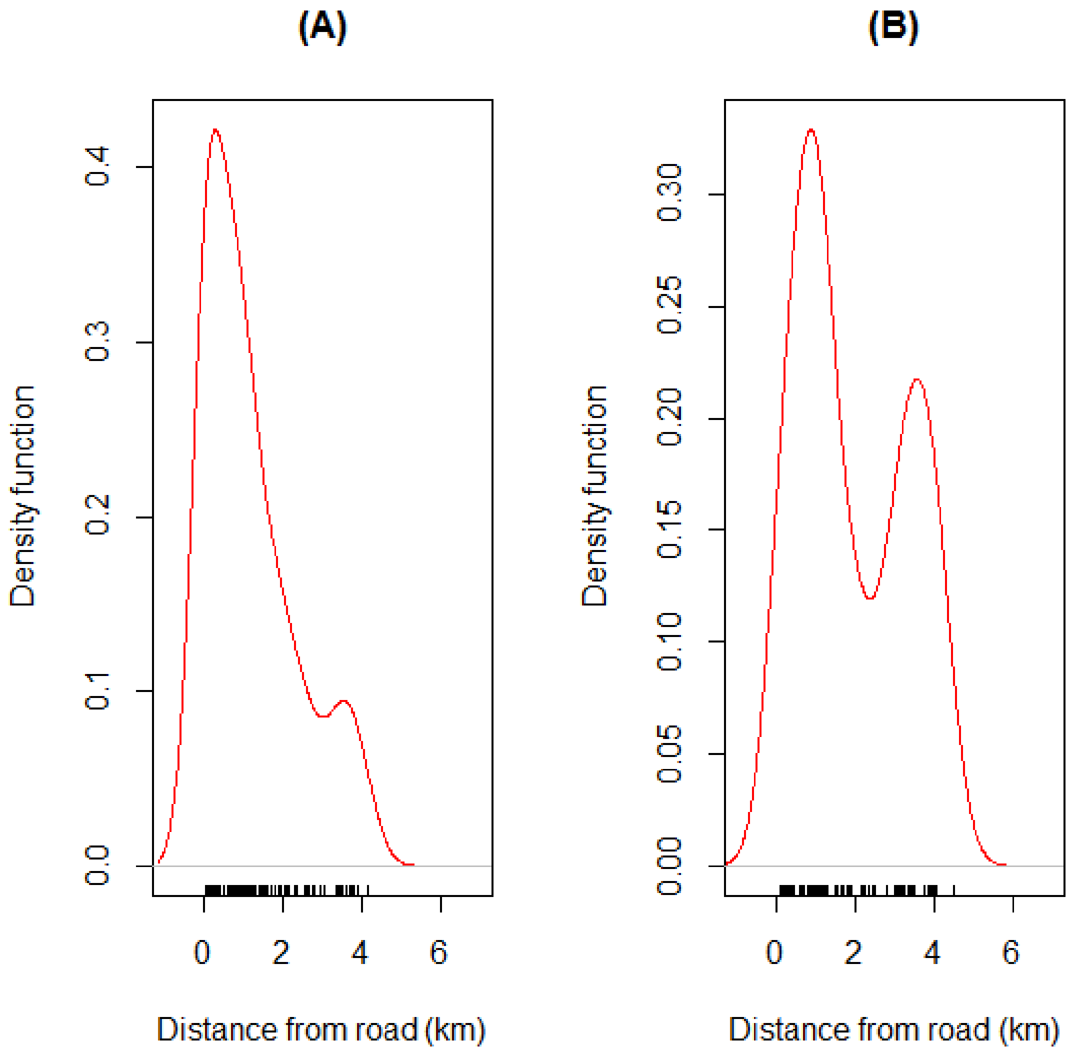

Fig. 9. *Kernel density plots of detections of (A) encroachment camps and (B) snares in relation to distance from State Route T156 cutting through forests in the State of Terengganu, Peninsular Malaysia.*

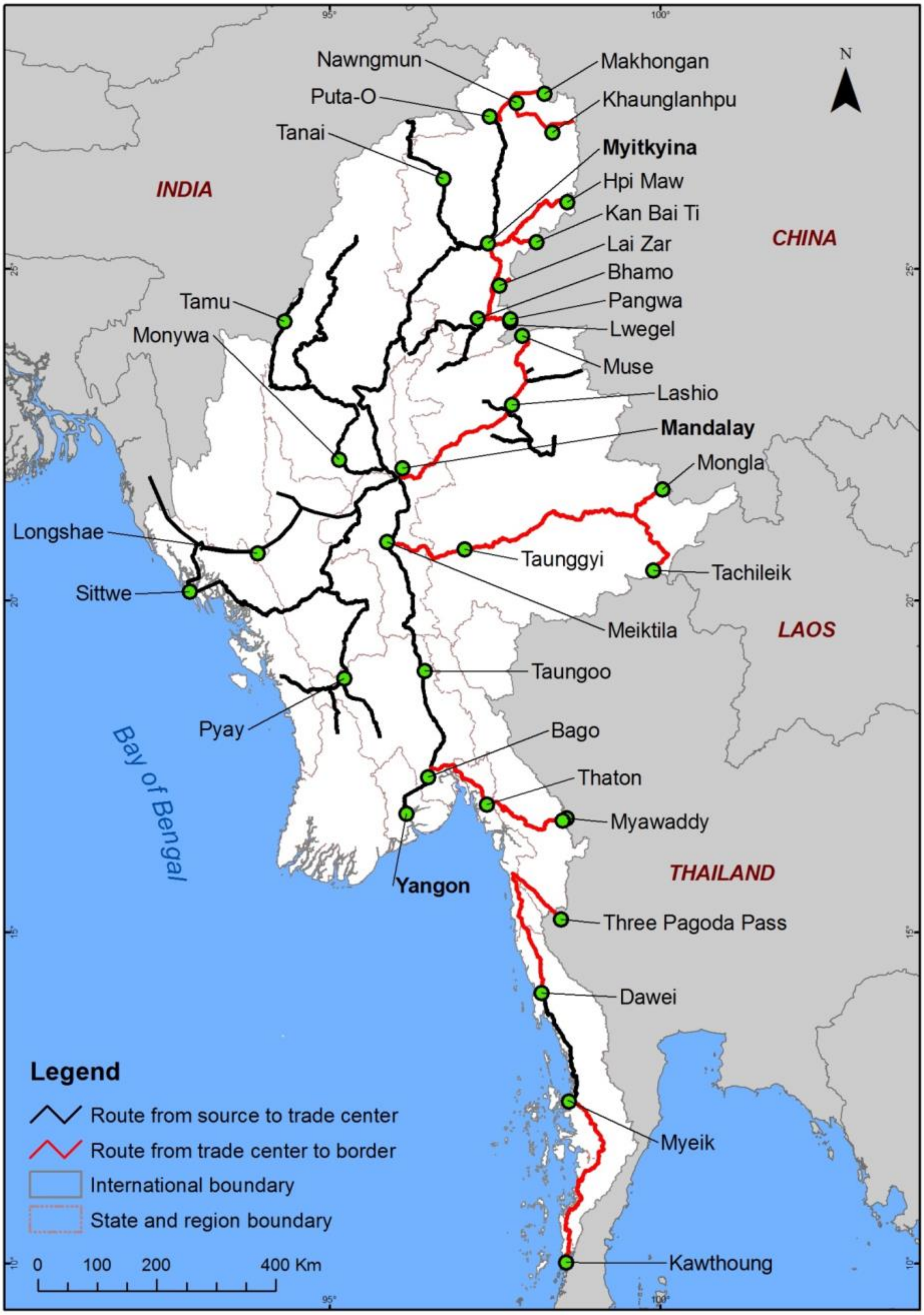

***Fig. 10**. Map of road networks in Myanmar functioning as conduits for the illegal trade of wildlife to border towns (circles) in other neighbouring countries. Source: Antony Lynam.*

***Table 6**. Supporting evidence from publications and grey literature corroborating expert claims that roads contribute to illegal hunting and trade of wildlife in Southeast Asia.*

Country	**Name of road/road network**	**Habitats where endangered mammals occur**	**Supporting evidence for threats from publications and grey literature**
Cambodia	National Highway 4	Kirirom and Bokor NP	Road has provided access into Kirirom to illegally hunt mammals (Khim & Talyor-Hunt 1993, Asian Development Bank 2005)
			Game is illegally sold along the road (Martin & Palmer 2008)
	Provincial Road Network 76-141	Eastern Plains Landscape*	Increased encounter rate of hunting signs along extension of road through the Seima (WCS unpublished data 2004 – 2012)
Indonesia			
Sumatra	Sanggi-Bengkunat/Krui Liwa Roads	Bukit Barisan Selatan NP	Roads has provided access to poachers, who are removing Sumatran tigers from the park each year (O'Brien et al. 2003)
	Blangkejeren-Kutacane Road	Gunung Leuser NP	Roads hsa contributed to hunting in park (Singleton et al. 2004), and declines of Sumatran Orangutans (van Schaik et al. 2001)
	Logging road networks	Tiger conservation landscapes†	Logging road networks has contributed poacher access into Sumatran Tiger habitats (Gillison 2001)
			Logging highways has increased levels of human and Sumatran Tiger conflicts (Eyes on the Forest 2008)
Lao PDR	Route 9	Phou Xang He and Dong Phou Vieng NBCAs	Road upgrade has increased the risk of cross-border trafficking of mammals (Asian Development Bank 2008)
	Route Network 12-1E-8	Nakai-Nam Theun NBCA	Road has threatened Saola and Douc Langurs, particularly through poaching for illegal trade (Timmins & Duckworth 2004).
Malaysia			
East	Kalabakan-Sapulut Road	FRs in Tawau and Pensiangan Districts	One Sumatran rhinoceros individual poached along road after it became habituated to presence of people (Unet 2009)
	Logging road networks	FRs, Kelabit highlands	Logging roads from Pan Borneo Highway allowed poachers to access Bornean Pygmy Elephant habitats (J Payne pers. comms.)
			Main Line West logging road has increased poaching threat to Bornean Pygmy Elephants (ST Wong, pers. comms.)
Peninsular	Federal Route 4	Royal Belum State Park, Temengor FR	Roadside patrols removed snares, rescued a snared tiger and arrested poachers (New Straits Times 2009; Clements et al. 2010a)
			More snares per unit survey effort were detected closer to road than forest interior (MD Rayan, unpublished data)
	Federal Route 8	Tamana Negara NP, Titiwangsa Main Range	Road re-alignment to within 2 km of national park has provided greater access for poachers (Sharma 2009)
	State Route T156	Tembat, Petuang and Hulu Telemong FRs	See Results
Myanmar	Wildlife trade route network	All mammal habitats in Myanmar	See Results

	Ledo road	Hukaung Valley WS	Road has provided unrestricted access to poachers, who actively supply wild meat to local markets (Rabinowitz 2004)
Vietnam	Ho Chi Minh Highway	Protected areas§	Highway has led to hunting in adjacent protected areas (Gray 2009), especially to Saola populations (Stone 2006)
			Arterial roads branching from highway function as conduits for illegal wildlife trade (World Bank 2005; Shepherd et al. 2007)
	Roads in banteng habitats	Ea So, Yok Don and Krong Trai NR, Vinh Cuu NP	Roads has increased extirpation risk of Banteng due to increased accessibility for hunters (Nguyen 2009)
	Roads in	Cat Tien NP	Roads has provided access to poachers targeting mammals such as the Javan rhino (Polet & Ling 2004)

* Mondulkiri PF, Seima PF, Lumphat, Snoul, Phnum Prech and Phnum Namlier WS

† Kerinci Seblat NP, Tesso Nilo and Bukit Tigapuluh landscapes, Bukit Rambang Baling, Kuala Kampar-Kerumutan, Rimbo Panti-Batang Gadu, proHUsed Senepis-Buluhala Tiger National Park

§ Cuc Phuong and Phong Nha-Ke Bang NP, Vu Quang NR

NOTE: BCA = Biodiversity Conservation Area; FR = Forest Reserve; PA = Protected Area; PF = Protection Forest; NBCA = National Biodiversity Conservation Area; NP = National Park; NS = Nature Reserve; RF = Recreation Forest; WS = Wildlife Sanctuary

DISCUSSION

To our knowledge, this is the first study to identify roads that are endangering mammals and their habitats in Southeast Asia. We corroborated expert claims of the negative impacts of roads with diverse evidence from the literature, and empirically demonstrated how roads contribute to forest conversion and illegal hunting and wildlife trade.

Before suitable measures to mitigate the environmental impacts of roads can be adopted, conservation planners and practitioners must better understand the nature of road development in their respective countries.

Drivers of road construction

Roads are not always built to benefit society, as is often claimed. While the expansion of road infrastructure has alleviated poverty in many countries (Jones 2006), surveys in Lao PDR revealed that the poorest rural residents ranked the value of roads or access to markets as only 8^{th} out of 12 potential measures to improve their income levels (Government of Laos 2000). Their income levels are typically too low to afford the supplies that roads bring into their areas (Robichaud et al. 2001).

Indeed, road development projects are sometimes initiated with questionable benefits that result in collateral environmental damage. In Lao PDR, almost two-thirds of timber supplies over the last five years have come from clearances associated with development projects that include road construction (International Centre for Environmental Management 2003). In Sumatra, the Governor of Aceh pushed for more proposed roads through the Leuser ecosystem under the expanded Ladia Galaska road scheme, putatively to decrease transportation time of timber and agricultural commodities and free enclaved villages from isolation (Gaveau et al. 2009). However, critics argue that financial benefits would only be reaped by security forces and local elites from illicit business opportunities (Singleton et al.

2004), rather than providing a net benefit to local communities (Robertson 2002; van Beukering et al. 2003).

Socio-political factors also pose a serious challenge to opposition of roads on environmental grounds. For example, the Ladia Galaska road scheme is largely supported by the Achenese people, not only because it would greatly improve intra-provincial transport efficiency (especially for agricultural commodities such as palm oil; Gaveau et al. 2009), but also because they would be less reliant on roads going through neighbouring provinces (M. Linkie, personal communication).

Ultimately, government financial capacities may determine whether a road threatens biodiversity. In Vietnam, the Ho Chi Minh Highway is now regarded as the 'single largest long-term threat to biodiversity' in the country (Gray 2009). Before its construction, an option of diverting it around Vietnam's oldest national park was rejected by the government to avoid costs of $20 million to resettle 900 households (Reuters 2001). Under rare circumstances, an economic crisis might even help abate the impacts of roads on biodiversity. During the financial crisis in 1998, for example, the Indonesian government cut back on funds for the construction and maintenance of major highways, causing delays of up to seven years in some road projects in Kalimantan (Sunderlin 2002).

Road impacts vary

The degree to which a road affects its surrounding biodiversity can vary depending on its age. In some sub-regions in Southeast Asia, paved roads are no longer a contemporary biodiversity threat. In Sabah, for example, the threats of roads to biodiversity were more apparent during periods of massive land conversion in the 1990s, but now these same roads are surrounded by largely depauperate oil palm plantations (J Payne, personal communication). The same phenomenon can be seen in Thailand, where the national road network has been established now for many years and most roadside forests are now largely devoid of endangered mammals (R. Steinmetz, personal communication).

Instead, logging road networks can be more detrimental to biodiversity in Southeast Asia than in other tropical regions such as the Amazon, where selective logging occurs at a low intensity and roads are sparse (Nepstad et al. 2001; but see Redford 1992). In Malaysian Borneo, for example, satellite images revealed that a total of 364,489 km of logging roads was built between 1990 and 2009 (Bryan et al. 2013). Throughout most of Kalimantan, logging roads are considered the primary cause of most deforestation problems in protected areas instead of paved roads (Curran et al. 2004), with logging-road densities of up to 0.242 km/km^2 in West Kalimantan (Appendix 4; Fig. 2C) compared with paved road densities of 0.0015 km/km^2. By increasing forest access and creating much dry, flammable slash, logging also appears to increase forest fires; 76% of 258 fire-prone zones in Kalimantan contained logging roads (Steenis & Fogarty 2001).

Sometimes, roads can contribute to forest conversion further away from them. In East Kalimantan, in order to escape detection from police and forestry officials, people migrated away from the Balikpapan-Samarinda Road into the Bukit Soeharto Recreation Forest to illegally clear land for pepper plantations (Vayda & Sahur 1996).

In rare instances, road development may even be used as a wildlife conservation strategy. In Vietnam, the widening of a road near Cat Tien National Park was deemed an appropriate measure to discourage elephant movement to areas where they could potentially be killed in human-dominated landscapes (Varma et al. 2008).

Road mitigation strategies for non-governmental conservation planners and practitioners

What can conservation planners and practitioners do to minimise the negative impacts of roads on endangered mammals?

(1) *Increase engagement with stakeholders responsible for road development*. Thus far, agencies responsible for road development are rarely included as main project partners in species conservation plans (Department of Wildlife and National Parks 2008; Ministry of Forestry 2007). Because roads can be the precursor of forest conversion and hunting, it would be wise to include road-relevant stakeholders in the early stages of conservation planning. In Sumatra, timely discussions by the Wildlife Conservation Society (WCS) with villagers and local government officials prevented a road from cutting through Bukit Barisan Selatan National Park (Wildlife Conservation Society 1999). In the long run, such engagements can facilitate greater transparency and improved lines of communication between protected area managers and road authorities. Such communication gaps are common in countries such as Lao PDR, where heads of protected areas are rarely consulted before nearby roads are constructed (Robichaud et al. 2001). It is unsurprising that state government infrastructure projects are one of the key drivers of deforestation in northern Lao PDR (Travers et al. 2011).

(2) *Negotiate for greater enforcement effort along existing roads cutting through endangered species habitats*. Along Federal Route 4 in Peninsular Malaysia (Table 1; Fig. 2), government enforcement agencies stepped up their roadside patrols in response to the large number of snares detected by the World Wide Fund for Nature (WWF)-Malaysia's patrols (A. Zafir, personal communication). In Lao PDR, road check points were recommended by the Wildlife Conservation Society (WCS) as a vital measure to curb tiger poaching and illicit trade in ungulate prey species inside core tiger conservation areas found on either side of Route 1C bisecting the Nam Et-Phou Loeuy National Biodiversity Conservation Area (Johnson et al. 2004).

(3) *Call for environmental and social impact assessments to be audited and made transparent to the public*. Regional impact assessments should be conducted for major roads and highways, while smaller roads should not be spared from assessments even if funds are

constrained (ICEM 2003). Unfortunately, impact assessments for forest clearance projects are not mandatory, and are mostly weak in Southeast Asia (Quintero et al. 2010), while negative impacts of road construction highlighted in impact assessments rarely deter projects from going ahead. For example, most of the proposed roads in the Ladia Galaska scheme have not undergone Environmental Impact Assessments (EIAs), and those that did flouted regulations nonetheless (Robertson 2002). In Lao PDR, the upgrade of Route 3 proceeded even after warnings from consultants about the negative impacts of the road construction (Marris et al. 2002).

(4) *Raise public awareness of the environmental impacts of existing and proposed road projects*. In Kalimantan, roadside campaigns to raise awareness of fire prevention and suppression (Solichin 2002) indirectly helped to prevent further loss of fire-prone mammal habitats. In Peninsular Malaysia, increasing media attention given to the poaching issues along Federal Route 4 (NST 2009; NST 2011) helped galvanise more support from enforcement agencies (TRAFFIC 2011). In Sumatra, media campaigns by NGOs convinced donor agencies such as the World Bank to discontinue financial assistance to the Indonesian state budget to prevent misuse of funds in road expansion projects such as the Ladia Galaska road scheme (Down to Earth 2002). However, heightened awareness may not always reap immediate dividends. Banks continue to finance road projects in the Greater Mekong sub-region even though their own evaluation reports acknowledge that transnational roads contribute to human and wildlife trafficking (Asian Development Bank 2008).

Road mitigation strategies for government agencies

Ultimately, lobbying efforts by conservation planners and practitioners can only go so far without political will. What measures can governments undertake to mitigate the impacts of roads on endangered mammal habitats?

(1) *Maintain and improve forest connectivity on either side of existing roads.* In Cambodia, the preservation of forests on both sides of Provincial Road 48 and 76 was highlighted as a key strategy to ensure the dispersal of arboreal species such as the Yellow-cheeked Crested Gibbon (*Nomascus gabriellae*; Channa & Gray 2009). The integration of green infrastructure options (e.g. underpasses, overpasses, road signs and culverts) into proposed road designs, along with measures in place to evaluate their efficiency of use (Chapter 5), may also be beneficial for the movement of mammals through fragmented habitats (Goosem et al. 2001; Quintero et al. 2010; van der Grift et al. 2013).

(2) *Strengthen efforts against wildlife poaching and trafficking along roads, particularly those leading to border checkpoints.* In Myanmar, we have identified places where additional law enforcement effort is needed (Fig. 10). In Lao PDR, roads in general have been blamed for increasing the burden on protected area staff to combat increased hunting pressure from locals, foreigners and road construction crews (Robichaud et al. 2001). Therefore, external agencies such as customs and immigration departments should be solicited to aid wildlife departments in the arrest of suspects at these border checkpoints. Furthermore, their personnel should be sufficiently equipped with wildlife species identification and enforcement skills (Shepherd & Nijman 2008).

(3) *Improve sustainable forest management regimes in selectively logged forests to minimise threats from logging roads*. To reduce hunting pressure in old logging concessions, closing (blocking or destroying) logging roads after cessation of logging can facilitate migration of wildlife and minimise access for poachers and illegal loggers (Laurance 2001;

Meijaard et al. 2004; Linkie et al. 2008; Meijaard & Sheil 2008). Most importantly, forestry departments should prioritise the closure of logging roads that contribute to the transport of illegally harvested timber; this is especially important at the Malaysian-Indonesian boundary on Borneo where satellite images have detected 137 points with cross-border logging road intrusions (Obidzinski et al. 2007). When new logging roads are constructed through previously undisturbed mammal habitats, greater law enforcement must be afforded for newly accessible resources (International Centre for Environmental Management 2003), together with publicised policies and measures that deter workers from poaching (Quintero et al. 2010).

(4) *Resolve land rights and tenure prior to road construction*. One of the key drivers of habitat loss is the absence of land and resource tenure along roads. This has resulted in an uncontrolled influx of locals seeking to clear and claim land along the roads (Asian Development Bank 2005). To minimise illegal settlements along roads bordering important biodiversity areas, road projects should complete the allocation of alternate lands for villages prior to road construction.

(5) *Integrate road planning across relevant government agencies*. Ad hoc planning with little or no cross-sectoral communication between governmental departments is often the root of environmental problems associated with roads. In Lao PDR, an Environment Unit has encouragingly been created within the Department of Roads to ensure environmental concerns are considered in road construction programmes (International Centre for Environmental Management 2003). In Malaysia, the Department of Wildlife and National Parks laudably worked together with the Public Works Department to incorporate underpasses along a new highway to facilitate mammal migration in important wildlife corridors (Chew 2007). However, multi-agency road planning must take place at appropriate government levels. For instance, conservation and development plans in Lao PDR should be

developed at provincial rather than local levels as most threats to protected areas, especially road construction, are likely to originate from the former (International Centre for Environmental Management 2003).

(6) *Conduct projections of economic and biodiversity loss prior to road development*. In Sumatra, the government plans to expand the Ladia Galaska road scheme, an all-weather road network in Aceh. However, it is feared that this road development will further reduce and fragment mammal populations (Caldecott & Miles 2005), especially two of the three largest remaining Orangutan populations (Wich et al. 2008). Indeed, a study projected that the total economic value of the Leuser ecosystem under selective use would be greater than a 30-year deforestation scenario (van Beukering et al. 2003), which would certainly be realized under an expanded Ladia Galaska road scheme cutting through the protected area. Predictive models have also shown that forest areas near roads are highly vulnerable to deforestation, with areas at high risk of deforestation ($p > 0.8$) predicted to increase by 40% (Fig. 11; Gaveau et al. 2009). Furthermore, Orangutan habitat is predicted to further decline by 16% (1,137 km^2) in 2030, resulting in the loss of an estimated 1,384 individuals (or 25% of the current global population; Gaveau et al. 2009). Such projections can help guide decision-making involving road planning.

(7) *Explore compensation schemes that can minimise the need for, or impact of proposed roads in important biodiversity areas*. Inter-governmental REDD (Reduced Emissions from Deforestation and Degradation) projects, such as the recent Norway-Indonesia pact (Clements et al. 2010b), has probably helped prevent the construction of new logging roads through peat swamps and natural forests. Governments can also make it compulsory for financial lending institutions to implement carbon-deposit and refund systems such as that developed by Reid (2013). Under this mechanism, a road developer is obliged to buy credits equal to the net carbon emissions expected from deforestation along an existing or

proposed road. These credits then serve as deposits over fixed periods. At the end of each period, the road developer is allowed to sell credits equal to the difference between expected and actual deforestation – this means the developer would redeem all deposits on all the forest maintained intact and retire remaining credits to cover deforestation that actually occurred. According to Reid (2013), one advantage for the developer is that there is a conservation incentive beyond the construction phase to avoid all deforestation because forest conversion, as we have shown, can intensify after a road is built. If the developer has taken steps to minimise deforestation along the road, the developer will financially benefit from rising market prices for carbon in the long run. At the same time, financing governments should conduct due-diligence studies prior to a road project overseas. In Lao PDR, for example, it was unlikely that the Australian Government was aware of the potential environmental consequences from the rehabilitation project along Route 9 (Asian Development Bank 2010). If a road must be built, offset mechanisms should be explored such as Payment for Ecosystem Services (PES). In Lao PDR, the economic value of an area in Nakai-Nam Theun Protected Area (Appendix 4; Fig. 4C) that was inundated by a hydro-electric dam project was offset by a contribution of US$31.5 million to create a management authority (Quintero et al. 2010). However, the effectiveness of these funds has come under intense scrutinity from both conservation and development agencies (AJ Lynam personal observation).

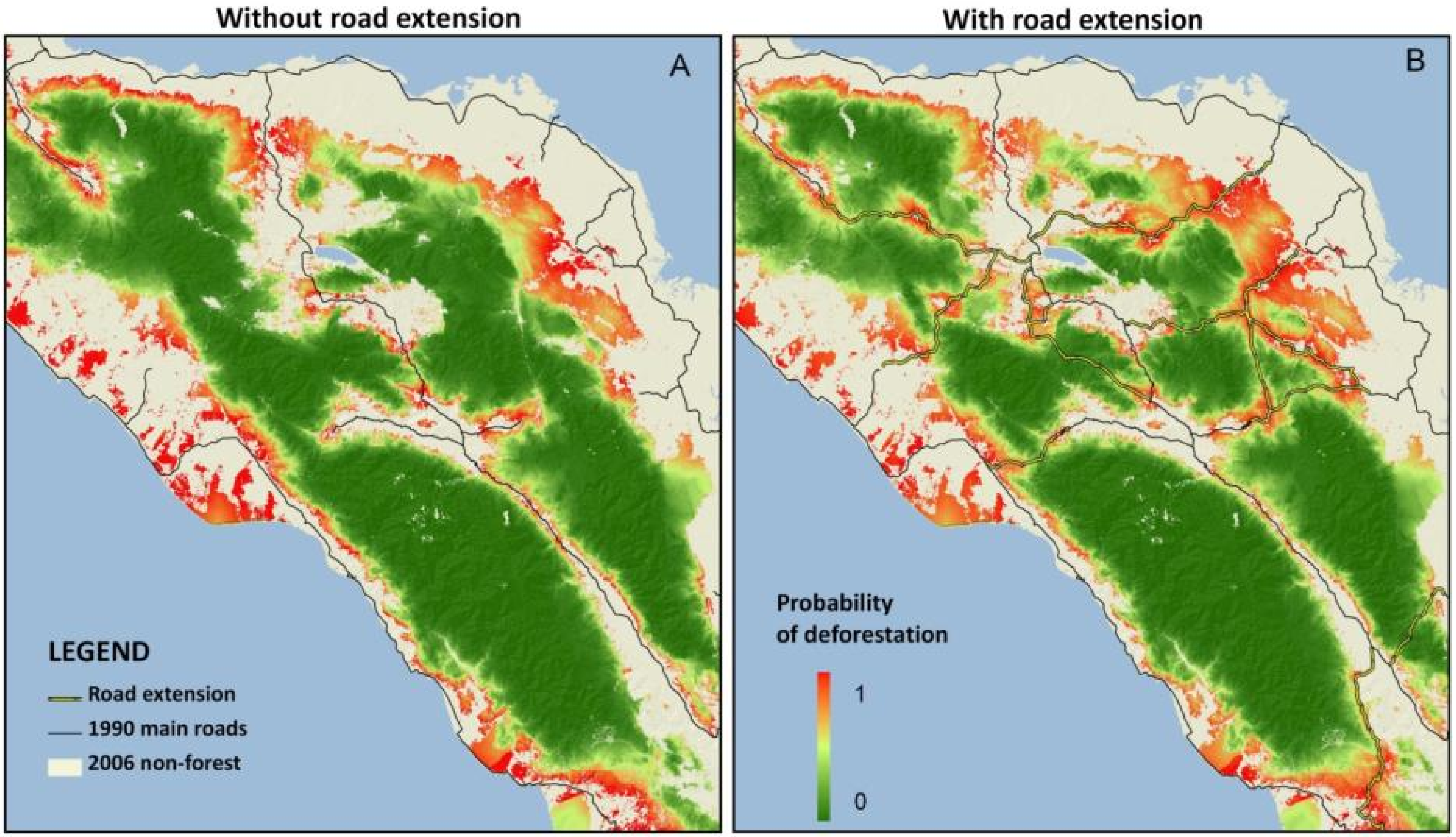

Fig. 11. *Probability map of deforestation (A) without further Ladia Galaska road extension, and (B) with road extension. Source: Gaveau et al. 2009,'The future of forests and Orangutans* (Pongo abelii) *in Sumatra: predicting impacts of oil palm plantations, road construction, and mechanisms for reducing carbon emissions from deforestation', Environmental Research Letters, vol. 4, no. 3, p. 034013.*

CONCLUSIONS

With the help of experts, we now know where existing and proposed roads are endangering mammals in Southeast Asia. Efforts should be made to stop or re-route proposed roads that are potentially detrimental to biodiversity. Indeed, there is precedence for proposed roads to be rerouted in regions such as Kalimantan and Sumatra (Wildlife Conservation Society 1999; Jepson et al. 2002). At existing roads, implementation of mitigation strategies should ideally be focused on roads that pass through habitats with the highest number of threatened mammal species with the best chances of population recovery (i.e. species at tipping points), especially when conservation resources are limited. Measuring the 'distance' of a species to extinction, however, continues to pose a challenge in conservation planning.

End of chapter 2

JCU-affiliated papers by GRC <u>cited</u> in this chapter:

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Chapter 3: The SAFE index: using a threshold population target to measure relative species threat

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JCU-affiliated papers by GRC <u>published</u> from this chapter:

1. **Clements, GR**, Bradshaw, CJA, Brook, BW & Laurance, WF 2011, 'The SAFE index: using a threshold population target to measure relative species threat', *Frontiers in Ecology and the Environment*, vol. 9, no. 9, pp. 521-525.
2. Bradshaw, CJA, **Clements, GR**, Brook, BW & Laurance, WF 2011, 'Better SAFE than sorry'. *Frontiers in Ecology and the Environment*, vol. 9, no. 9, pp. 487-488.

INTRODUCTION

Conservation biologists have long studied the processes underlying species' extinctions and have sought to devise ways to prevent or mitigate extinctions resulting from human impacts. Recent debates over the likely magnitude of the current extinction crisis have largely focused on the proportion of all species that could disappear during this century (e.g. Brook et al. 2006a; Laurance 2007; Bradshaw et al. 2009). However, species' extinctions due to anthropogenic factors are just the endpoint conservationists wish to avoid. Today, many species are declining across large swathes of their former geographic ranges, and some species' populations are becoming so seriously diminished in numbers that they are less likely to withstand random catastrophes (Ewens et al. 1987) or maintain their original functional roles in ecosystems (Larsen et al. 2005) and their evolutionary potential (Franklin & Frankham 1998).

Earlier terms describing the imperiled status of species that had undergone major declines include the *living dead* (Janzen 1986) and *extinction debt* (Tilman et al. 1994), both of which embody the notion of short-term persistence but a long-term consignment to extinction. *Local extinction* or *extirpation* describes the loss of local populations (Laurance 1991; Pimm & Askins 1995), but typically has a narrow frame of reference, such as a particular island or habitat fragment. The concept of *ecological extinction* was coined in reference to the reduction of a species to such low abundance that it "no longer interacts significantly with other species" (Estes et al. 1989), but determining the critical threshold-abundance values for specific species can be impractical.

Some claim that population extinctions (extirpations) are more useful proxies of diminishing biological capital than are species extinctions (Ceballos & Ehrlich 2002), especially when it can take a long time for threatened species to be recognized as officially extinct (i.e.

failure to detect the species despite years of searching; McInerny et al. 2006). Here, my collaborators and I advocate the use of a more heuristic measure of relative threat that describes a *Species' Ability to Forestall Extinction*, or the SAFE index:

$$\text{SAFE index} = \log_{10}(N) - \log_{10}(\text{MVP}_t) \qquad \text{(eqn. 8)}$$

where N is the species' population estimate throughout the species' known range (ie all populations combined) and MVP_t is an empirically supported threshold MVP target, which is currently set at 5000 individuals according to median demographic and genetic estimates of minimum population-size requirements among widely different taxonomic groups (Brook et al. 2006b; Traill et al. 2007, 2010). On precautionary grounds, we suggest using the lower confidence-limit estimates of N, and the upper confidence-limit for MVP size, where such estimates exist for the species of interest and are considered statistically robust (Traill et al. 2010).

One might argue that a numerically-explicit measure of biodiversity loss already exists in the form of percentage range loss, an index used by Ceballos and Ehrlich (2002) to compare historical and present distributions of 173 declining mammal species across six continents. We therefore investigated whether our SAFE index can better predict relative species threat (according to the IUCN Red List) than does percentage range loss.

METHODS

We constructed binary and ordinal logistic regressions to determine which of the two metrics, the SAFE index or percentage range loss, better predicts the IUCN threat categories of mammal species for which extant population sizes were available (95 of 173 species from Ceballos & Ehrlich [2002]) on the Red List website (International Union for the Conservation of Nature 2013). We extracted percentage range-loss data (current range area/original range area; Appendix 5) from Ceballos and Ehrlich (2002). Our binary responses consisted of “threatened” and “near/not threatened” after pooling four (“Extinct”, “Critically Endangered”, “Endangered”, and “Vulnerable”) and two (“Near Threatened” and “Least Concern”) IUCN threat categories, respectively. Our ordinal responses consisted of six IUCN threat categories, ranked according to their indicative risk levels (ie “Extinct” to “Least Concern”). In the binary logistic regression, we fitted generalized linear models (GLMs) using the R statistical environment 3.0.0 (R Development Core Team 2010), assigning to candidate models a binomial distribution and logit link function. To control for phylogenetic relatedness, we also fitted generalized linear mixed-effect models (GLMMs) to the data using *ORDER* as a random effect (Bradshaw and Brook 2010). For the ordinal logistic regression analysis, we used the *polr* function (implemented in the MASS library of the R package), which fits a proportional-odds logistic regression model to an ordinal factor response. We calculated the relative likelihoods and weights of models using Akaike’s Information Criterion (AIC) corrected for small sample sizes (Burnham and Anderson 2002). We compared relative statistical evidence among models using the information-theoretic evidence ratio (*ER*), which is the AIC_c weight of one model divided by another. The *ER* is a concept akin to Bayesian odds ratios (McCarthy 2007) and is preferable to null-hypothesis testing because the likelihood of the alternative model is explicitly evaluated (Bradshaw and Brook 2010). For each model, we also calculated the percentage deviance explained (%DE) as a measure of goodness-of-fit, and compared each model’s %DE to determine the proportion of variance in the response attributable to each predictor.

RESULTS

We provide SAFE indices for 95 mammal species in Appendix 5. Using a MVP target of 5000 individuals (Traill et al. 2010) on a logarithmic scale, we calculate that an extinct species would have a SAFE index of –3.7 (i.e. assuming “extinction” equates to $N = 1$ because $\log_{10}[0]$ is unresolvable: Fig. 12). Such a non-linear scale is particularly beneficial for the management of species with low population sizes, because slight population fluctuations will result in acute changes in SAFE indices that can help trigger urgent conservation interventions (e.g. Javan and Sumatran Rhinos; Fig. 12). Negative SAFE indices indicate that a species is below the threshold MVP target of 5000 individuals (e.g. if $N = 4000$, then SAFE index = –0.1), whereas positive SAFE indices indicate the species is above that threshold (e.g. if $N = 6000$, then SAFE index = 0.08).

If taxon-specific SAFE indices incorporating population and MVP-size uncertainties are desired, then species abundance estimates (N) can be substituted with lower and upper confidence-limit estimates (e.g. 1996 and 2447 for Grevy’s Zebra [*Equus grevyi*], respectively; Appendix 5), whereas the generalized threshold MVP target (MVP_t) of 5000 individuals can also be replaced by the lower and upper 95% confidence limits of taxon-specific MVP thresholds (e.g. 2261 and 5095 for mammals, respectively; Traill et al. [2007]; Fig. 12). To incorporate these differences, we calculated three additional variants of the SAFE index, to represent a greater range of uncertainty (Appendix 5); as before, we fitted both GLMs and GLMMs to these indices, to determine their relative capacity to predict Red List threat categories for mammals.

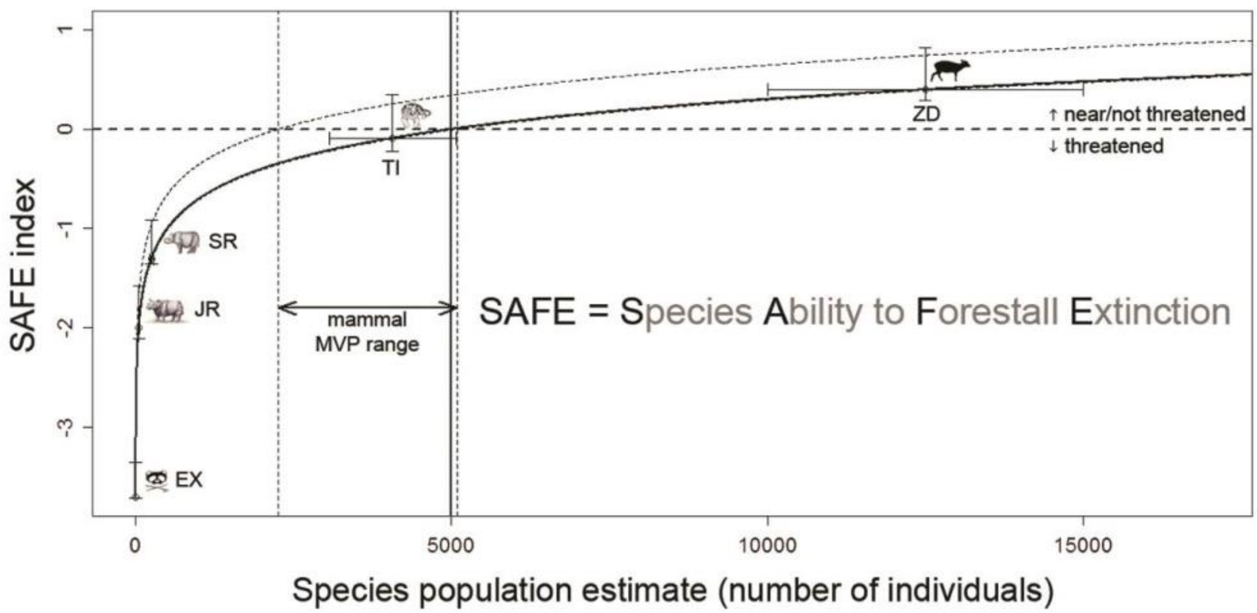

Fig. 12. *Plots of SAFE indices against species population estimates with: (1) an empirically supported threshold minimum viable population (MVP) target (solid line and curve; 5000 individuals according to Traill* et al. *[2010]); and (2) lower and upper 95% confidence limits of mammal-specific MVP thresholds (dashed lines and curves; 2261 and 5095 individuals, respectively, according to Traill* et al. *[2007]). An extinct species (EX), the Javan Rhinoceros (JR;* Rhinoceros sondaicus*), Sumatran Rhinoceros* (*SR;* Dicerorhinus sumatrensis)*, Tiger* (*TI;* Panthera tigris)*, and Zebra Duiker* (Cephalophus zebra) *are highlighted (with vertical and horizontal confidence intervals) to illustrate their decreasing relative threat and increasing potential for long-term persistence (from left to right).*

Binary logistic regression revealed that our SAFE index is a better predictor of mammal IUCN threat categories than is percentage range loss (i.e. the former had higher model weights and described ~60% of the deviance, as compared with only ~17 % for the latter; Table 7). Despite including *ORDER* as a random effect, GLMM results were similar: model weights were identical and the %DE shifted only slightly (Table 7). The model with our SAFE index also had far higher bias-corrected support relative to the model, with only percentage range loss (*ER* = 2.61×10^{10} times providing as much support). Similarly, ordinal logistic regression showed that the SAFE index was a better predictor of relative species threat than percentage range loss; the former had a higher model weight (0.97 versus 0.03) and explained a higher percentage of deviance in the probability of being threatened (6% versus 4%; %DE values here are lower than those in the binomial models because the variance is spread over more IUCN threat categories in the ordinal regression). GLMs and GLMMs showed that the three uncertainty variants of the SAFE index were still far better predictors of mammal threat status than was percentage range loss, but still did not outperform (in terms of %DE) the original SAFE index based on an MVP value of 5000 individuals (Appendix 6).

Table 7. *Generalized linear model (GLM) and generalized linear mixed-effect model (GLMM) sets used to examine the relationship between the probability (Pr) of a species being threatened for 95 mammal species and predictors.*

Model	***k***	***–LL***	**ΔAIC$_c$**	***w*AIC$_c$**	**%DE**
GLM					
Pr(*threat*) ~ *SAFE index*	2	–22.57	0.00	1.00	59.5
Pr(*threat*) ~ *% range loss*	2	–46.37	47.59	0.00	16.8
Pr(*threat*) ~ *1*	1	–55.75	64.28	0.00	0.00
GLMM					
Pr(*threat*) ~ *SAFE index* + (1|*ORDER*)	3	–20.93	0.00	1.00	59.7
Pr(*threat*) ~ *% range loss* + (1|*ORDER*)	3	–45.16	48.44	0.00	13.1
Pr(*threat*) ~ *1* + (1|*ORDER*)	2	–51.99	59.97	0.00	0.00

Notes: *Only single-term models were considered to test the relative ability of the SAFE index versus percentage range loss in predicting extinction threat. The analytical theme represented by each model (SAFE index, % range loss, the intercept-only model, and ORDER as a random effect), and the information-theoretic ranking of models investigating the predictors of mammal IUCN threat categories according to Akaike's Information Criterion corrected for small sample size (AIC_c) are shown. k = number of parameters, –LL = maximum log-likelihood, ΔAIC_c = difference in AIC_c for each model from the most parsimonious model, $wAIC_c$ = AIC_c weight, and %DE = percent deviance explained in the response variable by the model under consideration. Two data points were removed for the GLMM because there was only one representative species in its respective Order: Riverine Rabbit* (Bunolagus monticularis) *and Asian Elephant* (Elephas maximus).

DISCUSSION

The SAFE index is attractive for at least three reasons. First, it has a far superior ability to predict IUCN threat categories, as compared with the percentage range loss of a species. Second, it does not rely on the difficult-to-obtain demographic data needed to construct detailed population viability analyses necessary for predicting extinction risk. Finally, it leverages some recent meta-analyses on the MVP size estimates for well-studied groups (Traill et al. 2007).

On the basis of numeric, meta-analytic, and genetic evidence, MVP estimates (standardized to a time scale of 40 generations and 99% persistence probability) show marked consistency among taxa whose populations range around 5000 adult individuals (Traill et al. 2007, 2010). Whether practitioners choose this standard MVP value and a simple median population-size estimate for target species, or instead elect to use more conservative values, the inherent uncertainties must be acknowledged. SAFE indices for taxonomic groups such as invertebrates should also be treated with caution, as population size per se *may* be less important in determining their extinction risk than the number of populations and the dispersion of those populations. Regardless, the SAFE index provides a more meaningful and fine-grained interpretation of the relative threat of species extinction than do the IUCN threat categories alone. The IUCN has yet to base its threat categories on predictions from population viability analyses because of inadequate data or models for most listed species (Traill et al. 2010).

We believe that the SAFE index could serve as a quantitative measure of relative threat status that can be more readily understood by the general public, donors, and policy makers, who may not appreciate the need to consider population viability in conservation and who do not understand the IUCN categorical classifications. For example, the Asian Elephant (*Elephas maximus*) has a SAFE index of 0.92 (N = 41 410), whereas the index for Tigers (*Panthera tigris*) is –0.21 (N = 3062). Although both species are classified as "Endangered" (International Union for the Conservation of Nature 2013), the latter arguably warrants more urgent conservation attention (Clements et al. 2010a). However, this does not necessarily mean we should reduce efforts to protect endangered species with positive SAFE indices, such as the Asian elephant, because other threats such as population fragmentation and poaching may be higher for certain species.

More than half (57%) of all mammal species in our analyses appear to be at vulnerability thresholds, or "tipping points", with SAFE indices between 1 and –1 (Fig. 13). Donors with limited resources might wish to focus on such species; the tiger, for instance, has a SAFE index ranging from –0.21 to 0.35 (Figs. 12 and 13). Roughly one-quarter of the species in our analysis are very close to extinction, with SAFE indices below –2 (Fig. 13). Under such desperate circumstances, those considering conservation triage (Walker 1992) might elect to channel resources toward species such as the Sumatran Rhinoceros (*Dicerorhinus sumatrensis*) rather than the precarious Javan Rhinoceros (*Rhinoceros sondaicus*); these species have SAFE indices of –1.36 and –2.10, respectively (Fig. 13).

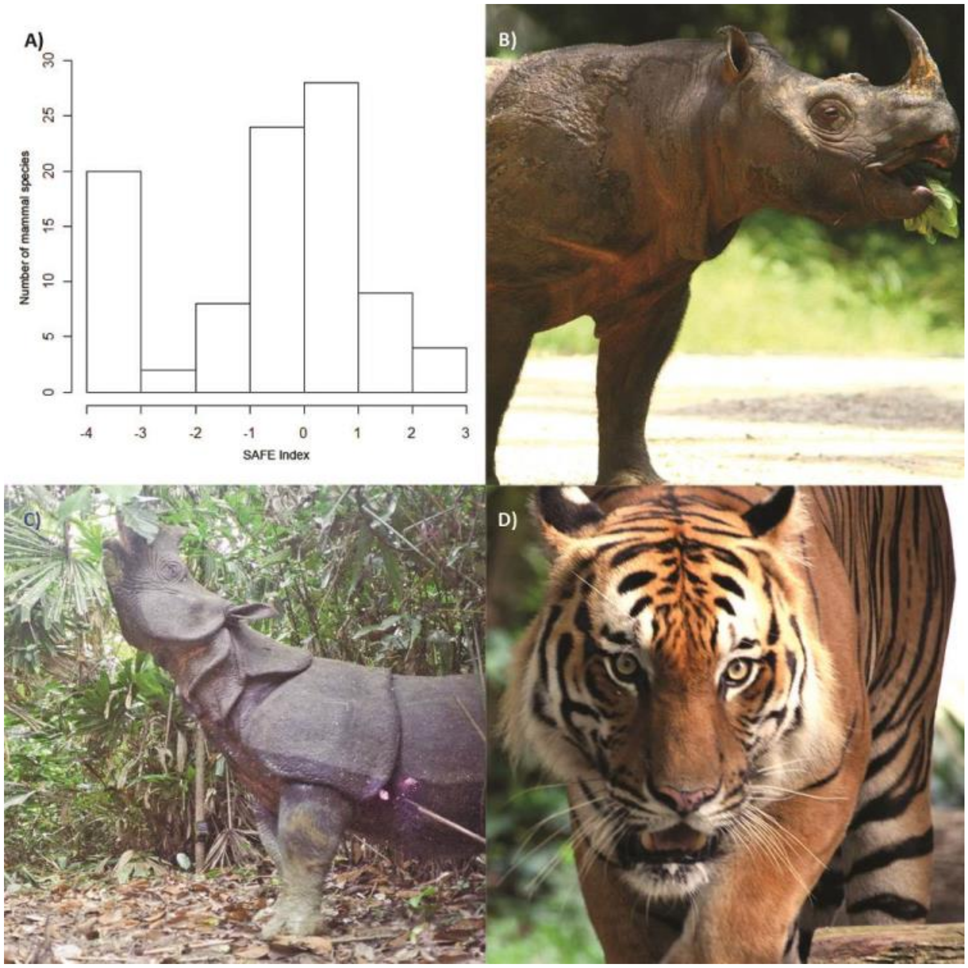

Fig. 13. *(A) Histogram of SAFE indices across the 95 mammal species in our analysis, indicating ~23% close to extinction (i.e. SAFE indices < –2) and ~ 57% at "tipping points" (i.e. SAFE indices between 1 and –1). Practitioners of conservation triage may want to prioritise resources on (B) the Sumatran Rhinoceros* (Dicerorhinus sumatrensis) *instead of (C) the Javan Rhinoceros* (Rhinoceros sondaicus) *(–1.36 versus –2.10). Alternatively, donors with limited resources may want to channel their conservation efforts toward (D) the Tiger (–0.21 to 0.35) or other species at "tipping points".*

Applying the SAFE index

As discussed in Chapter 1, being able to differentiate how much more one species is endangered than other may be challenging for conservation planners, especially when there is no numerically-explicit metric accompanying an IUCN threat category.

To help decisions on which road (Table 1) to focus mitigation measures at, we calculated SAFE indices for IUCN Critically Endangered or Endangered mammal species that likely to occur along each road (Appendix 7). Based on the indices, we identified the road(s) that warrant priority conservation attention based on their passage through habitats containing the most number of species at ‘tipping points’ (SAFE indices raning from -1 to +1; Clements et al. 2011). If there were ties in the number of species, both roads were chosen.

Our prioritistion exercise (Table 8) identified one road/road network in each country that should be prioritised for the implementation of mitigation strategies (see Chapter 2). One advantage of using the SAFE index for this prioritisation exercise is that it considers the distance of the *entire* species from extinction across it range states. Alternatively, SAFE indices can be recalculated using country-specific minimum-viable population estimates of each endangered mammal species (i.e. carrying capacities in different forests) for a country-level prioritisation exercise, but such data are usually unavailable.

Table 8. *Summary of road(s) in Southeast Asia that warrant priority conservation attention (X) based on their passage through habitats with the most number of species whose popuylations are at tipping points (SAFE indices ranging from -1 to +1)*

Country/ Subregion	**Name of road/road network**	**Priority**	**SAFE indices (in ascending order) of mammals recorded around road**
Cambodia	National Highway 4		-0.70, -0.17, 0.00, 0.82, 0.92
	Provincial Road Network 76-141	X	-0.70, -0.17, 0.00, 0.32, 0.92, 0.93
	National Road 48		-0.30, -0.17, 0.82, 0.92
Indonesia			
Kalimantan	Bontang-Sangata Road		0.95, 1.70
	Balikpapan-Samarinda Road		1.70, NA
	Logging road networks	X	0.00, 0.95
Sumatra	Sanggi-Bengkunat/Krui Liwa Roads	X	-1.36, -0.51, -0.17, 0.04, 0.65, 0.92
	Blangkejeren-Kutacane Road		-1.36, -0.17, 0.16, 0.92
	Logging road networks		-0.17, 0.16, 0.92
Lao PDR	Route 9		-0.17, 0.92, 0.93, NA
	Route Network 12-1E-8	X	-0.82, -0.30, -0.17, 0.32, 0.92, 0.93, NA
	Route Network 17A-3		-0.59, -0.30, -0.17, 0.92
Malaysia			
East	Kalabakan-Sapulut Road	X	-1.36, 0.92
	Logging road networks	X	-1.36, 0.00, 1.70
	Access roads for dams		1.70
Peninsular	Federal Route 4		-0.51, -0.17, 0.45, 0.65, 0.92, NA
	Federal Route 8	X	-0.51, -0.30, -0.17, 0.45, 0.65, 0.92, NA
	State Route T156		-0.51, -0.30, -0.17, 0.45, 0.92, NA
Myanmar	Wildlife trade route network		See Results
	Roads in E, W and NW sector of Alaungdaw Kathapa NP	X	-0.30, -0.17, 0.00, 0.92
	Ledo road		-0.17
Vietnam	Ho Chi Minh Highway	X	-1.47, -1.40, -0.82, 0.92, NA
	Roads in banteng habitats		0.00
	Roads in Cat Tien NP		0.92

Better SAFE than sorry

The SAFE index (Clements et al. 2011) has attracted interest from our peers (see Akçakaya et al. 2011; Beissinger et al. 2011; McCarthy et al. 2011). Nevertheless, their critique has further emphasised the need for a more heuristic measure of species extinction threat. The main points of contention can be summarised as follows: (1) SAFE merely echoes the existing IUCN Red List categorisation and is therefore redundant; (2) SAFE should not be proffered as a replacement for the Red List; (3) SAFE simplifies mathematically to a measure of a species' abundance and therefore provides no additional risk information; and (4) minimum viable population (MVP) size, on which SAFE is based, is species-specific and so a threshold abundance applied to all species cannot be used. Based on our response to our colleauges (Bradshaw et al. 2011), we outline below why each of these arguments is unsupported.

(1) SAFE merely echoes IUCN Red List categorisation. Contrary to the implicit assertion in the three critiques, most Red List criteria on which threat categorisations are founded are not related to a population's size per se. Rather, the three most-used criteria are based on a measured or perceived *reduction* in population size (criterion A) or geographic range (criterion B). Criteria C (indicating small population size and decline or fragmentation) and D (small size only) also set population-size thresholds for long- and short-term persistence (Critically Endangered: 250 and 50 individuals; Endangered: 2500 and 250 individuals; Vulnerable: 10 000 and 1000 individuals [although Vulnerable D2 is based only on restricted area of occupancy], respectively), yet these thresholds are arbitrary and not derived from any empirical risk assessment (these are "set at what are generally judged to be appropriate levels, even if no formal justification for these values exists" [International Union for the Conservation of Nature 2013]). The abundance thresholds for Critically Endangered and Endangered are typically one to two orders of magnitude lower than

nearly all quantitative MVP size estimates (Traill et al. 2010; Brook et al. 2011). Only criterion E is based on integrative modeling – population viability analysis (PVA) – which explicitly estimates extinction risk. Of the 95 mammal species we assessed for the SAFE index, 63 are IUCN threat-listed. Of these, 51% are not assessed by the IUCN on population size thresholds at all, and only one assessment is even partially based on PVA. Indeed, based on a recent (July 2011) examination of Critically Endangered, Endangered, or Vulnerable species, not one of 1370 mammal or 1288 bird species relies entirely on criterion E data, and only 4 mammal and no bird assessments include any PVA information. Hence, the assertion that the SAFE index (a measure of distance from MVP) simply reproduces the Red List is demonstrably incorrect. It is debatable to what extent the Red List categories predict real extinction risk (O'Grady et al. 2004); regardless, they must largely invoke reductions in geographic range and population size to do so.

(2) SAFE replaces the Red List. Under no circumstances did we assert that the SAFE index should replace the Red List, or that conservation based prioritisation should be based "solely on population size". We clearly called for SAFE to be used in conjunction with the Red List to provide a more heuristic measure of relative species-extinction threat. We agree that assessments made on population size (and their distance to MVP) alone are inadequate to explain all elements of risk – claiming otherwise would be astonishingly naïve (Brook et al. 2011). The contribution of SAFE to the existing Red List categories is that, in addition to reflecting susceptibility to stochastic extinction processes, it provides a continuous measure both *among* and *within* risk categories (somewhat analogous to RAMAS software's Red List fuzzy-number categorisation method [www.ramas.com/redlist.htm]). This is pertinent given the ambiguous nature of categorical terms like "endangered", "threatened", and "vulnerable" that are often confused by lay persons and used interchangeably or inconsistently in national level legislation.

In a triage context, the choice to invest in conserving particular species can be informed, at least partially, by MVP (Traill et al. 2010) and SAFE by indicating how urgently a species requires attention.

*(3) SAFE simplifies to population size (*N*)*. We incorporated a logarithmic transformation in SAFE to ease interpretability for our "distance from extinction and to MVP" concept across many species, and for standardisation purposes. For example, take hypothetical species A and B – comprising 200 and 2 000 000 individuals, respectively – and assume a threshold MVP target of 5000. Even for specialists, explaining the relative risk as "species A is 4800 individuals away from the threshold target", and "species B is 1 999 500 individuals above the threshold" becomes a confusing mix of largely irrelevant numbers and qualifiers. We maintain that it is far easier to infer whether species A is in trouble based on a negative SAFE index (–1.40, in this case), and that species B is at far less risk based on its positive SAFE value (2.60). As we originally stated in our paper, the threshold MVP value need not necessarily be 5000; if one has sufficient data to estimate, for instance, a taxon-specific MVP, then different denominator values could be used for different taxa (Traill et al. 2010; Brook et al. 2011). This process would act to normalise comparisons of SAFE-based extinction risks among groups (taxa or otherwise) with intrinsically different MVP sizes. Commonly used biodiversity evenness metrics such as Shannon's Index also use logarithms to make large and small sample sizes comparable.

(4) MVP size is not generalisable. Several authors took exception to our concept of a generalisable MVP size for use as a target threshold, based mainly on arguments raised in a recent critique (Flather et al. 2011). We have addressed these concerns elsewhere (Brook *et al*. 2011), but summarise our principal defense here. Although MVP does vary among species, the key emergent result is that thousands, and not hundreds, of individuals are needed to minimise

the risk of stochastic extinction − this is the essence of the MVP "rule of thumb" (Traill et al. 2010). PVAs are unavailable to estimate MVPs for most species, so generalisations are required in most instances. The alternative – to argue that the problem is too intractable and uncertain and that all species are unique − leads nowhere in terms of practical conservation management.

CONCLUSIONS

It is surprising that a heuristic concept designed to enhance conservation decision making has evoked such spirited criticisms from the progenitors of the Red List (Akçakaya et al. 2011) and other conservation decision-theory specialists (Beissinger et al. 2001; McCarthy et al. 2011). Putting aside arguments about uncertainty and relative merit, the real test of the SAFE concept's utility will be determined by whether it can contribute usefully to on-the-ground conservation decisions.

End of chapter 3

Chapter 4: Roads, wildlife and indigenous people in Peninsular Malaysia

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INTRODUCTION

The negative impacts of roads on biodiversity are relatively well-documented (see Chapter 2 and reviews by Laurance et al. 2009; Fahrig & Rytwinski 2009; Benítez-López et al. 2010). However, the impacts of roads on the livelihoods of local people have usually been regarded as positive. In fact, it has been proposed that, by acting as 'magnets' for colonists, properly planned roads in already-degraded areas can increase farming efficiency and improve local livelihoods while actually reducing net deforestation at a regional scale (Laurance & Balmford 2013).

Road expansion and improvement to benefit local people can, however, concomitantly drive biodiversity loss. Evidence of this can be found in the human-dominated landscapes of Southeast Asia, where biodiversity is imperiled by agricultural expansion, logging and overhunting (Sodhi et al. 2010a). In Indonesia, for example, the paving of the Blangkejeren-Kutacane Road in Sumatra created greater access for illegal colonisation, unauthorised logging, creation of roadside forest gardens, and hunting within Gunung Leuser National Park (Wind 1996; Singleton et al. 2004). In Vietnam, roads have also increased local demand for bushmeat in once-remote areas due to an influx of workers (Long & Hoang 2007). In Lao PDR, a recent evaluation showed that repair works to a highway, ostensibly to alleviate poverty, inadvertently increased the risk of cross-border trafficking of mammals (Asian Development Bank 2008). As such, there is a need to better understand how the presence of a road and its expansion impact the livelihoods of forest-dependent indigenous peoples and the biodiversity around them.

In Peninsular Malaysia, one group of indigenous people from whom such information can be gleaned is the Orang Asli, who make up roughly 0.5% of the national population (Nicholas 2004). The Orang Asli people consist of 18 subgroups that can be grouped into three main ethnic groups: Negrito, Senoi and Proto-Malay. They are considered to be descendants of the Peninsula's earliest inhabitants, with some groups dating back at least 25,000 years (Nicholas 2004). Around 89% of the Orang Asli live in rural areas and are heavily involved in activities related to agriculture and forest-resource harvesting (Nicholas 2004). As such, it is important to understand the degree to which the Orang Asli are dependent on roads for their livelihoods, and the factors responsible for their level of support for the road. For example, certain demographic factors (e.g. length of residency and education level) that are known to be determinants of indigenous attitudes towards conservation (Mehta & Heinen 2001) and ecosystem services (Sodhi et al. 2010b) may be important predictors of support for roads and their expansion. It is also plausible that the Orang Asli's perceived or actual reliance on roads for their livelihoods may influence their level of support.

In the interests of biodiversity conservation, it is also important to investigate the impacts of roads on the hunting practices of indigenous people. In Ecuadorian Amazon, for example, an oil road was reported to have transformed once semi-nomadic indigenous hunters into commercial poachers (Suárez et al. 2009). In East Africa, impoverished villagers in the Serengeti claimed that poor roads, which can limit development opportunities and alternative livelihood strategies, have forced them to hunt more animals for survival (Fyumagwa et al. 2013). In Peninsular Malaysia, the Orang Asli are known to hunt mammals for both subsistence and commercial purposes (Andaya 2008; Azrina et al. 2011), and individuals have been implicated (Yeng 2010) or caught (Jamaludin, 2008) in illegal hunting activities throughout Malaysia. In the State of Perak, for example, there has been evidence of high poaching pressure in forests beside the highway bisecting the Belum-Temengor Forest Complex (Fig. 14; Clements et al. 2010a). Recent studies have shown that these forests are still intensely used by the Malayan Tiger *Panthera tigris jacksoni* and Barking Deer *Muntiacus muntjak* (Darmaraj 2012), which are highly sought after by poachers for traditional Chinese medicine and game, respectively. However, the extent to which Orang Asli hunters are involved in poaching activities in forests along the highway in Belum-Temengor is unknown. Because resource harvesting practices are known to be affected by certain demographic factors, such as education level (Lee et al. 2009), it is also conceivable that such factors can influence their decision to hunt in forests near roads. It is also worthwhile examining whether and how such decisions are affected by the perceived state of mammals along the highway; for example, some may not prefer hunting in roadside forests because they believe animals are less abundant there.

Working within an important mammal habitat in Peninsular Malaysia that has been bisected by a highway, I interviewed Orang Asli communities to quantify their level of (1) support for the highway, (2) support for additional roads to their village, and (3) use of roadside forests for hunting. Next, I investigated how the demography, livelihood activities, and perceived impacts of the highway on Orang Asli livelihoods affected their responses. My findings have important implications for conservation practitioners working in important mammal conservation areas with indigenous communities, whose outlook on roads must be carefully considered when new roads are proposed or built.

METHODS

Study area

The Belum-Temengor Forest Complex (3,546 km^2) is considered an important mammal conservation area; for example, it contains more than half of the total mammal species documented in Peninsular Malaysia (Darmaraj 2012), and lies in the heart of one of the world's priority Tiger Conservation Landscapes (TCL no. 16; Dinerstein et al. 2006). In 1982, a 203 km-long paved road known as the East-West Highway (Federal Route 4) was built through this forest complex. The complex consists of three forest categories: a protected area (Royal Belum State Park), production forest reserves (Banding, Temengor and Gerik), and state land forests (Fig. 14). In the protected area, forest clearance is strictly prohibited. In production forest reserves, selective logging is mainly carried out, but they can also be cleared for rubber tree planations (Aziz et al. 2011). In state land forests, forests can be cleared by the state government for development. Orang Asli villages can be found in each forest category, but the Orang Asli are legally allowed to hunt for subsistence in each of them despite the land falling under different management categories.

The two main sub-ethnic groups of Orang Asli living in Belum-Temengor are the Jahai and Temiar, although another sub-ethnic group, the Semai, is present as a minority. Most of these villages were formed after the construction of the Temengor dam in 1979; flooding of the forests to form the lake necessitated the resettlement of semi nomadic Orang Asli living within Royal Belum State Park and Temengor Forest Reserve. While several Orang Asli groups live in villages without direct vehicular access to the highway (Fig. 14), most have been resettled by the government in villages within production forest reserves and state land forests with direct vehicular access to the highway via old logging roads.

Interviews

In Belum-Temengor, a total of 10 Orang Asli villages were visited (Fig. 14; Table 9). Interviews were administered by five people fluent in Bahasa Malaysia, the national language that is also spoken by the Orang Asli in addition to their own dialects.

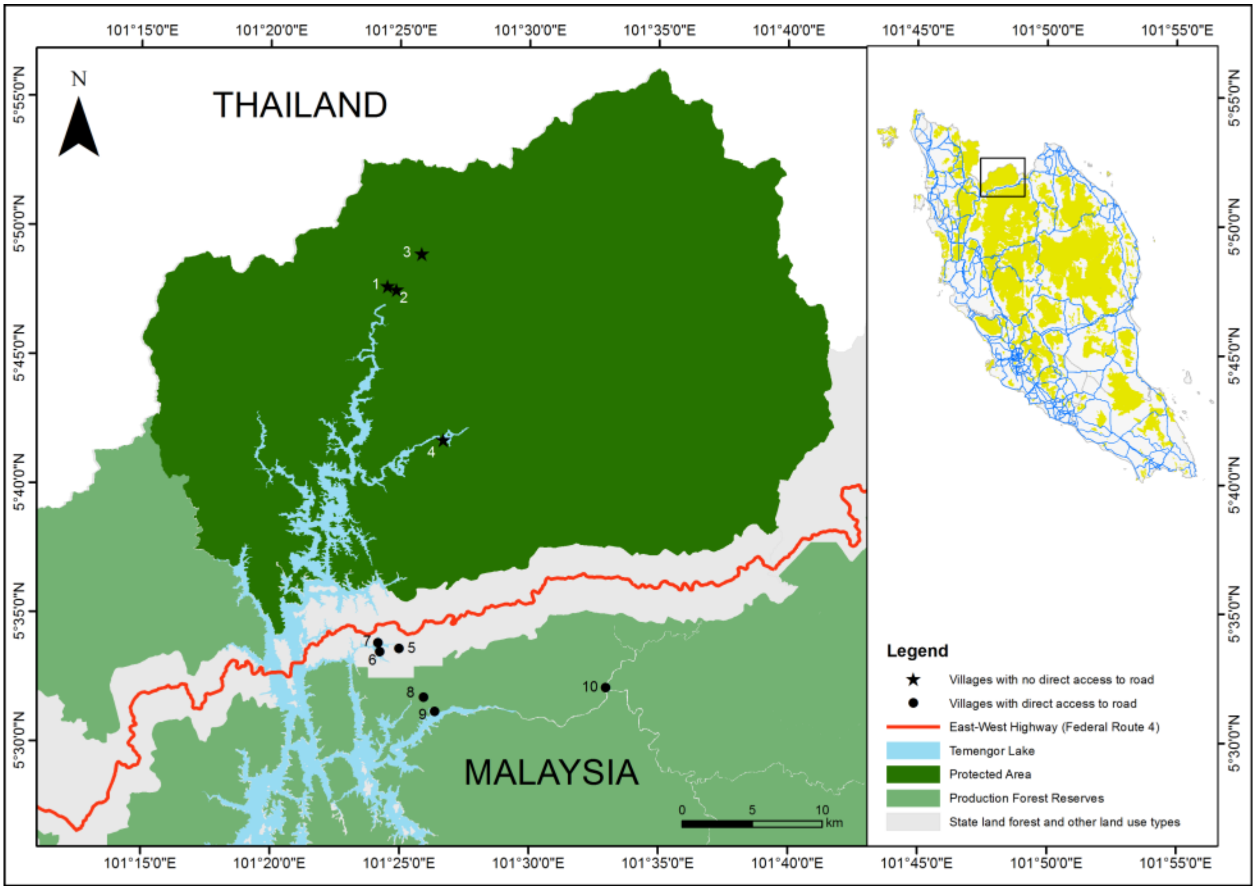


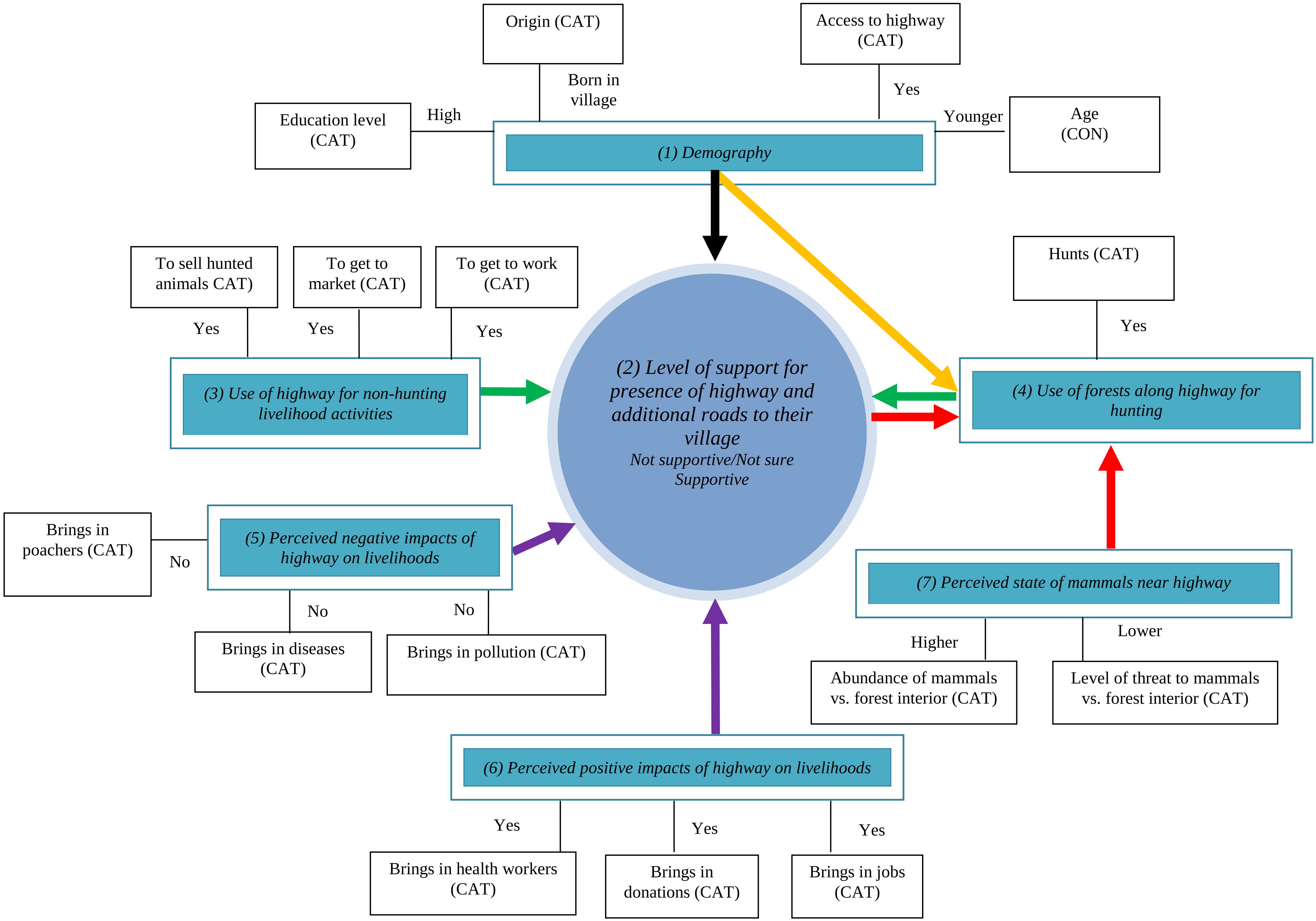
Fig. 14. *Map of the Belum-Temengor Forest Complex and location of 10 villages where Orang Asli were interviewed in the State of Perak, Peninsular Malaysia. Refer to Table 9 for names of villages corresponding to the numbers.*

Table 9: *Summary table of Orang Asli village names, their geographic coordinates, the number of households visited within the village, and the number of households interviewed.*

	Village Name	X-cord	Y-cord	No. of households visited	No. of households interviewed	% of households interviewed
1	Bongor Hilir	101.407	5.793	25	13	52
2	Bongor Kecil	101.413	5.791	20	13	65
3	Sungai Kejar	101.429	5.814	20	6	30
4	Sungai Tiang	101.443	5.694	85	59	69
5	Banun/Raba	101.415	5.559	30	27	90
6	Desa Damai	101.403	5.557	21	16	76
7	Pengkalan Permai	101.401	5.563	10	8	80
8	Semelor	101.431	5.528	27	10	37
9	Pulau Tujuh	101.438	5.519	14	10	71
10	Sungai Tekam	101.548	5.535	12	7	58
			Total	264	169	64

All households in each village were visited, and interviews were conducted with the household head; if the head was unavailable, he was substituted by another permanent household member. Only males were interviewed, as females are not usually involved in hunting activities. Before each interview, consent to participate was obtained prior to the commencement of the interview and following on from an explanation of the interview schedule. Ethics approval for conducting these interviews was obtained from the James Cook University Ethics Committee (Ethics Approval Application ID H3655). During each interview, a photograph of the highway with adjacent forests was shown to the respondents whenever reference was made to the highway and roads. The interview consisted of open-ended and fixed-response questions (Appendix 8) to obtain answers to seven information groups (Fig. 15): (1) demography (i.e. education, origin, age and having direct access to the highway (via old logging roads); (2) level of support for the highway and construction of additional roads to their village; (3) use of the highway for non-hunting livelihood activities (i.e. to get to work or market or to sell hunted mammals); (4) use of the highway for hunting in adjacent forests; (5) perceived negative impacts of the highway on livelihoods (i.e. brings in pollution, disease or poachers); (6) perceived positive impacts of the highway on livelihoods (i.e. brings in health workers, donations or jobs); and (7) perceived state of mammals near the highway (i.e. level of threat to mammal; abundance of mammal; score 1-lower, 2 – no difference, 3 - higher). The order of “yes” and “no” answers in various questions was alternated to prevent the natural tendency of some respondents to pick the first answer. Each interviewer scored the reliability of answers from each respondent (‘high’ – displayed an understanding of more than half the questions; ‘low’ – displayed a poor understanding of more than half the questions); those with a ‘low’ reliability score were excluded from the analyses.

Fig. 15*: Hypothetical relationships (arrows) among seven information groups that required responses from Orang Asli respondents: (1) demography (i.e. education, origin, age and having direct access to highway [via old logging roads]); (2) level of support for highway and construction of additional roads to their village; (3) use of highway for non-hunting livelihood activities (i.e. to get to work or market or to sell hunted animals); (4) use of highway for hunting in adjacent forests; (5) perceived negative impacts of highway on livelihoods (i.e., brings in pollution, disease or poachers); (6) perceived positive impacts of highway on livelihoods (i.e., brings in health workers, donations or jobs); and (7) perceived state of mammals near highway (i.e. level of threat to mammals and abundance of mammals in comparison to forest interior). A hypothetical response is provided next to each covariate. For example, we may find that when we examine demographic covariates, indigenous people who have: (1) a higher education level, (2) originated from the village, (3) direct access to the highway, or (4) are younger, may hypothetically support the presence of the highway and additional roads.*


Origin (CAT)
Born in village
Access to highway (CAT)
Yes
Education level (CAT)
High
Younger
Age (CON)
(1) Demography
To sell hunted animals CAT)
To get to market (CAT)
To get to work (CAT)
Yes
Yes
Yes
Hunts (CAT)
Yes
(3) Use of highway for non-hunting livelihood activities
(2) Level of support for presence of highway and additional roads to their village
Not supportive/Not sure
Supportive
(4) Use of forests along highway for hunting
Brings in poachers (CAT)
No
(5) Perceived negative impacts of highway on livelihoods
No
No
Brings in diseases (CAT)
Brings in pollution (CAT)
(7) Perceived state of mammals near highway
Higher
Lower
Abundance of mammals vs. forest interior (CAT)
Level of threat to mammals vs. forest interior (CAT)
(6) Perceived positive impacts of highway on livelihoods
Yes
Yes
Yes
Brings in health workers (CAT)
Brings in donations (CAT)
Brings in jobs (CAT)

Statistical analyses

First, we constructed binary logistic regression model sets to examine which of four demographic variables (black arrow; Fig. 15), four livelihood activities (dark green arrows; Fig. 15) and six perceptions (purple arrows; Fig. 15) were the most important predictors of support for the highway and construction of additional roads to the respondents' village.

Second, we constructed similar regression models to elucidate which of four demographic variables (orange arrow; Fig. 15) and two perceptions of the highway's impact on mammals (red arrows; Fig. 15) were the most important predictors of the respondents' decision to hunt in forests near the highway. All analyses were conducted in R statistical environment 3.0.0 (R Development Core Team 2010). Before constructing each model set, we ran collinearity tests among covariates. As our response covariates were categorical, we used the *hetcor* function implemented in the polycor library to compute a heterogeneous correlation matrix consisting of Pearson product-moment correlations between factors. We retained predictor covariates with coefficients $<|0.5|$ for construction of each model set. We believe it is important to account for possible non-independence of answers within each Orang Asli village and ethnic group. We constructed mixed-effects binary logistic regression models using the *lmer* function implemented in the lme4 library, with *Village* and *Ethnicity* coded as random effects. To avoid over-parameterising, we ran three global models that included all possible combinations of both random effects, and retained the random effect with the lowest Akaike's Information Criterion (AIC) value for construction of each model set. For each model set, we included all possible combinations of covariates without interactions plus a null model. Following the guidelines described by Bolker et al. (2009), we fitted generalised linear mixed-effect models (GLMMs), assigning a binomial distribution and logit link function to each candidate model, and using

Restricted Maximum Likelihood (REML) to estimate model parameters. After calculating AIC$_c$ values (AIC corrected for small sample size) and weights as a means to assess the relative distance of the models from the 'truth', we compared relative statistical evidence between models using the information-theoretic evidence ratio (*ER*) which is the AIC$_c$ weight of one model divided by another; this concept is akin to Bayesian odds ratios (McCarthy 2007) and is preferable to a classic null-hypothesis significance test because the likelihood of the alternative model is explicitly evaluated (Bradshaw & Brook 2010). Finally, we calculated the percentage deviance explained (%DE) as a measure of goodness-of-fit for each model, and compared %DEs to determine the proportion of variance in the response that was attributable to each predictor (e.g. Clements et al. 2011).

RESULTS

A total of 264 Orang Asli households were visited in nine villages, but only 169 male household heads agreed to be interviewed. After discarding interviews that were considered unreliable, data from 144 households could be used. The mean ± SD age of the respondents was 36 ± 13 years (range: 16-70), with around 55% originally born in their villages. More than half (54%) of the respondents had received some form of formal education.

Out of the 144 Orang Asli households, 84% supported the presence of the highway, while 65% supported the idea of constructing additional roads to their village. Use of the highway for livelihood activities among all respondents was low, with 28%, 24%, 19% and 2% of respondents, respectively, using the highway for work, to get to market, to hunt, and to sell hunted animals. Among Orang Asli with access to the highway, 57%, 47%, 21% and 4% of respondents, respectively, used the highway for work, to get to market, to hunt, and to sell hunted animals. Perceptions of the highway having a negative impact on their livelihoods were evenly divided, with 47%, 49%, and 49% of respondents, respectively, perceiving the highway to bring in pollution, disease, and poachers. However, more respondents perceived the highway to have a positive impact on their livelihoods, with 76%, 56% and 66% of respondents, respectively, perceiving the highway to bring in health workers, charitable donations and jobs. When we asked respondents whether they actually wanted these benefits, 92%, 94%, and 90% of them respectively indicated that they did. Finally, the majority of the respondents (63%) perceived that threats to wildlife along the highway were higher compared to wildlife in the forest interior, with respondents believing that sources of the threats (in order of decreasing importance) were: (1) logging, (2) roadkills, (3) infrastructure development, (4) non-indigenous locals, (5) foreigners, (6) Orang Asli from other villages and (7) Orang Asli from their own villages. Furthermore, the majority (79%) felt that the abundance of wildlife along the highway was lower compared to the forest interior.

Two model sets revealed that having direct access to the highway was the most important demographic predictor of support for the existing road and for additional roads to their village (Table 10). Based on the Evidence Ratio (ER), the top-ranked models in each model set containing this predictor had 1.6 and 2.3 times more support than the second-ranked model, respectively (Table 10). This predictor described ~5.5% and ~7.6% of the deviance in both model sets, respectively (Table 10), with beta coefficients indicating a positive relationship.

Among the four livelihood activities, use of the highway to get to market was the most important predictor of support for the road (Table 11). Based on the ER, the top-ranked model containing this predictor had 2 times more support than the second-ranked model (Table 11). This predictor described ~8.5% of the deviance (Table 11), with its beta coefficient indicating a positive relationship. However, none of the livelihood activities were important in predicting support for additional roads to their village.

Among the perceived positive and negative impacts of the highway on livelihoods, the perception of the highway bringing in both health workers and jobs was the most important predictor of support for the road (Table 12). Based on the ER, the top-ranked model containing this predictor had 1.7 times more support than the second-ranked model (Table 12). This predictor described ~18.0% of the deviance (Table 12), with its beta coefficient indicating a positive relationship. Again, none of the respondents' perceptions of the highway bringing in both health workers and jobs were important in predicting support for additional roads to their village.

Respondents who were willing to reveal species they hunted in forests along the highway (n = 9) identified squirrels, monkeys, Wild Pig (*Sus scrofa*), Sambar (*Rusa unicolor*), Barking Deer (*Muntiacus muntjak*) and Sunda Pangolin (*Manis javanica*). However, there was no important demographic predictor for use of the highway to hunt in adjacent forests; instead, the most important predictor was the perceived threat to mammals in forests near the highway (Table 13). Based on the ER, the top-ranked model containing this predictor had 1.5 times more support than did the second-ranked model (Table 13); this predictor described ~ 4.7% of the deviance (Table 13), with its beta coefficient indicating a negative relationship.

Table 10*: Top-ranked generalised linear mixed-effect models (GLMM) examining the relationship between Orang Asli support for (1) the presence of the highway (SUP) and (2) additional roads to be built to their village (MOR) amd demographic predictors (ACC – having direct access to the highway; AGE – age of respondent; EDU – having received education; ORG – originating from the village). Village (VIL) was coded as a random effect. Only models with < 2 $dAIC_c$ are shown.*

Model	***k***	**LL**	**AIC_c**	**$dAIC_c$**	**$wAIC_c$**	**%DE**
SUP ~ ACC+(1|*VIL*)	4	-52.60	113.48	0.00	0.28	5.50
SUP ~ ACC+AGE+(1|*VIL*)	5	-52.03	114.49	1.01	0.17	6.52
SUP ~ ACC+EDU+(1|*VIL*)	5	-52.33	115.10	1.62	0.13	5.97
MOR ~ ACC+(1|VIL)	4	-66.53	141.36	0.00	0.35	7.58
MOR ~ ACC+AGE+(1|VIL)	5	-66.28	143.00	1.65	0.15	7.93
MOR ~ ACC+ORG+(1|VIL)	5	-66.32	143.08	1.73	0.15	7.87

Term abbreviations are defined as follows: k = number of parameters, LL = maximum log-likelihood, $dAIC_c$ = difference in AIC_c for each model from the most parsimonious model, $wAIC_c$ = AIC_c weight, and %DE = percent deviance explained in the response variable by the model under consideration.

Table 11: *Top-ranked generalised linear mixed-effect model (GLMM) examining the relationship between support for the presence of the highway (SUP) and use of the highway for livelihood activities (HUN – to hunt along forests by the highway; MAR – to get to market; SHA – to sell hunted animals) among Orang Asli respondents. Village (VIL) was coded as a random effect. Due to the effects of collinearity with (MAR), one activity (use of highway to get to work; WOR) was excluded in the final model set, especially when it had a relatively higher AICc value than did MAR when compared among a single-predictor model set. Only models < 2 dAIC$_c$ are shown.*

Model	***k***	**LL**	**AIC$_c$**	**dAIC$_c$**	***w*AIC$_c$**	**%DE**
SUP ~ MAR+(1|VIL)	4	-50.93	110.14	0.00	0.46	8.49
SUP ~ HUN+MAR+(1|VIL)	5	-50.53	111.50	1.36	0.23	9.20
SUP ~ SHA+MAR+(1|VIL)	5	-50.70	111.84	1.70	0.19	8.90

Term abbreviations are defined as follows: k = number of parameters, LL = maximum log-likelihood, dAIC$_c$ = difference in AIC$_c$ for each model from the most parsimonious model, wAIC$_c$ = AIC$_c$ weight, and %DE = percent deviance explained in the response variable by the model under consideration.

Table 12: *Top-ranked generalised linear mixed-effect model (GLMM) examining the relationship between support for the highway (SUP) and perceived impacts of the highway on livelihood activities (HEA – brings in health workers; JOBS – brings in jobs; DIS – brings in disease; POA – brings in poachers) among Orang Asli respondents. Village (VIL) was coded as a random effect. Due to the effects of collinearity with (DIS), one perception (highway brings in pollution; POL) was excluded in the final model set, especially when it had a relatively higher AICc value than did DIS when compared among a single-predictor model set. Only models < 2 dAIC$_c$ are shown.*

Model	***k***	**LL**	**AIC$_c$**	**dAIC$_c$**	***w*AIC$_c$**	**%DE**
SUP ~ HEA+JOB+(1|VIL)	5	-33.01	76.56	0.00	0.28	17.96
SUP ~ DIS+HEA+JOB+(1|VIL)	6	-32.50	77.77	1.21	0.15	19.22
SUP ~ POA+HEA+JOB+(1|VIL)	6	-32.83	78.42	1.86	0.11	18.41

Term abbreviations are defined as follows: k = number of parameters, LL = maximum log-likelihood, dAIC$_c$ = difference in AIC$_c$ for each model from the most parsimonious model, wAIC$_c$ = AIC$_c$ weight, and %DE = percent deviance explained in the response variable by the model under consideration.

Table 13*: Top-ranked generalised linear mixed-effect model (GLMM) examining the relationship between the use of the highway for hunting in adjacent forests (HUN) and different predictor covariates among Orang Asli respondents. Predictor covariates include as support for the presence of the highway (SUP), perceived threat to mammals along the highway (THR) and perceived abundance of mammals along the highway (ABC), with village (VIL) coded as a random effect.. Only models < 2 $dAIC_c$ are shown.*

Model	***k***	**LL**	**AIC_c**	**$dAIC_c$**	**$wAIC_c$**	**%DE**
HUN ~ THR+(1|VIL)	4	-61.15	130.63	0.00	0.32	4.67
HUN ~ ABD+THR+(1|VIL)	5	-60.44	131.38	0.74	0.22	5.78
HUN ~ SUP+THR+(1|VIL)	5	-60.82	132.13	1.50	0.15	5.19

Term abbreviations are defined as follows: k = number of parameters, LL = maximum log-likelihood, $dAIC_c$ = difference in AIC_c for each model from the most parsimonious model, $wAIC_c$ = AIC_c weight, and %DE = percent deviance explained in the response variable by the model under consideration.

DISCUSSION

To our knowledge, this is the first study in a key biodiversity area of Peninsular Malaysia that has investigated the outlook of indigenous peoples on roads and the potential effects of roads on their livelihoods. Our main finding is that in such areas where roads need to be opposed on environmental grounds, it may be difficult to garner support from indigenous people who already have access to a previously constructed road from which they derive socio-economic benefits.

Most Orang Asli who support the presence of the road and construction of additional roads already have direct access to the road. One of the most frequently cited reasons for its support is “ease of travelling”. In fact, our regression models suggest that once an Orang Asli individual has direct access to a road, he is likely to support it and the construction of additional ones to his village. This has important implications for conservation planning. For instance, it might be increasingly difficult to obtain local support against new road projects that threaten wildlife habitats, especially from people who already have direct access to a previously constructed road. As an example, in East Africa, interviews with villagers in the Serengeti National Park who already have access to a poor road revealed that most of them supported its improvement despite protests from environmentalists (Fyumagwa et al. 2013).

For people without direct access to the highway, one possible reason behind their lack of support for the highway and additional roads is the existence of alternative modes of transport such as boats, or walking along forests trials by foot. Among this same group of respondents, several individuals were aware of the threat posed by road construction to forests. For example, comments made against roads included “wanting to see the forest remain in its pristine state”, “roads destroy nature”, and “additional roads will take up too much land such that (one day) there may be nothing left to eat”. Laurance et al. (2001) and Kirby et al. (2006) highlight similar attitudes among indigenous groups living in the Brazilian Amazon, whereby those in isolated communities tend to retain traditional values and belief systems whereas those closer to roads can be 'corrupted' by cash offers from illegal loggers and miners. Therefore, conservation practitioners who want to limit new roads may not always encounter opposition from indigenous people living in roadless wildlife habitats, especially in places like Royal Belum State Park where alternative means of transport are available.

Despite a relatively high level of support for the highway and additional roads, our results show that actual use of the highway by respondents for livelihood activities appears relatively low, even after considering only those with direct access to roads. Indeed, the Orang Asli appear to remain heavily dependent on forest resources around their villages for agriculture and forest resource harvesting (Nicholas 2004), for subsistence and/or commercial trade. From our surveys, out of 164 respondents who provided information on their occupation, 68% were engaged in part or full-time natural resource harvesting in surrounding forests, such as harvesting agarwood and fish, whereas only 1% appeared to have jobs that require regular use of the highway, such as nature guides in nearby resorts and assisting conservation NGOs.

Nevertheless, our results show that the Orang Asli believe that the highway has a net positive impact on their livelihoods. Access to markets in towns, and perceived benefits brought by the highway such as jobs and health workers, appear to be important factors in determining positive support for the highway (Table 12). Their desire for access to healthcare is unsurprising because many Orang Asli still suffer from diseases associated with under-development (Chee 1996), despite the availability of sufficient information for government health workers to reduce preventable illnesses (Baer 1999). Given that a large proportion of the Orang Asli still live below the poverty line, with 50% belonging to the very poor cf. to 2.5% nationally (Nicholas

2004), it is also not surprising that they desire greater access to jobs. Therefore, if conservation practitioners want to limit new roads threatening key biodiversity areas, they must find ways to improve the social welfare of resident indigenous communities, especially in places where the desire for access to markets, healthcare and jobs is great. If basic socio-economic needs are not met, there is a danger that forest communities may increasingly hunt animals for markets if a road and transportation is already available (e.g. in the Congo Basin; Wilkie et al. 2000). Indeed, in East Africa, environmentalists have recently come under criticism for putatively overstating the threat of a planned road to wildlife in the Serengeti and for ignoring the needs of impoverished local communities (Homewood et al. 2010; Fyumagwa et al. 2013).

One of the objectives of our study was to examine whether roads influence hunting patterns of the Orang Ali in forests along the highway in our study area. Our interviews showed that the majority of respondents preferred to hunt in forests just outside their village, particularly for self-consumption. Based on their list of targeted species, and assuming this was a complete and comprehensive list, the Orang Asli are probably not contributing to the poaching of large charismatic mammals such as tigers and elephants, despite such poaching having been documented in forests along the same highway (Looi 2009). Indeed, hunting evidence collected by researchers and NGO-led anti-poaching patrols strongly suggests that the poaching problem along the highway may be largely attributed to well-organised commercial syndicates comprising foreigners from Indochina, or non-indigenous locals from nearby towns (GR Clements, personal observation). Indeed, our regression model set revealed that the decision by respondents not to hunt in the forests along the highway is mainly influenced by the belief that animals in roadside forests are more threatened (Table 13), citing non-indigenous locals and foreigners as a higher poaching threat than themselves. Therefore, enforcement efforts along the highway must be increased to ensure that species that can be legally hunted by the Orang Asli do not become extirpated by foreign poachers.

Overall, the proportion of variance explained by predictors in our regression models appears to be within an acceptable range (see meta-analysis by Møller and Jennions 2002). Nevertheless, we acknowledge that we may have yet to measure better predictors of the level of support for the highway and their decisions to hunt beside it. In order for our findings to be more representative, interviews should be carried out at other Orang Asli villages in the central and southern parts of Peninsular Malaysia. As the configuration of villages in our study area correlated the covariate of "having direct access to highway" with "distance from highway" and "forest category", future survey designs should also aim to select areas that would allow such confounding effects to be disentangled.

CONCLUSIONS

To our knowledge, governments in Southeast Asia generally do not solicit the opinions of indigenous people regarding road construction. In other tropical regions such as parts of New Guinea (Laurance et al. 2010), indigenous people have even lost their legal rights to impede development projects. In Sarawak, Malaysian Borneo, indigenous people have resorted to blockades against unwanted road development projects for fear of land grabs (Lawrence 2006). In Peninsular Malaysia, there have also been conflicts due to land tenure rights of the Orang Asli not being acknowledged nor granted legal standing (Nicholas 2004).

Ultimately, conservation practitioners interested in protecting key biodiversity areas in Peninsular Malaysia must solicit and include the participation of Orang Asli residents during enagement with state governments (e.g. Gill et al. 2009; Hood & Bettinger 2008; Nicholas 2005;

Aziz et al 2010) - this includes seeking their opinions on how planned road projects would affect their livelihoods and hunting practices. There are signs that indigenous people such as the Orang Asli are becoming more vocal in calling for a greater say in development activities affecting their customary lands (Kuek 2012; The Star 2011), with some measure of success (The Star 2010; Bernama 2012). If development projects such as roads are unavoidable, a compromise must ideally be reached so that the road does not jeopardise the surrounding biodiversity, but still manages to bring important socio-economic benefits to indigenous communities.

End of chapter 4

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Chapter 5: Can road underpasses 'bridge the gap' for large mammals in fragmented habitat linkages within Peninsular Malaysia?

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INTRODUCTION

Crossing structures such as underpasses may prove useful for improving habitat connectivity for mammals, but there are also potential costs and downsides associated with such structures. For example, underpasses may increase vulnerability of animals to predation and poaching (Ford & Clevenger 2010). Also, crossing structures can be expensive; building a structure to maintain a linkage can sometimes costs much more than constructing the road that would sever it (e.g. Simberloff & Cox 1992). Given that Malaysia was ranked the 7th most underfunded country in the world for biodiversity conservation (Waldron et al. 2013), conservation planners need to know whether underpasses are economically worthwhile investments.

At two of Peninsular Malaysia's linkages where 20 underpasses have been constructed (Figs. 16 and 17), we first determine whether these linkages are still of high conservation importance for mammals amidst fragmentation threats. Second, we investigate how fragmentation has affected forest use of a focal group of native large mammals around these underpasses. Third, we evaluate whether all 20 underpasses have been effective as crossing structures for these focal species. Fourth, we assess the efficiency at which individual underpasses have been used by these focal species in order to identify underpasses that warrant management interventions. Finally, we investigate whether the presence of underpasses at the end of forest trails can potentially mitigate the impacts of the road on these focal species.

METHODS

Study area

The boundaries of our forest blocks in each linkage were demarcated to minimise confounding effects from other land uses. Because mammal population densities are known to decline up to 5 km from infrastructure such as roads (Benítez-López et al. 2010), we set the northern and southern ends of each forest block to be at a distance of ~ 4-5 km from either side of the road – this also increases the probability of obtaining a gradient of forest use by mammals.

At linkage 7 (Fig. 1) in eastern Peninsular Malaysia (hereafter known as the 'eastern linkage'), our survey area comprised two forest blocks (Fig. 16). Both forest blocks span portions of four forest reserves (Tembat, Petuang, Hulu Telemong and Hulu Nerus). We excluded the stretch of road between both forest blocks from our survey because we wished to avoid confounding effects from active clear-felling of forests for new hydroelectric dams (Fig. 16). Both forest blocks skirt the largest artificial reservoir in Southeast Asia – Lake Kenyir, which was completed in 1985. At linkage 8 (Fig. 1) in western Peninsular Malaysia (hereafter known as the 'western linkage'), our survey area comprised a single forest block (Fig. 17). This forest block spans four reserves (Gunung Inas, Belukar Semang, Bintang Hijau and Kenderong).

All forest reserves contain lowland-hill dipterocarp forests that were first selectively logged in the 1970s. Within all forest blocks in both linkages, we believe that confounding effects from sustained resource harvesting on forest use of large mammals were minimal because during our survey period, there was no logging within the forest reserves and no permanent human settlements in the area. Nevertheless, there were signs of encroachment and we sought to examine their effects on forest use of large mammals.

To estimate traffic intensity at each linkage, we monitored vehicles (≥ 4 wheels) at a stationary point on the road during peak hours (0830-1030 hrs and 1400-1630hrs) on work days during our camera trapping period in the eastern (23/2/11-14/6/11) and western linkage (20/11/12-5/3/13). Traffic intensity in the eastern linkage appeared to be relatively lower, with vehicle encounter rates of 0.7 vehicles/min (1249 survey minutes) and 4.3 vehicles/min (1260 min), respectively.

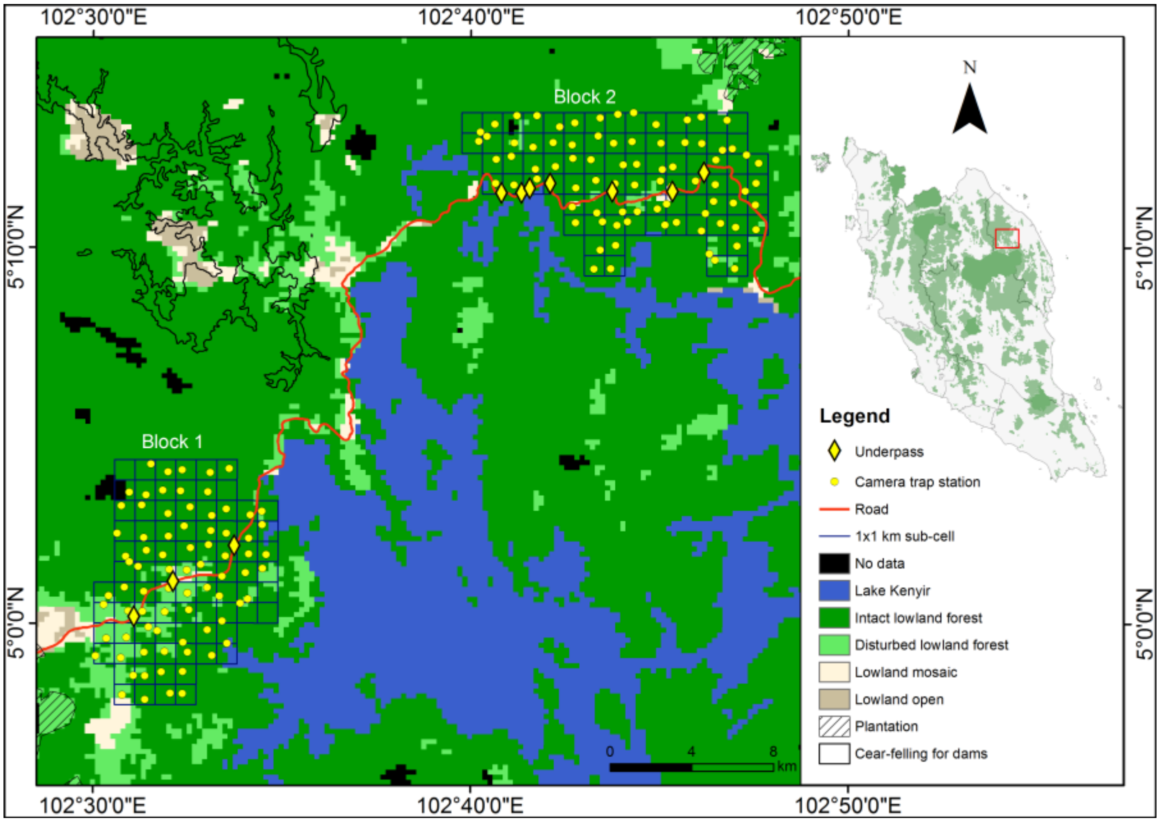

Fig. 16. *Map illustrating 42 cells (2x2 km) stratified into 168 sub-cells (1x1 km) in two forest blocks encompassing linkage 7 (eastern linkage), as well as 10 underpasses in the State of Terengganu, Peninsular Malaysia. Block 1 contains three underpasses, while Block 2 contains seven underpasses. Land cover layer is derived from MODIS 250-m resolution satellite images (Miettinen et al. 2012). Sub-cells without camera trap stations indicate data loss due to camera trap theft, malfunctions, damage from elephants, or blockage from vegetation.*

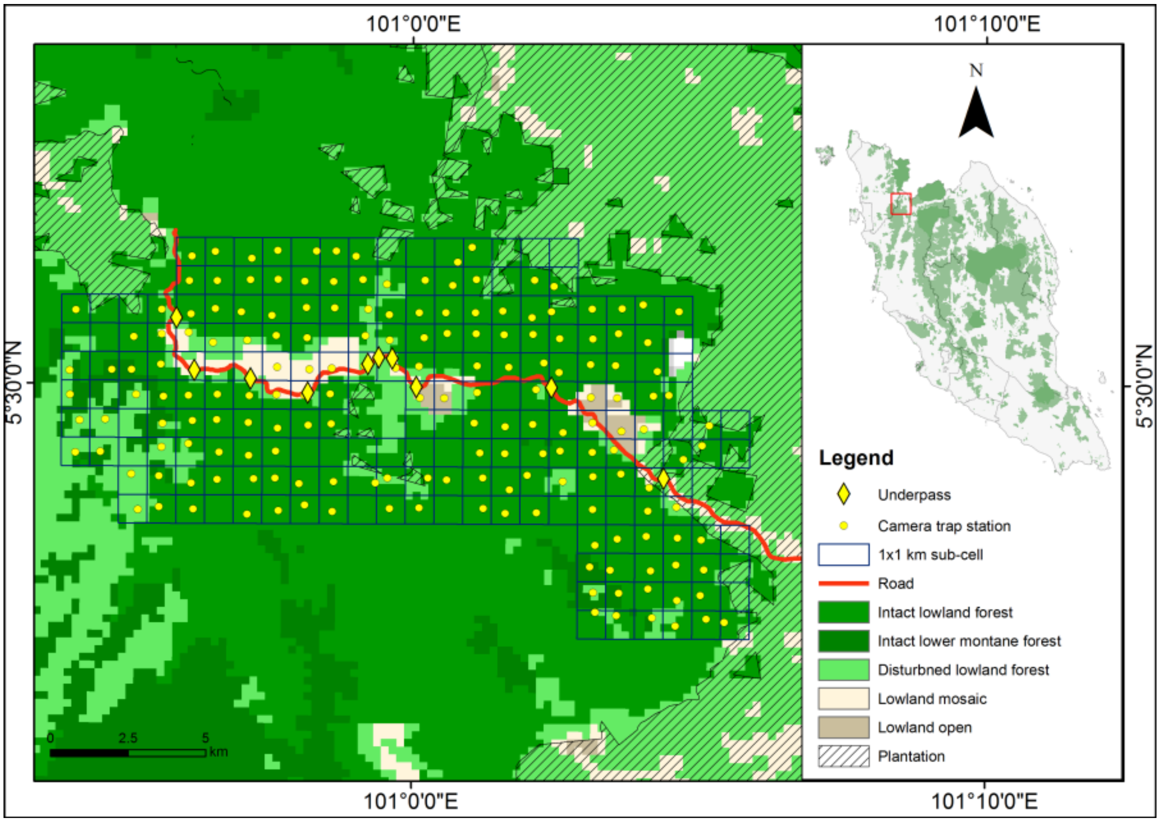

Fig. 17*. Map illustrating 56 cells (2x2 km) stratified into 224 sub-cells (1x1 km) within a single forest block encompassing linkage 8 (western linkage), as well as 10 underpasses in the State of Perak, Peninsular Malaysia. Land cover layer is derived from MODIS 250-m resolution satellite images (Miettinen et al. 2012). Sub-cells without camera trap stations indicate data loss due to camera trap theft, malfunctions, damage from elephants, or blockage from vegetation.*

Data collection

Ideally, we should be tracking animal movement beneath underpasses to monitor through-passages of underpasses by wild mammals. However, logistical constraints prevented us from using satellite collars, track-pads (e.g. Clevenger & Waltho 2000; Mata et al. 2008) or video surveillance systems (e.g. Kleist et al. 2007) to monitor through-passages of underpasses by wild mammals. Neverthelless, there appears to be a relationship between animal movement and habitat use, with animals preferring to travel through more suitable habitats (Chetkiewicz et al. 2006). Therefore, we make an important assumption in our study that if an animal is using an underpass, it is likely that the underpass is facilitating its crossing across the road.

In order to quantify forest and underpass use, we chose infrared camera traps (Model HC500; RECONYX Inc.,Wisconsin) to obtain detection/non-detection data of medium-large mammals. Economic analyses revealed that for longer study durations (> 1yr), it is more cost-effective to use camera traps to monitor wildlife crossing structures than methods such as track-pads (Ford et al. 2009). Furthermore, camera traps have been shown to be effective tools to quantify habitat use of cryptic mammals (Linkie et al. 2013). In order to obtain quality photos of our target species, we found that camera traps should ideally be ~50 cm above ground level at perpendicular distances of 2-5 m from the middle of the trail. Vegetation that could possibly cause false triggers was removed from the front of each camera trap. Each camera trap was calibrated to have a 1 s time-lapse and was set to RapidFire™ mode, which allowed photographs to be taken up to two frames per second. Camera-trap photos were catalogued using software Camera Base (version 1.5.1; http://www.atrium-biodiversity.org/tools/camerabase).

Forest camera trapping surveys

In the forests surrounding 10 underpasses at each linkage, we deployed camera traps to quantify forest use by native large mammals, which are defined as species with a body weight exceeding 20 kg (Morrison et al. 2007). Because a wild Sumatran Rhinoceros *Dicerorhinus sumatrensis* has not been sighted for ~20 years in the peninsula (Ahmad Zafir et al. 2010), we expect 12 native large mammal species to occur in both linkages.

Camera trapping in the eastern linkage was conducted between April 2011 and March 2012, across dry (April-November) and monsoon seasons (December-March). Camera trapping in the western linkage was conducted in between May 2012 and Feb 2013, across dry (May-November) and monsoon seasons (December-February). At the eastern linkage, each forest blocks was stratified into 21 cells (2 x 2 km), whereas the forest block in the western linkage was stratified into 56 cells of equivalent sizes. Within each 2 x 2 km cell, a camera trap was first deployed in the upper-left sub-cell (1 x 1 km). After one sampling period (~ 60 trap-days), that camera was rotated to the upper-right sub-cell. This systematic rotation occurred two more times, to the bottom-left and bottom-right sub-cells, until the entire cell was surveyed in a ‘Z’ shape manner after four sampling occasions; this meant that both linkages had up to 42 and 56 operational camera traps, respectively, during each of four sampling periods (two in dry and two in monsoon seasons). Within each sub-cell, deployment of camera traps followed two criteria: 1) placement in habitat where detection probabilities for large mammals are known to be high, such as animal trails, ridges or old logging roads; and 2) placement as close as possible to the centre of the sub-cell to minimise clumping, but shifted to next best location within the sub-cell if unsuitable terrain, such as steep cliffs or water bodies, was encountered.

Underpass camera trapping surveys

The carriageway width of each underpass spanned two lanes, while the lengths of underpasses ranged from 59-261 m in eastern linkage and 72-252 m in the western linkage. Beneath the 10 underpasses at each linkage, we deployed camera traps to quantify underpass use by mammals. We first identified dry land between underpass columns (Appendix 9) that could potentially be used by mammals. Trees were absent beneath underpasses, and any vegetation would typically consist of grass, shrubs or a combination of both. Substrates range from mud to sand with small rocks. Due to the absence of trees, camera traps were attached to metal stakes that were cemented to the ground to prevent theft. Camera traps were situated as close as possible to underpass columns to prevent mammals from passing behind the cameras. At column spaces that were too wide (≥ 10 m), two cameras were placed facing each other or away from each other to cover detection spaces of ~ 5 m. Where necessary, the vegetation in front of each camera trap was cleared to create a detection space, which was maintained every two months throughout the deployment of cameras beneath underpasses. Each underpass was constructed over 1-2 water features (e.g. streams or small rivers; Appendix 9), but we did not attempt to detect animals passing through these water bodies because possible flooding might damage the cameras. Memory cards in each camera were replaced every two months. Every underpass had no evidence permanent human habitation, but there were signs of temporary habitation in the form recreation camps for fishing. At the eastern linkage, a total of 43 camera traps were deployed at column spaces of 10 underpasses between May 2011 and March 2012. At the western linkage, a total of 66 camera traps were deployed at column spaces of 10 underpasses between June 2012 and January 2013.

Forest trail camera-trapping surveys

Due to the presence of open grass verges on either side of the highway that would result in extremely low detection probability of mammals, we could not deploy cameras along the highway to detect mammals at sections with and without an underpasses In order to maximize detection probability, cameras had to be deployed on trails leading to the highway or underpass instead. In the eastern linkage, forests trails either terminate at the road or an underpass. Between April 2012 and March 2013, camera traps were deployed on trails (mean ± SD distance from road = 70 ± 32 m) that terminated at underpasses (n = 10) and at the road (n = 30).

Statistical analyses

Fragmentation effects on large mammal forest use around underpasses

We used a likelihood-based occupancy framework (Mackenzie et al. 2002; Mackenzie et al. 2005) to estimate forest use (ψ_f) by large mammal species that had sufficient detections within each linkage (see Results). Species detection histories (see Chapter 1; eq. 1) were stratified into 23 temporal sampling occasions spanning 15 trap nights to facilitate calculation of detection probabilities (p). At both linkages, we examined the effect of three site covariates related to fragmentation on forest use of large mammals: (1) distance to road, (2) distance to nearest plantation, and (3) forest cover type as a proxy of logging intensity (a binary variable contrasting relatively intact lowland forest vs. regrowth, open, or mosaic lowland forest, based on a 2010 land-cover layer derived from MODIS 250-m resolution satellite images; Miettinen et al. 2012). For the eastern linkage, the effect of one additional fragmentation covariate, distance to reservoir edge, was investigated. To explicitly account for variation in detection probability, two sampling covariates were modelled for both linkages: (1) number of trap nights that cameras were operational during each sampling occasion, and (2) daily rainfall (recorded from weather stations nearest to each linkage, installed by the Department of Meteorology). It was important to account for the former sampling covariate because cameras deployed for longer durations have a higher probability of detecting a species, whereas the latter covariate may have an effect on the probability of detecting species affected by dry and monsoon seasons. For the eastern linkage, we included 'block type' as a sampling covariate because both forest blocks were isolated from each another and may have characteristics that could have inherently affected detection probability of certain mammal species. For example, Block 1 contains a known release site for translocated elephants and this would result in a higher detection probability for this species relative to Block 2. Furthermore, different blocks could have uneven rates of landscape change that could bias comparisons of species occurrence data (Betts et al. 2007).

Using the single-species, single-season occupancy framework in PRESENCE v5.3 software (Hines, 2006), we elucidated the combination of covariates that best explained estimates of forest use ($\hat{\psi}_f$) and detection probability (p) in each linkage. Prior to the modelling, we used the *hetcor* function implemented in the polycor library in R package 3.0.0 (R Development Core Team 2013) to compute a heterogeneous correlation matrix consisting of Pearson product-moment correlations between continuous and categorical covariates. We retained covariates with coefficients $<|0.5|$ for model construction. All continuous covariates were normalised to z-scores prior to modelling because if the range of raw data from covariates is over several orders of magnitude (e.g. distance covariates), the numerical optimisation algorithm in programme PRESENCE may fail to find the correct parameter estimates (Donovan & Hines 2007). Instead of performing multi-model inference on habitat use and detection probabilities simultaneously, a two-step approach was used (Darmaraj 2012). First, detection probability (p) was modelled where the parameter was either assumed constant or allowed to vary with individual or additively combined sampling covariates, with all site covariates included in each model (MacKenzie 2006). Second, the influence of covariates on forest use of a species was modelled where the parameter was either assumed constant or allowed to vary with individual or additively combined covariates, while maintaining the top ranked model for detection probability as derived from the first step.

Models were ranked using Akaike's Information Criterion corrected for small sample sizes (AICc; Burnham & Anderson 2002). Model fit was evaluated by comparing the observed Pearson chi-square statistic from the global model with chi-square statistics from 999 simulated parametric bootstrap datasets (MacKenzie & Bailey 2004). If overdispersion was present in the constant model, we used its overdispersion factor ($\hat{c}$) to inflate the corresponding standard errors by a factor ($\sqrt{\hat{c}}$) and using a quasi-likelihood over-dispersion parameter (QAICc) for model selection (Burnham & Anderson 2002). Covariates that were likely to be important predictors of forest use of mammals would be present in models with the highest AIC weight. Finally, for each model considered, we calculated the percentage deviance explained (%DE) as a measure of goodness-of-fit, and compared each model's %DE to that of the next most parsimonious model to examine what proportion of the variance in the response was attributable to individual covariates.

To obtain model-averaged mean estimates for forest use ($\hat{\psi}_{\bar{f}}$) for each species within each linkage, estimates across all models with AIC weights < 0.90 were averaged using this formula (Burnham & Anderson, 2002):

$$\hat{\psi}_{\bar{f}} = \sum_{i=1}^{m} w_i \, \hat{\psi}_i \qquad \text{(eqn. 9)}$$

where w_i is the AIC weight for model *i*, *m* is the number of candidate models, and $\hat{\psi}_i$ is the estimate of forest use or detection probability for model *i*. Comparing overlaps of 95% CIs also allowed us to ascertain evidence of differences in mean estimates for forest use ($\hat{\psi}_{\bar{f}}$).

Effectiveness of underpasses as crossing structures for large mammals

We examined whether all 20 underpasses can generally serve as *effective* crossing structures for focal large mammal species spanning carnivorous, herbivorous and omnivours guilds. For each species, we calculated the mean difference in the number of detections (per camera trap) between forest sites and underpasses in each 15-day sampling occasion (n = 17 in the eastern linkage, and n = 15 in the western linkage). A positive mean difference in the number of detections signifies that a species is more likely to be detected in the forest versus at the underpasses. We computed the 95% non-parametric percentile bootstrap confidence interval (CI) of this statistic by sampling with replacement across sampling occasions 9999 times (Davison & Hinkley 1997). If underpasses were potentially effective crossing structures for a species, the number of detections in the forest would be similar to that at an underpass (i.e the 95% CI overlaps with zero). Conversely, the potential of underpasses as effective crossing structures for a species would be low if the number of detections are significantly lower (i.e. 95% CI does not overlap with zero). This approach accounts for variability in spatial and temporal sampling effort at underpasses and in the forests surrounding them.

Efficiency of underpass use by large mammals

It is important to assess the efficiency of underpass use by large mammals in order to ascertain whether management interventions are necessary. As such, we evaluated the efficiency at which underpasses are used by a subset of the 12 large mammal species for which forest-use models have been constructed. To facilitate comparisons, we created an underpass use index (UUI), which explicitly considers forest use of species within the vicinity of an underpass, and accounts for imperfect detection:

$$UUI = \psi_u / \hat{\psi}_u \qquad \text{(eqn.10)}$$

where ψ_u is the observed underpass use, calculated by obtaining the proportion of total sampling occasions that a particular species was detected from pooled detection histories of all cameras beneath an underpass; and $\hat{\psi}_u$ is the expected underpass use estimate, calculated by averaging estimates of the probability of forest use ($\hat{\psi}_f$) by a particular species at two camera trap locations nearest to the underpass on opposite sides. If $\psi_u < \hat{\psi}_u$, an underpass is said to be used sub-optimally and may require management interventions to improve its use, whereas if $\psi_u = \hat{\psi}_u$, an underpass is said to be optimally used by species around it. If $\psi_u > \hat{\psi}_u$, an underpass is exceeding its efficiency of use and may warrant management intervention to prevent it from functioning as a sink. We assessed the efficiency of underpass use by (1) each species (η_s) and (2) across all species (η_a), based on ratios of $\sum$ UUIs to the maximum possible UUIs.

Finaly, we investigated whether there was a link between fragmentation and underpass use efficiencies of each species. First, we identified the most pervasive fragmentation covariate in forest-use models. Next, we correlated the odds ratios (i.e. exp[beta]) of forest-use models (*n=12*) that contained this covariate with the underpass use efficiencies *(η_s)* of each of the six focal species in each linkage.

Ability of underpasses to mitigate road impacts on large mammals

The eastern linkage was a suitable site to investigate whether underpasses can mitigate the impact of the road, which was clearly contributing to the mortality of mammals (Appendix 14). Therefore, we designed an experiment to investigate whether trail use by large mammals would increase when the trail ended at an underpass rather than at the road.

Using a likelihood-based occupancy framework, we estimated trail use for five focal mammal species with sufficient detections (Table 14) - Barking Deer, Asian Elephant, Asian Tapir, Sun Bear and Wild Pig, from two ecological guilds: herbivores and omnivores. Similar to the detection histories of cameras deployed in the forests, we used temporal sampling occasions spanning 15 trap nights, yielding a total of eight sampling occasions. To explicitly account for variation in detection probability, we included 'trail width' as a sampling covariate because it may be more difficult for cameras to detect species at the far end of wider trails. In addition, we included any potential sampling covariates that had an effect on forest-use estimates of our focal species. However, we excluded 'trap nights' (because underpass cameras were stationary and were largely deployed for similar durations) and 'rainfall' (because the effect of rain could be confounded by the sheltering effect of underpasses). The presence/absence of an underpass at the end of a trail (Appendix 11) was coded as our site covariate.

Nearing the completion our camera trapping surveys at trails in the eastern linkage, electric fences were constructed along the road in Block 1 by the government, ostensibly to channel animals away from the road and into the underpasses – this allowed a unique opportunity to carry out a "before and after" experiment to investigate whether the fences could induce a barrier effect. We deployed camera traps on trails (n = 18) that were cut off (within 5-10 m) by the fence for the same sampling duration and at the same locations used to quantify trail use (see preceding paragraph). We conducted a McNemar test on a 2x2 classification table to investigate whether there was a significant change in the detections/non-detections of the same five species before and after the fences were built.

RESULTS

Conservation importance of linkages for large mammals

Due to missing data from camera trap theft, malfunctions and damage, we could only utilise data from 158 of 168 possible camera trap stations in the eastern linkage, and 182 of 224 possible camera trap stations in the western linkage. A total of 37 and 35 native mammal species were recorded over 38,631 camera-trap nights at the eastern and western linkages, respectively (Appendices 12 & 13).

The eastern linkage appears to be of greater conservation importance for mammals compared to the western linkage. In terms of numbers of IUCN-threatened mammals (EN-VU), 17 and 13 species were recorded in the eastern and western linkages (Appendices 12 and 13), respectively. Evidence of their detections is provided in Appendix 14. Of Peninsular Malaysia's 12 extant native large mammal species, 100% and 75% were detected in the eastern and western linkages, respectively (Tables 14 and 15). For six focal large mammal species, model-averaged mean forest-use estimates were relatively high for all species ($\hat{\psi}_{\bar{f}} \geq 0.5$) in the eastern linkage, but were relatively low in the western linkage ($\hat{\psi}_{\bar{f}} \leq 0.5$) for three species (Fig. 18), the Asian Elephant (0.25), Asian Tapir (0.44) and Sun Bear (0.42). The non-overlaps of 95% CIs provide evidence that model-averaged mean forest-use estimates for four species, Barking Deer, Asian Elephant, Asian Tapir, and Sun Bear, were relatively higher in the eastern linkage (Fig. 18).

***Table 14**: Summary statistics for 12 large mammal species expected to occur in forests, underpasses and access trails in the eastern linkage, Terengganu, Peninsular Malaysia. Surveys were conducted between 07 April 2011 and17 March 2013. Guilds = herbivorous (H), carnivorous (C), and omnivorous (O) ecological guilds. N = independent detections (0.5 hr intervals); PCRI = Photographic Capture Rate Index (N/1000 trap nights); Stations = number of camera trap stations that detected the species. Species marked with * had sufficient data for forest-use and trail-use models.*

Guild	Common Name	Species	N	PCRI	Stations
		Forests (10,502 trap nights)			
H	Asian Elephant*	*Elephas maximus*	144	13.71	60
H	Asian Tapir*	*Tapirus indicus*	282	26.85	100
H	Barking Deer*	*Muntiacus muntjak*	1334	127.02	152
H	Gaur	*Bos gaurus*	0	0	0
H	Sambar Deer	*Rusa unicolor*	2	0.19	2
H	Serow	*Capricornis sumatraensis*	12	1.14	11
C	Clouded Leopard*	*Neofelis nebulosa*	74	7.05	42
C	Dhole	*Cuon alpinus*	5	0.48	5
C	Leopard	*Panthera pardus*	32	3.05	27
C	Malayan Tiger	*Panthera tigris jacksoni*	25	2.38	19
O	Sun Bear*	*Helarctos malayanus*	140	13.33	62
O	Wild Pig*	*Sus scrofa*	472	44.94	128
		Underpasses (11,278 trap nights)			
H	Asian Elephant	*Elephas maximus*	100	8.87	24
H	Asian Tapir	*Tapirus indicus*	186	16.49	23
H	Barking Deer	*Muntiacus muntjak*	680	60.29	30
H	Gaur	*Bos gaurus*	0	0	0
H	Sambar Deer	*Rusa unicolor*	0	0	0
H	Serow	*Capricornis sumatraensis*	0	0	0
C	Clouded Leopard	*Neofelis nebulosa*	1	0.09	1
C	Dhole	*Cuon alpinus*	0	0	0
C	Leopard	*Panthera pardus*	1	0.09	1

C	Malayan Tiger	*Panthera tigris jacksoni*	0	0	0
O	Sun Bear	*Helarctos malayanus*	10	0.89	8
O	Wild Pig	*Sus scrofa*	191	16.94	32

Trails terminating at underpass or road (16,066 trap nights)

H	Asian Elephant*	*Elephas maximus*	148	9.21	28
H	Asian Tapir*	*Tapirus indicus*	394	24.52	44
H	Barking Deer*	*Muntiacus muntjak*	907	56.45	48
H	Gaur	*Bos gaurus*	3	0.19	3
H	Sambar Deer	*Rusa unicolor*	7	0.44	5
H	Serow	*Capricornis sumatraensis*	5	0.31	5
C	Clouded Leopard	*Neofelis nebulosa*	7	0.44	5
C	Dhole	*Cuon alpinus*	1	0.06	1
C	Leopard	*Panthera pardus*	18	1.12	13
C	Malayan Tiger	*Panthera tigris jacksoni*	30	1.87	13
O	Sun Bear*	*Helarctos malayanus*	74	4.61	33
O	Wild Pig*	*Sus scrofa*	510	31.74	47

Table 15: *Summary statistics for 12 large mammal species expected to occur in forests and underpasses in the western linkage, Perak, Peninsular Malaysia. Surveys were conducted between 12 May 2012 and 17 February 2013. Guilds = herbivorous (H), carnivorous (C), and omnivorous (O) ecological guilds. N = independent detections (0.5 hr intervals); PCRI = Photographic Capture Rate Index (N/1000 trap nights); Stations = no. of camera traps that detected the species. Species marked with * had sufficient data for forest- and trail-use models.*

Guild	Common Name	Species	N	PCRI	Stations
		Forests (12,063 trap nights)			
H	Asian Elephant*	*Elephas maximus*	39	3.23	24
H	Asian Tapir*	*Tapirus indicus*	125	10.36	59
H	Barking Deer*	*Muntiacus muntjak*	884	73.28	163
H	Gaur	*Bos gaurus*	0	0	0
H	Sambar Deer	*Rusa unicolor*	20	1.66	7
H	Serow*	*Capricornis sumatraensis*	70	5.80	35
C	Clouded Leopard*	*Neofelis nebulosa*	73	6.05	54
C	Dhole	*Cuon alpinus*	0	0	0
C	Leopard	*Panthera pardus*	0	0	0
C	Malayan Tiger	*Panthera tigris jacksoni*	2	0.17	2
O	Sun Bear*	*Helarctos malayanus*	130	10.78	57
O	Wild Pig*	*Sus scrofa*	2079	172.35	184
		Underpasses (13,841 trap nights)			
H	Asian Elephant	*Elephas maximus*	27	1.95	12
H	Asian Tapir	*Tapirus indicus*	0	0	0
H	Barking Deer	*Muntiacus muntjak*	60	4.33	8
H	Gaur	*Bos gaurus*	0	0	0
H	Sambar Deer	*Rusa unicolor*	0	0	0
H	Serow	*Capricornis sumatraensis*	71	5.13	16
C	Clouded Leopard	*Neofelis nebulosa*	5	0.36	3
C	Dhole	*Cuon alpinus*	0	0	0
C	Leopard	*Panthera pardus*	0	0	0
C	Malayan Tiger	*Panthera tigris jacksoni*	0	0	0
O	Sun Bear	*Helarctos malayanus*	0	0	0
O	Wild Pig	*Sus scrofa*	2089	150.93	46

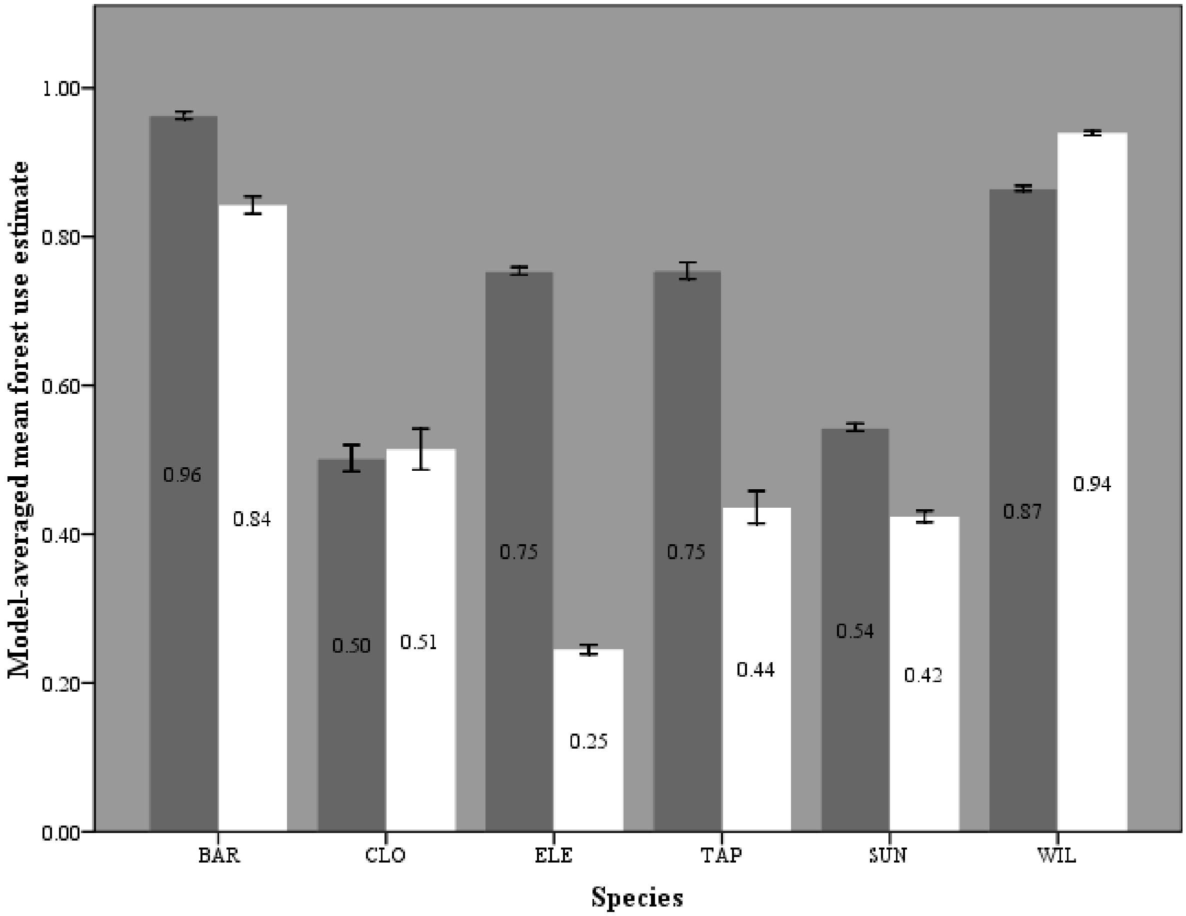

Fig. 18. *Comparison of model-averaged mean forest-use estimates ($\hat{\Psi}_{\bar{f}}$) and 95% confidence intervals (CIs) for six large mammal species in eastern (grey bars) and western (white bars) linkages in Peninsular Malaysia. Legend: BAR = Barking Deer; CLO = Clouded Leopard; ELE = Asian Elephant; TAP = Asian Tapir; SUN = Sun Bear; WIL = Wild Pig.*

Fragmentation effects on large mammal forest use around underpasses

Each sampling covariate we measured had an effect (Table 16), either individually or additively, on forest-use estimates of our focal large mammal species. In the eastern linkage, all three sampling covariates (number of trap nights, block type and rainfall) had an effect on the detection probability of our focal species after accounting for the effect of site covariates (Table 16). The effects of these sampling covariates on detection probability appeared to be relatively strong (see Møller & Jennions 2002) because the percentage deviance (% DE) of top-ranked models, which describes the proportion of variance in the response attributable to individual covariates, was relatively high (2.64-12.84%; Table 16). In the western linkage, both sampling covariates (number of trap nights and rainfall) had an effect on detection probability of six of seven focal large mammal species after accounting for the effect of site covariates. Again, the effects of these sampling covariates on detection probability appeared to be relatively strong because the percentage deviance (% DE) of top-ranked models was relatively high (1.23-8.02%; Table 16). Only one species, the Asian Elephant, did not appear to be affected by our sampling covariates (Table 16).

***Table 16**: Detection probability* (p) *models that explicitly accounted for all site covariates* ψ (all) *possibly affecting forest use for focal large mammal species in the eastern and western linkages in Peninsular Malaysia. Only species whose forest-use estimates* ($\hat{\psi}_f$) *are affected by site covariates and candidate models with* Δ*AICc < 2 and are shown. Models highlighted in grey are the 'best' models to explain forest use of a particular species based on an ideal combination of high %DE (relatively stronger covariate effect) and low k (parsimony).*

Candidate models	**AICc**	**ΔAICc**	**wAICc**	**k**	**DE**	**%DE**	$\hat{c}$
			Eastern Linkage				
Asian Elephant							
ψ(all),*p*(blok+rain+trap)	514.89	0.00	0.57	9	495.67	8.97	0.49
ψ(all),*p*(blok+trap)	515.47	0.58	0.43	8	498.50	8.45	NA
Asian Tapir							
ψ(all),*p*(blok+rain+trap)	828.68	0.00	0.88	9	809.46	5.48	0.95
Barking Deer							
ψ(all),*p*(trap)	960.09	0.00	0.44	7	945.34	12.84	1.10
ψ(all),*p*(rain+trap)	961.00	0.91	0.28	8	944.03	12.96	1.09
ψ(all),*p*(blok+trap)	961.94	1.85	0.17	8	944.97	12.87	1.10
Clouded Leopard							
ψ(all),p(blok+trap)	431.04	0.00	0.56	8	414.07	4.05	0.61
ψ(all),p(blok+rain+trap)	432.08	1.04	0.33	9	412.86	4.33	0.70
*Sun Bear**							
*ψ(*all),*p*(trap)	100.16	0.00	0.24	7	559.22	2.64	2.01
ψ(all),*p*(.)	100.29	0.13	0.23	6	574.39	0.00	1.84
ψ(all),*p*(rain+trap)	101.55	1.39	0.12	8	553.75	3.59	3.29
ψ(all),*p*(rain)	101.63	1.47	0.12	7	568.79	0.97	2.44
ψ(all),*p*(blok+trap)	101.94	1.78	0.10	8	556.31	3.15	2.80
ψ(all),*p*(blok)	101.97	1.81	0.10	7	571.06	0.58	2.18
Wild Pig							
ψ(all),*p*(trap)	995.82	0.00	0.54	7	981.07	7.71	0.77
ψ(all),*p*(blok+rain+trap)	997.18	1.36	0.28	9	977.96	8.00	0.77

Western Linkage							
Asian Tapir							
ψ(all),p(trap)	530.00	0.00	0.62	6	518.00	1.23	0.80
ψ(all),p(rain+trap)	531.55	1.55	0.29	7	517.55	1.32	0.80
Barking Deer							
ψ(all),p(trap)	1098.38	0.00	0.53	6	1085.90	8.02	0.94
ψ(all),p(rain+trap)	1098.65	0.27	0.47	7	1084.01	8.18	0.95
Clouded Leopard							
ψ(all),p(trap)	446.83	0.00	0.61	6	434.35	4.45	1.28
ψ(all),p(rain+trap)	447.75	0.92	0.39	7	433.11	4.72	1.24
Serow							
ψ(all),p(trap)	332.09	0.00	0.53	6	319.61	3.15	0.86
ψ(all),p(rain+trap)	332.39	0.30	0.45	7	317.75	3.72	0.99
Sun Bear							
ψ(all),p(rain+trap)	533.59	0.00	0.55	7	518.95	4.17	1.23
ψ(all),p(trap)	534.03	0.44	0.45	6	521.55	3.69	1.14
Wild Pig							
ψ(all),p(trap)	1119.67	0.00	0.73	6	1107.19	8.02	0.81
ψ(all),p(rain+trap)	1121.63	1.96	0.27	7	1106.99	8.04	0.81

***Note**: ψ (all) (see Table 4 for acronym definitions) = ψ (road+fors+plan+lake) for eastern linkage and ψ (road+fors+plan) for western linkage. AIC_c = Akaike's Information Criterion corrected for small sample size; ΔAIC_c = difference in AIC_c for each model from the most parsimonious model; $wAIC_c$ = AIC_c weight, k = number of parameters; DE = deviance; % DE = % deviance explained in the response variable by the model under consideration; $\hat{c}$ = overdispersion factor. Sampling covariates included in each model: trap = no. of trap nights that cameras were operational during each sampling occasion; block = block type; rain = daily rainfall. Species marked with * indicates models were evaluated based on quasi likelihood Akaike's Information Criterion corrected for small sample sizes ($QAIC_c$) due to evidence of overdispersion.*

Between the eastern and western linkage, large mammals in the former appear to be the less impacted by fragmentation. Fragmentation in the eastern linkage affected forest use of only three species, Clouded Leopard, Asian Tapir and Barking Deer, comprising carnivorous and herbivorous guilds. In the western linkage, fragmentation affected forest use of six species, Clouded Leopard, Asian Elephant, Asian Tapir, Barking Deer, Serow and Sun Bear, comprising carnivorous, herbivorous and omnivorous guilds. The top-ranked models for these affected species in the western linkage did not display strong evidence of over-dispersion ($\hat{c}$ = 0.78-1.58; Table 17). However, the percentage deviace (%DE) of these models, which describes the proportion of variance in the response attributable to individual fragmentation covariates, was relatively low (0.57-1.89%; Table 17), except for the Clouded Leopard (3.85%; Table 17). This implies that either the effects of fragmentation on forest use of large mammal species in both linkages are generally weak, or more important fragmentation correlates were not measured in our study.

Among our focal large mammal species in both linkages, fragmentation had the strongest effect on the Clouded Leopard. In the eastern linkage, the chosen forest-use model of this species had a higher %DE (1.07) than models of other focal species (Table 17). Beta coefficients indicated that forest use of this species declined with increasing proximity to the reservoir, with forest-use estimates predicted to fall below 50% at distances of < 2.5 km from the reservoir (Fig. 19). In the western linkage, the chosen forest-use model of this species also had a higher %DE (3.26) than models of other focal species (Table 17). Beta coefficients indicated that its forest-use estimates declined with less intact forest cover and increasing proximity to the road, with forest-use estimates predicted to fall below 50% at distances of < 4.0 km from the road (Fig. 19).

Table 17. *Forest-use* (ψ) *models that explicitly account for sampling covariates affecting detection probability* (p) *(Table 16) for focal large mammal species in eastern and western linkages, Peninsular Malaysia. Only species whose forest-use estimates* ($\hat{\psi}_f$) *are affected by site covariates and candidate models with* Δ*AICc < 2 and are shown. Models highlighted in grey are the 'best' models to explain forest use of a particular species based on an ideal combination of high %DE (relatively stronger covariate effect) and low k (parsimony).*

Candidate models	**AICc**	**ΔAICc**	**wAICc**	**K**	**DE**	**%DE**	$\hat{c}$
Eastern Linkage							
Asian Tapir							
ψ(lake),*p*(blok+rain+trap)	824.08	0.00	0.20	6	811.52	0.40	0.96
ψ(lake+plan),*p*(blok+rain+trap)	824.97	0.89	0.13	7	810.22	0.56	0.96
ψ(.),*p*(blok+rain+trap)	825.20	1.12	0.12	5	814.81	0.00	0.96
ψ(lake+plan+road),*p*(blok+rain+trap)	825.83	1.75	0.08	7	811.08	0.46	0.95
Barking Deer							
ψ(plan+road),*p*(trap)	957.28	0.00	0.18	5	946.89	0.76	1.11
ψ(forst+plan+road),*p*(trap)	957.90	0.62	0.13	6	945.34	0.92	1.11
ψ(plan),*p*(trap)	958.81	1.53	0.08	4	950.55	0.37	1.07
ψ(fors+plan),*p*(trap)	958.91	1.63	0.08	5	948.52	0.58	1.09
ψ(road),*p*(trap)	959.22	1.94	0.07	4	950.96	0.33	1.06
Clouded Leopard							
ψ(lake),*p*(blok+trap)	424.56	0.00	0.26	5	414.17	1.07	0.69
ψ(plan),*p*(blok+trap)	426.49	1.93	0.10	5	416.10	0.60	0.61
Western Linkage							
Asian Elephant							
ψ(plan),*p*(.)	231.49	0.00	0.27	3	225.36	1.01	0.92
ψ(.),*p*(.)	231.74	0.25	0.24	2	227.67	0.00	1.03
ψ(fors+plan),*p*(.)	233.07	1.58	0.12	4	224.84	1.24	0.49
Asian Tapir							
ψ(plan),*p*(trap)	527.52	0.00	0.56	4	519.29	1.89	0.78
ψ(road+plan),*p*(trap)	529.12	1.60	0.25	5	518.78	1.98	0.79

Barking Deer							
ψ(road),p(trap)	1094.85	0.00	0.48	4	1086.62	0.74	0.94
ψ(fors+road),p(trap)	1096.38	1.53	0.22	5	1086.04	0.79	0.94
*Clouded Leopard**							
ψ(fors+road),p(trap)	275.21	0.00	0.31	5	437.03	3.26	1.58
ψ(fors+plan+road),p(trap)	275.72	0.51	0.24	6	434.35	3.85	1.65
ψ(fors+plan),p(trap)	276.42	1.21	0.17	5	439.03	2.81	1.54
ψ(road),p(trap)	277.18	1.97	0.11	4	443.77	1.76	1.41
Serow							
ψ(road),p(trap)	328.88	0.00	0.43	4	320.65	1.74	0.92
ψ(plan+road),p(trap)	330.59	1.71	0.18	5	320.25	1.86	0.81
ψ(fors+plan+road),p(trap)	330.63	1.75	0.18	5	320.29	1.85	0.86
Sun Bear							
ψ(road),p(trap+rain)	428.19	0.00	0.24	5	520.30	0.57	1.18
ψ(.),p(trap+rain)	428.47	0.28	0.21	4	523.29	0.00	1.23
ψ(fors),p(trap+rain)	429.19	1.00	0.15	5	521.55	0.33	1.32
ψ(fors+road),p(trap+rain)	429.26	1.07	0.14	6	518.98	0.82	1.23
ψ(plan+road),p(trap+rain)	430.13	1.94	0.09	6	520.06	0.62	1.23

***Note**: Site covariates included in each model: road = distance to road edge; plan = distance to nearest plantation edge; resv = distance to reservoir edge; fors = forest cover type (as a proxy of logging intensity). AIC_c = Akaike's Information Criterion corrected for small sample size; ΔAIC_c = difference in AIC_c for each model from the most parsimonious model; $wAIC_c$ = AIC_c weight, k = number of parameters; DE = deviance; % DE = % deviance explained in the response variable by the model under consideration; $\hat{c}$ = overdispersion factor. Sampling covariates included in each model: trap = no. of trap nights that cameras were operational during each sampling occasion; rain = daily rainfall. Species marked with * indicates models were evaluated based on quasi likelihood Akaike's Information Criterion corrected for small sample sizes ($QAIC_c$) due to evidence of overdispersion.*

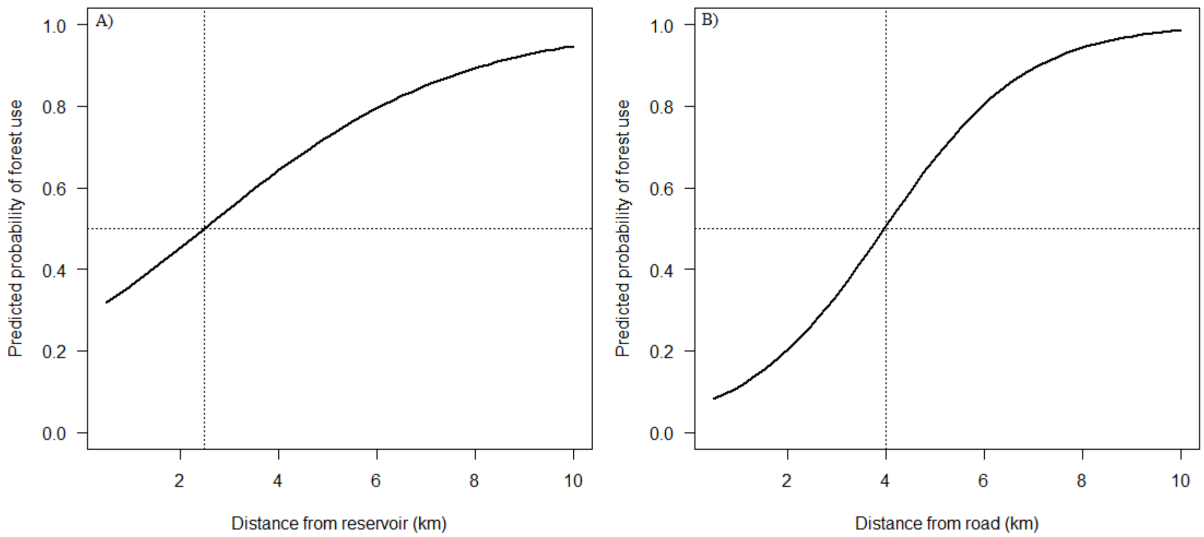

Fig. 19. *Relationship between predicted probability of forest use ($\hat{\psi}_f$) of the Clouded Leopard* (Neofelis nebulosa) *and distance from the (A) reservoir and (B) road in the eastern and western linkage, respectively, based on untransformed β coefficients from the chosen forest-use model (Table 17). Dotted lines = predicted probability of forest use of this species is 0.5 at (A) 2.5 km from the reservoir and 4.0 km from the road.*

Among the fragmentation covariates, the reservoir and road appeared to be the most important fragmentation covariates affecting large mammal forest use in the eastern and western linkage, respectively (Table 18). Beta coefficients indicated that the reservoir had a negative effect on forest use of the Clouded Leopard and Asian Tapir in the eastern linkage, while the road had a negative effect on forest use of large mammals across all ecological guilds - the Asian Elephant, Asian Tapir, Barking Deer, Clouded Leopard, Serow and Sun Bear in the western linkage (Table 18).

Effectiveness of underpasses as crossings structures for large mammals

A total of 18 native mammal species were detected at underpasses in each linkage (Appendices 15 and 16). Herbivores and omnivores were more commonly detected at underpasses than carnivores (see PCRI values in Tables 14 and 15). However, based on the mean differences in the number of detections (per camera trap) between forest sites and underpasses for seven focal large mammal species, it appears that all 20 underpasses are effective crossing structures for two herbivore species. Because the number of detections of a species in the forest was similar to that at an underpass (i.e. the 95% CI overlaps with zero; Fig. 20), our results show that underpasses in both linkages appear to be effective crossing structures for the Asian Elephant, while underpasses in the western linkage only are effective crossing structures for the Serow.

Table 18. *Directions and value (SE) of untransformed beta coefficients of fragmentation covariates affecting large mammal forest use in eastern (grey) and western (white) linkages from 'best' candidate models in Table 17.*

	Carnivore	**Herbivore**				**Omnivore**	
	CLO	**BAR**	**ELE**	**SER**	**TAP**	**SUN**	**WIL**
Forest quality				NA			
Distance to				NA			
Plantation		0.92(0.55)		NA			
Road		–0.69(0.39)		NA			
Reservoir	0.64(0.36)			NA	0.51(0.32)		
Forest quality	1.88(1.00)						
Distance to							
Plantation			0.40(0.27)		0.67(0.25)		
Road	0.88(0.48)	0.67(0.26)		0.52(0.23)		0.35(0.21)	

Note: Legend: CLO = Clouded Leopard; BAR = Barking Deer; ELE = Asian Elephant; SER = Serow; TAP = Asian Tapir; SUN = Sun Bear; WIL = Wild Pig.

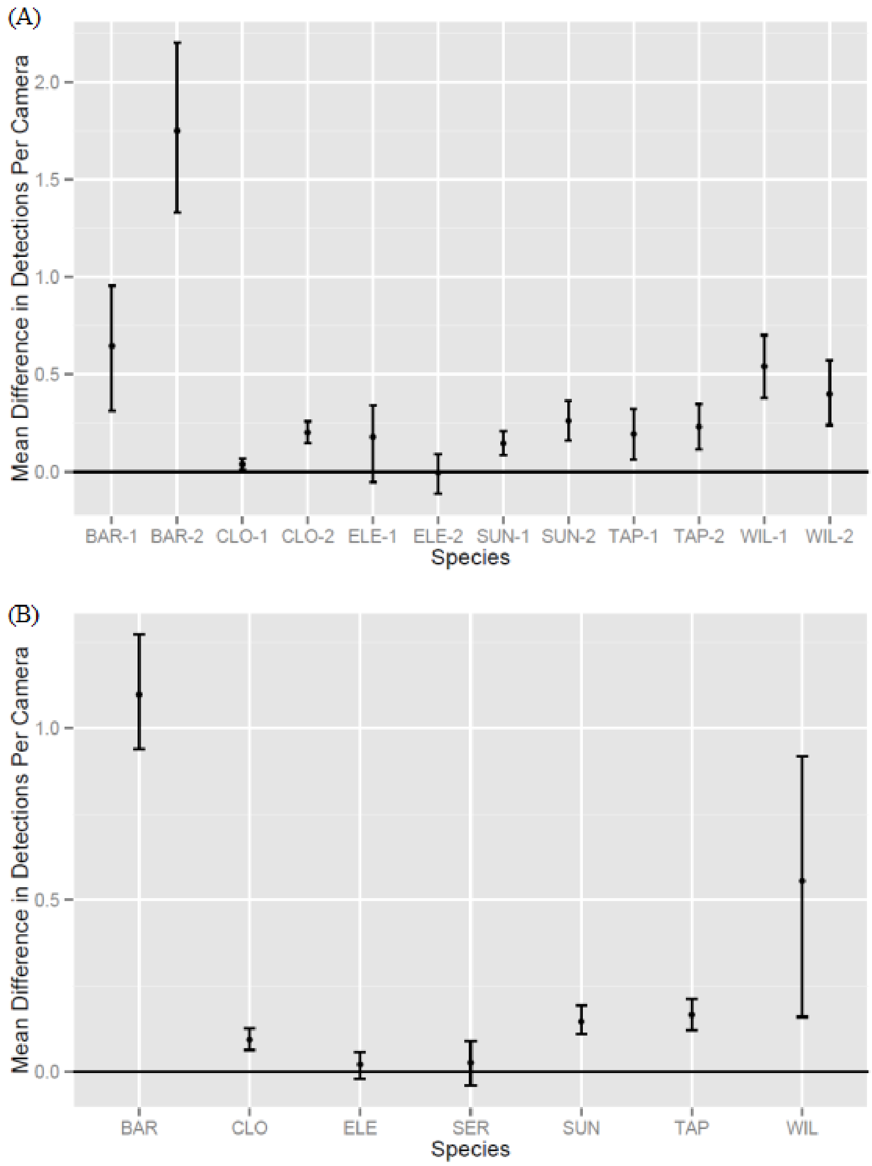

Fig. 20. *Plots of the mean difference in the number of detections (per camera trap) between forest sites and underpasses in each 15-day sampling occasion in the (A) eastern and (B) western linkage in Peninsular Malaysia. We computed the 95% non-parametric percentile bootstrap confidence interval (CI) of this statistic by sampling with replacement across sampling occasions 9999 times. Note: In (A), Bar-1 and Bar-2 indicates Barking Deer detections in Block 1 and 2 in the eastern linkage. Legend: CLO = Clouded Leopard; ELE = Asian Elephant; SER = Serow; SUN = Sun Bear; TAP = Asian Tapir; WIL = Wild Pig.*

Efficiency of underpass use by large mammals

Our underpass use indices, which account for forest use within the vicinity of each underpass and imperfect detection, indicate that underpasses in both linkages are not efficiently used by almost all focal large mammal species. In the eastern linkage, total underpass-use efficiency for each species was sub-optimal (η_s = 1-48%; Table 19). When we examine the underpass-use efficiencies for all species, they were also sub-optimal (η_a = 4-60%; Table 19), with underpass no. 2 ranked the highest (η_a = 60%; Table 19). In the western linkage, total underpass-use efficiency for each species was also sub-optimal (η_s = 0-69%; Table 20), except for the Serow (η_s = 101%; Table 20), which was heavily utilised two underpasses to cross the road. In fact, two large mammal species, Asian Tapir and Sun Bear (η_s = 0; Table 20), did not use underpasses even though they were expected to do so. However, underpass-use efficiencies for all species were generally sub-optimal (η_a = 11-87%; Table 20), with underpass no. 5 ranked the highest (η_a = 87%; Table 20).

Table 19: *Underpass use indices (UUIs) and efficiencies for each (η_s) and total number (η_a) of six focal large mammal species in the eastern linkage, Peninsular Malaysia.*

Underpass	CLO	ELE*	TAP	BAR	SUN*	WIL*	∑	η_a (%)
				Expected underpass use ($\hat{\psi}_u$)				
1	0.62	0.73	0.83	0.98	0.58	0.89		
2	0.57	0.77	0.82	0.99	0.56	0.86		
3	0.37	0.78	0.68	0.99	0.50	0.85		
4	0.31	0.80	0.63	0.99	0.49	0.84		
5	0.34	0.79	0.65	0.99	0.50	0.84		
6	0.34	0.79	0.63	0.99	0.50	0.84		
7	0.36	0.79	0.69	0.99	0.49	0.84		
8	0.45	0.76	0.71	0.98	0.54	0.87		
9	0.40	0.80	0.63	0.98	0.55	0.84		
10	0.51	0.75	0.72	0.96	0.57	0.88		
				Observed underpass use (ψ_u)				
1	0.00	0.15	0.70	1.00	0.15	0.30		
2	0.00	0.25	0.85	1.00	0.15	0.85		
3	0.05	0.00	0.35	0.75	0.05	0.75		
4	0.00	0.25	0.10	0.70	0.00	0.35		
5	0.00	0.05	0.00	0.00	0.05	0.05		
6	0.00	0.05	0.10	0.00	0.00	0.15		
7	0.00	0.00	0.00	0.00	0.00	0.50		
8	0.00	0.00	0.06	0.25	0.05	0.40		
9	0.00	0.00	0.65	0.80	0.05	0.65		
10	0.00	0.10	0.35	0.05	0.05	0.05		
				Underpass use index (UUI)				
1	0.00	0.21	0.85	1.02	0.26	0.34	2.67	44.54
2	**0.00**	**0.32**	**1.04**	**1.01**	**0.27**	**0.98**	**3.62**	**60.39**
3	0.14	0.00	0.51	0.76	0.10	0.89	2.39	39.89
4	0.00	0.31	0.16	0.71	0.00	0.42	1.60	26.64
5	0.00	0.06	0.00	0.00	0.10	0.06	0.22	3.73
6	0.00	0.06	0.16	0.00	0.00	0.18	0.40	6.65
7	0.00	0.00	0.00	0.00	0.00	0.60	0.60	9.96
8	0.00	0.00	0.08	0.25	0.09	0.46	0.89	14.78
9	0.00	0.00	1.03	0.81	0.09	0.77	2.70	45.05
10	0.00	0.13	0.48	0.05	0.09	0.06	0.81	13.55
∑	0.14	1.10	4.30	4.62	1.00	4.75		
η_s (%)	1.37	11.04	43.00	46.16	10.01	47.53		

*Note: (1) Underpass use index (UUI) = $\psi_u / \hat{\psi}_u$, where ψ_u is the observed underpass use, calculated by obtaining the proportion of total sampling occasions that a particular species was detected from pooled detection histories of all cameras beneath an underpass; and $\hat{\psi}_u$ is the expected underpass use estimate, calculated by averaging estimates of the probability of forest use ($\hat{\psi}_f$) by a particular species at two camera trap locations nearest to the underpass on opposite sides (2) Underpass use efficiency by a particular species (η_s) and for all species (η_a) = ratio of $\sum$UUIs to the maximum possible UUI (i.e. 10 for each species, and 6 for all species) (3) UUIs for species marked with * have to be treated with caution because expected underpass use estimates were based on forest-use estimates derived from constant detection probability (4) Underpass marked in bold warrants priority conservation attention due to high underpass-use efficiency for focal species. Legend: CLO = Clouded Leopard; ELE = Asian Elephant; TAP = Asian Tapir; BAR = Barking Deer; SUN = Sun Bear; WIL = Wild Pig.*

***Table 20**: Underpass use indices (UUIs) and efficiencies for each (η_s) and total number (η_a) of seven focal large mammal species in the western linkage, Peninsular Malaysia.*

Underpass	CLO	ELE	TAP	BAR	SER	SUN	WIL*	∑	η_a
				Expected underpass use ($\hat{\psi}_u$)					
1	0.38	0.22	0.33	0.71	0.12	0.38	0.93		
2	0.35	0.27	0.52	0.77	0.14	0.37	0.95		
3	0.32	0.28	0.55	0.70	0.12	0.35	0.94		
4	0.33	0.26	0.52	0.82	0.23	0.38	0.97		
5	0.12	0.19	0.23	0.70	0.13	0.32	0.98		
6	0.15	0.22	0.35	0.75	0.14	0.33	0.98		
7	0.14	0.21	0.31	0.73	0.13	0.33	0.98		
8	0.27	0.23	0.36	0.72	0.13	0.35	0.95		
9	0.44	0.27	0.49	0.71	0.12	0.38	0.92		
10	0.35	0.20	0.23	0.74	0.14	0.39	0.93		
				Observed underpass use (ψ_u)					
1	0.20	0.00	0.00	0.00	0.07	0.00	1.00		
2	0.00	0.00	0.00	0.00	0.00	0.00	0.73		
3	0.00	0.00	0.00	0.00	0.00	0.00	1.00		
4	0.00	0.07	0.00	0.00	0.00	0.00	0.60		
5	0.00	0.13	0.00	0.00	0.60	0.00	0.67		
6	0.00	0.00	0.00	0.00	0.53	0.00	0.00		
7	0.00	0.00	0.00	0.00	0.13	0.00	0.00		
8	0.00	0.07	0.00	0.00	0.00	0.00	0.53		
9	0.00	0.13	0.00	0.47	0.00	0.00	1.00		
10	0.00	0.00	0.00	0.00	0.00	0.00	0.93		
				Underpass use index (UUI)					
1	0.53	0.00	0.00	0.00	0.56	0.00	1.08	2.17	31.02
2	0.00	0.00	0.00	0.00	0.00	0.00	0.77	0.77	11.02
3	0.00	0.00	0.00	0.00	0.00	0.00	1.06	1.06	15.12
4	0.00	0.26	0.00	0.00	0.00	0.00	0.62	0.88	12.59
5	**0.00**	**0.68**	**0.00**	**0.00**	**4.75**	**0.00**	**0.69**	**6.12**	**87.44**
6	0.00	0.00	0.00	0.00	3.77	0.00	0.00	3.77	53.91
7	0.00	0.00	0.00	0.00	0.98	0.00	0.00	0.98	13.96
8	0.00	0.31	0.00	0.00	0.00	0.00	0.56	0.87	12.39
9	0.00	0.49	0.00	0.65	0.00	0.00	1.08	2.23	31.82
10	0.00	0.00	0.00	0.00	0.00	0.00	0.99	0.99	14.21
∑	0.53	1.75	0.00	0.65	10.07	0.00	6.84		
η_s	5.33	17.46	0.00	6.54	100.67	0.00	68.45		

Note: (1) Underpass use index (UUI) = $\psi_u / \hat{\psi}_u$ *, where* ψ_u *is the observed underpass use, calculated by obtaining the proportion of total sampling occasions that a particular species was detected from pooled detection histories of all cameras beneath an underpass; and* $\hat{\psi}_u$ *is the expected underpass use estimate, calculated by averaging estimates of the probability of forest use (*$\hat{\psi}_f$*) by a particular species at two camera trap locations nearest to the underpass on opposite sides (2) Underpass use efficiency by a particular species (*η_s*) and for all species (*η_a*) = ratio of* $\sum$*UUIs to the maximum possible UUI (i.e. 10 for each species, and 7 for all species) (3) UUIs for species marked with * have to be treated with caution because expected underpass use estimates were based on forest-use estimates derived from constant detection probability (4) Underpass marked in bold warrants priority conservation attention due to high underpass-use efficiency for all species. Legend: CLO = Clouded Leopard; ELE = Asian Elephant; TAP = Asian Tapir; BAR = Barking Deer; SUN = Sun Bear; WIL = Wild Pig.*

Fragmentation effects on underpass use efficiencies of each large mammal species

For each of the six focal species in each linkage, we found that odds ratios of their forest-use models (*n=12*) with distance to road as the sole predictor, had a negative correlation (Spearman's Rho [rs]= -0.73; Fig. 21) with the underpass use effiency of each species *(η_s).* When we examined each linkage individually (*n* =6), a negative correlation was similarly observed. Thus, we maintained the combined correlation analysis of 12 pairs of variables. To summarize, the greater the magnitude of negative effects of distance to the road on forest use by a particular large mammal species (when odds ratio > 1), the lower would be its efficiency of underpass use.

Ability of underpasses to mitigate road impacts on large mammals

A total of 34 native mammal species were detected on trails leading to the underpass or the road in the eastern linkage (Appendix 17). However, we did not find any evidence of underpasses mitigating road impacts in the eastern linkage. Because trail-use models that accounted for the effect of underpass presence were not the top-ranked models and had ΔAICc values of > 2, trails leading to underpasses are probably not 'attracting' large mammal species away from trails leading to roads. Furthermore, it appears that the electric fences have not been effective at preventing road crossings thus far, because there were no significant changes in the detections of each species on trails before and after exclusion fences were constructed.

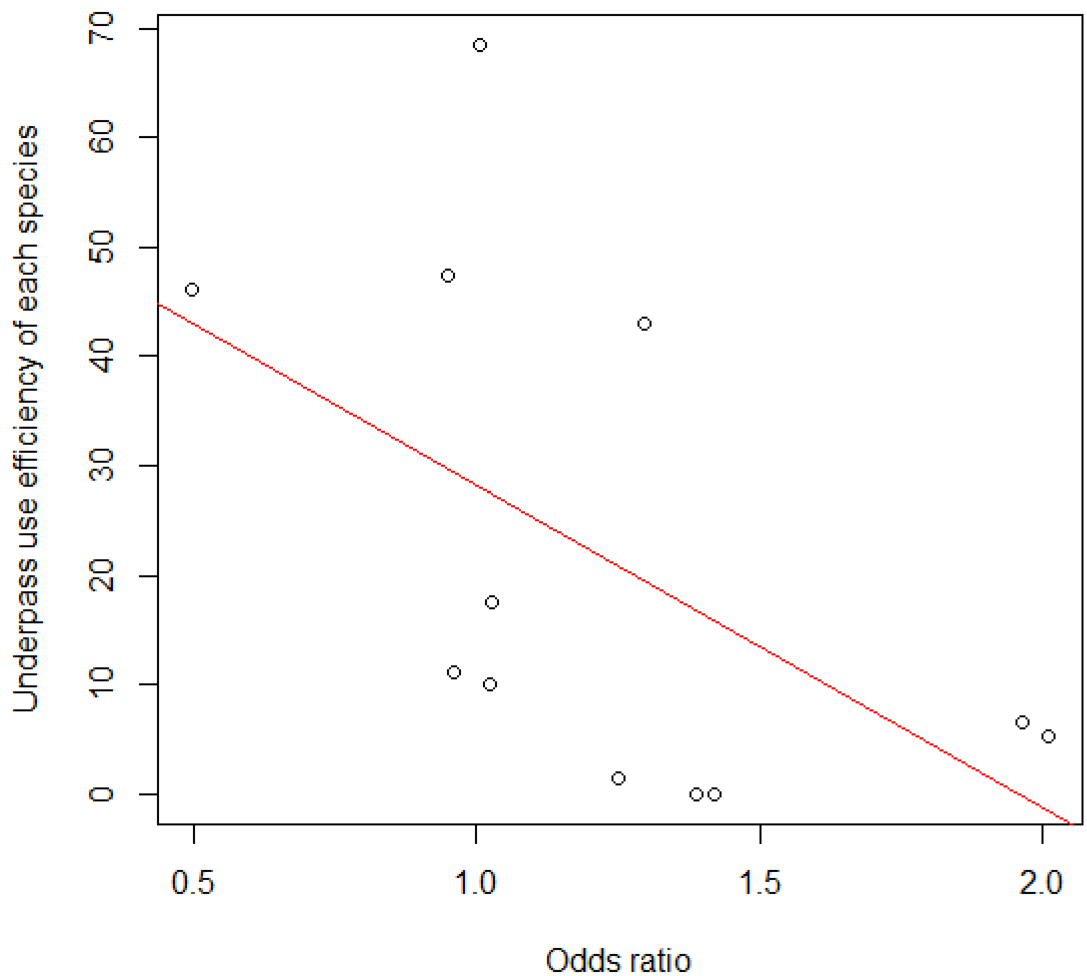

Fig. 21. *Spearman's rank-order correlation between the underpass use effiency (η_s) of each focal species (i.e. Asian Elephant, Asian Tapir, Barking Deer, Clouded Leopard, Sun Bear and Wild Pig) in each linkage and odds ratios of their forest-use models (n=12) with distance to road as the sole predictor. The negative correlation (Spearman's Rho [rs]= -0.73; Fig. 21) indicates that the greater the magnitude of negative effects of distance to the road on forest use by a particular species (when odds ratio > 1), the lower would be its efficiency of underpass use.*

DISCUSSION

Our study has contributed to a greater understanding of mammalian fauna in two important habitat linkages within Peninsular Malaysia. To our knowledge, the findings from this study represent the most comprehensive information available on the role of underpasses as crossing structures for mammals in the country, and possibly Southeast Asia.

Conservation importance of linkages for large mammals

Although many native mammals were recorded in the eastern and western linkages, we consider the former to be more important for mammal conservation based on the occurrence of (1) a larger number of IUCN threatened mammal species (2) a higher proportion of large mammal species and (3) relatively higher forest-use estimates for four threatened large mammal species. Prior to this study, only one other linkage in the country had been surveyed for mammals (PL 2; Fig. 1; Rayan et al. 2012a). In addition, the eastern linkage appears to be a very important conservation site for the Malayan Tiger because 11 individuals were recorded, compared to just one individual tiger recorded in the western linkage.

Although the forests in both linkages have been selectively logged, our study shows that they still contain high mammalian species richness – this underscores the conservation importance of selectively logged forests for native mammals (Rayan & Mohamad 2009; Giam et al 2011; Gibson et al. 2011; Clements et al. 2012; Rayan et al. 2012b; Edwards & Laurance 2013). Because logged forests in Malaysia are currently threatened by forest conversion to rubber plantations (Aziz et al. 2010), which can markedly reduce mammalian diversity (Fitzherbert et al. 2008), urgent measures should be taken to limit forest conversion in both linkages. One strategy to advance this aim is to change the status of the production forest reserves to a strictly protection category that prevents forest conversion.

Fragmentation effects on large mammal forest use around underpasses

Among our focal large mammal species, the Clouded Leopard was the most sensitive to fragmentation. This species appeared to avoid areas nearer to the reservoir, possibly due to the lack of canopied trees that provide cover and foraging grounds important for such carnivores. Such avoidance has also been documented in other large carnivores such as tigers (Sunarto et al. 2012). Given that forest-use estimates are predicted to fall below 50% at distances of < 2.5 km from the reservoir (Fig. 19A) and that Clouded Leopard was detected only once beneath underpasses near the reservoir (<1.7 km away), the expansion or creation of new reservoirs should be avoided as a precautionary measure to minimise further declines in this species's forest use (and possibly underpass use). Second, forest-use estimates of Clouded Leopard appear to decline in areas with less intact forest cover. Intact well-canopied forests are probably more intensely used by this species in order to capture arboreal mammals such as primates, which appear to be its preferred prey (Santiapillai 1986; Athreya & Johnsingh 1995). In Thailand, a radio-tracking study similarly detected Clouded Leopard more often in primary forests (Austin et al. 2007b). As such, corrective measures involving reforestation and arresting further fragmentation may have to be implemented. Finally, roads can also negatively affect Clouded Leopard forest use. This phenomenon has also been documented for other large carnivores such as tigers (Linkie et al. 2006). Logging roads appear to be utilised by Clouded Leopard (Wilting et al. 2006), but paved roads may be more of a deterrent. Given that Clouded Leopard forest-use estimates are predicted to fall below 50% at distances of < 4.0 km from the road in the western linkage (Fig. 19B), and that this species is more likely to be detected in intact forests than at an underpass (Table 17; Fig. 20), corrective measures involving reforestation should be implemented within a ~4.0 km buffer from the road as a precautionary measure to minimise further declines in forest use (and possibly underpass use).

Fragmentation can have varying degrees of effects on large mammal communities. Across all focal large mammal species, our results show that the presence of the reservoir was the most important fragmentation covariate negatively affecting forest use in the eastern linkage. Fragmentation caused by the creation of reservoirs was recently shown to be responsible for extinctions among mammal communities in Thailand (Gibson et al. 2013). In the western linkage, the presence of the road was the most important fragmentation covariate negatively affecting large mammal forest use. Studies elsewhere have also shown that mammal communities can be negatively affected by road presence (Vanthomme et al. 2013). In the eastern linkage, however, there was a positive effect of the road on forest use of the Barking Deer, possibly due to the presence of grazing grounds at grassy road verges. Indeed, positive associations of mammals with roads have been documented for deer species in other countries (Vanthomme et al. 2013).

Effectiveness of underpasses as crossing structures for large mammals

Although herbivores and omnivores have been more commonly detected at all 20 underpasses than carnivores (see PCRI values in Tables 14 and 15), they appear to be particularly effective crossing structures for only two herbivore species, the Asian Elephant and Serow (Fig. 20). Possible reasons for this include the availability of grazing grounds beneath underpasses and the relative openness between underpass columns, which may provide sufficient response time to escape predators (e.g. Clevenger & Waltho 2005). Because of high herbivore and omnivore use, however, underpasses could already or eventually function as prey-traps, which are areas where carnivores frequent to capture prey (Little et al. 2002; Dickson et al. 2005). This is undesirable from a wildlife management perspective because the role of an underpass is to facilitate animal *crossings*. Response variables that have been used to test the prey-trap hypothesis include the density and proximity of kill sites as a function of distance to crossing structures, particularly based on transect and telemetry data (Ford & Clevenger 2010). However, we did not detect any kill sites via sign surveys in and around the underpasses, and among four large carnivores present in surrounding forests, Leopard and Clouded Leopard were each detected just once under the underpasses, whereas Malayan Tiger and Dhole were not detected at all (Table 14). A possible reason is that the prey base for large carnivores is sufficient within the forests, based on relatively high forest-use estimates for prey such as Barking Deer and Wild Pig (Fig. 18). Although our data suggest that underpasses in both linkages do not function as prey-traps, further research is needed to explicitly test the prey-trap hypothesis.

All 20 underpasses appear to be ineffective crossing structures for carnivores (Fig. 20). For this ecological guild, reasons for avoiding crossing structures may be species- or landscape-specific (e.g. Clevenger & Waltho 2000). One possible reason for this is that fragmentation threats (e.g. roads; Fig. 21) may be deterring carnivore species from using forests around underpasses. Indeed, our results showed that fragmentation clearly affected forest use of the Clouded Leopard in both linkages. Yet another possible reason is that the road may not even be a barrier to carnivores. Based on detections by our camera traps on either side of the road and beneath underpasses, two individual Malayan Tigers even crossed the road without using the underpasses in the eastern linkage. In fact, large carnivores such as Leopards are known to cross roads without much difficulty (Ngoprasert et al. 2007). Finally, another possible reason for avoidance of underpasses by carnivores could be high levels of human activity, which has been shown to negatively affect trail use by large mammals (Rogala et al. 2011), including carnivores (Clevenger & Waltho 2000; Ngoprasert et al. 2007). Indeed, humans were more frequently detected at underpasses in both linkages than in forests (Appendix 18), especially for camping, fishing and sometimes hunting activities.

Efficiency of underpasses use by large mammals

The efficiency at which individual underpasses have been used by large mammals has been sub-optimal to date. Nevertheless, we recommend that the two underpasses that were most efficiently used by all species in each linkage recieve priority protection to minimise anthropogenic threats (Tables 18 and 19). Although we measured structural (e.g. length, width) or landscape attributes (e.g. distance to nearest forest) for 20 underpasses, we considered this sample size too small to able to elucidate statistically meaningful factors responsible for their relatively high efficiency of use. Several management interventions can be implemented to increase underpass use efficiencies overall.

Beneath underpasses, warning signs may help deter human activity, particularly at the western linkage where detections of humans beneath underpasses were more than twice that of the eastern linkage (PCRI = 226 vs. 111; Appendix 18). Warning signs to prevent people from bringing domesticated animals to underpasses should also be erected, particularly in the western linkage where cattle herds and dogs have been detected at underpasses (Appendix 18). This is important to minimise competition between cattle and native browsers (Chaiyarat & Srikosamatara 2009), and between dogs and native carnivores (Vanak & Gompper 2009).

Regular law enforcement patrols should also be conducted at undpassess to deter hunting, which may be one reason for low efficiencies of underpass use by large mammals. Roads have been shown to increase access for hunting of mammals (Yakulic et al. 2011; Vanthomme et al. 2013). In fact, our surveys in the eastern linkage detected numerous snare signs (n = 131) and encroachment camps (n = 125), both of which were more frequently detected near to rather than far from the road (Fig. 9). Finally, reforestation of open areas between forest edges and underpasses may help improve the efficiency of use by carnivores, which need greater hiding cover. Passage rates for cougars, for example, were found to be negatively correlated with distance to forest cover (Clevenger & Waltho 2005). Tigers may eventually use the underpasses if there is thicker forest undercover (Sunarto et al. 2013). Ultimately, any management intervention must be accompanied by a long-term monitoring programme to assess its effectiveness.

Our Underpass Use Index provides an alternative to methods (see Clevenger & Waltho 2005; McCollister & van Manen 2010) that have been developed to quantify underpass use. We account for the intensity of forest use by a species within the vicinity of an underpass, as well as imperfect detection. Indeed, we have shown that sampling covariates such as the amount of rainfall or the number of nights that camera traps are deployed can affect the detection probabilities of mammals (Table 16); similar effects have also been reported previously for the Malayan Tiger and Sambar (Darmaraj 2012). When quantifying forest use around underpasses, it is important to account for rainfall as a sampling covariate, especially when habitat use can differ in dry and wet seasons for certain species (e.g. the Clouded Leopard; Austin et al. 2007b). Likewise, it is essential to incorporate trap nights as a sampling covariate because detection probability decreases when operational times for camera traps are shorter. Further, we have demonstrated that it is important to incorporate disconnected study sites (e.g. Block 1 and Block 2 in the eastern linkage) as a sampling covariate. For example, Block 1 (Fig. 16), which is a known release site for translocated Asian Elephants, would certainly have higher detection probabilities of this species around underpasses within the block.

While we have accounted for imperfect detection in the expected underpass use parameter of the index, we acknowledge that our observed underpass-use parameter is essentially a naïve underpass-use estimate that does not account for imperfect detection – this would underestimate actual underpass use if detection probability is not equal to one. To address this, we deployed camera traps at each underpass column, and pooled detections from all cameras under each underpass to maximise detection probability. Nevertheless, future studies should strive to survey a larger number of underpasses (n = 50) in order to obtain enough variability in site covariates to obtain observed underpass-use estimates that account for imperfect detection.

Most importantly, we have shown that the the underpass use efficiencies of a particular species are strongly linked with the negative impacts from fragmentation, such as those resulting from the presence of the road (Fig. 21). Therefore, the effects of fragmentation (e.g. from the road) on forest use of large mammals around underpasses should be minimised in order to maximise the efficiency at which they are used by large mammals.

Ability of underpasses to mitigate road impacts on large mammals

We did not find evidence that underpasses are actually mitigating road impacts in the eastern linkage, where mammal roadkills have been detected. In fact, two of the carnivore roadkills, one of a Leopard (*Panthera pardus*) and the other a Leopard Cat (*Prionailurus bengalensis*), were actually detected on the underpasses (rather than underneath). Very few studies have examined whether the impacts of roads on animals have actually been mitigated following the construction of crossing structures (Lehnert & Bissonette 1997; Dodd Jr. et al. 2004). Indeed, most study durations are too short to differentiate transient from long-term effectiveness of corridors (van der Grift et al. 2013), and very few employ before-after designs (Glista et al. 2009). While we considered such designs to improve our current study, they were impossible to incorporate given the constraints of the landscape available to us (see Material and Methods). Instead, we adopted a treated-untreated design, where we investigated use of trails that either terminated at the road or an elevated portion of the road (i.e an underpass). However, we acknowledge that any potential effects of underpass presence may be masked by confounding factors such as resource availability or predator-prey interactions along different trails. Also, our experimental set-up suffers from an inability to confirm actual animal crossings from one trail to the other, due to the presence of open grass verges beside roads (Appendix 11) and the lack of directly opposable trails.

The effectiveness of wildlife-exclusion fences to channel animals away from roads and into underpasses remains debatable. Fences have been erected along roads with heavy traffic to reduce mortality of medium- to large-sized mammals (Groot & Hazebroek 1996; Romin & Bissonette 1996), with some success (Putman 1997). In fact, one study suggested that the effectiveness of wildlife crossing structures such as overpasses in facilitating connectivity and gene flow may be attributed to fences, which serve to prevent road use and simultaneously increase the use of crossing structures (Corlatti et al. 2009). Indeed, the absence of operational wildlife-exclusion fences may explain why all 20 underpasses are inefficiently used by large mammals, and why we did find evidence of a ‘funnelling effect’ for five large mammal species at underpasses after electric fences were constructed. A future design involving the simultaneous deployment of camera traps at trails and underpasses over a longer duration is needed to ascertain whether there has been an increase in detections of mammals since the fences were erected. Because fences can sometimes be more detrimental than roads to migratory wildlife populations (Homewood et al. 2010), genetic studies should be conducted to monitor whether gene flow of large mammals is affected.

CONCLUSIONS

While our analyses have shown that underpasses in Peninsular Malaysia do not appear to be mitigating road impacts (i.e. no difference in use of trails that either terminated at the road or at an underpass), it could also be argued that as long as they are being used at some moderate level, it is indicative of a capacity to move across the road (and maintain connectivity, gene flow etc). If the primary function of an underpass is to serve as a crossing structure to prevent population isolation by the road, then it appears to have some potential to fulfill that role, at least for herbivores. Ultimately, conservation planners and practitioners have to recognise that it may be unrealistic to expect underpasses to function as effective crossing structures for all mammalian ecological guilds. For example, certain species may still continue to cross roads despite the presence of crossing structures (Tigas et al. 2002). As studies have also shown that animal passage rates are influenced by underpass structural attributes (McDonald & St-Clair 2004; Clevenger & Waltho 2000; Clevenger et al. 2001; Clevenger & Waltho 2005; Gagnon et al. 2011), future studies should aim to identify key structural predictors of underpass use by large mammals in Peninsular Malaysia.

End of chapter 5

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Chapter 6: General Conclusions

Here, I emphasise the gravity of the threats posed by the proliferation of roads in Southeast Asia. Based on my research findings, I also discuss how conservation researchers, planners and practitioners can overcome challenges when mitigating road impacts in this region.

The burgeoning network of roads in Southeast Asia clearly represents yet another threat to the region's biodiversity, which is already being ravaged by threats such as industrial logging, forest fires, wildlife overexploitation, invasive species and disease (Koh et al. 2013). In East Asia, the proportion of paved roads nearly tripled in a span of just five years between 2005 and 2010 (World Bank 2013). In Borneo, satellite images revealed that the density of logging roads is so high that, if placed end-to-end, these roads would circle Earth nine times (Brady et al. 2013). These statistics are particularly worrying for conservation practitioners working in developing countries, especially when road expansion is tied to economic growth and linked with land degradation (Wilkie et al. 2000).

I have shown that the environmental impacts of existing roads in Southeast Asia are predominantly negative. Not only do roads fragment important mammal habitats, they contribute to forest conversion and illegal hunting and wildlife trade. Similar impacts have been documented in other tropical regions, such as the Amazon (Laurance et al. 2001; Nepstad et al. 2001) and Central Africa (Laurance et al. 2006, 2007; Blake et al. 2008). For the first time, we know where existing and planned roads are most threatening endangered mammals in Southeast Asia (Table 1). This would have been difficult without expert interviews, which have already proven to be a useful method in gaining insights into other contemporary threats to biodiversity (e.g. Laurance et al. 2012). In Peninsular Malaysia, for example, three roads identified by experts were shown to cut through habitats important for the endangered Asian Tapir (Fig. 2). In

Cambodia, the construction of the road bisecting Snoul Wildlife Reserve appears to have intensified forest conversion (Table 4). In Myanmar, road networks are clearly functioning as conduits for the illegal wildlife trade into border towns in neighbouring Lao PDR, Thailand and China (Fig. 10), where the market demand for animal products in traditional Chinese medicine is insatiable.

Apart from the threats posed by existing roads in Southeast Asia, conservation planners and practitioners in the region are also facing challenges to manage the environmental impacts of planned roads (Table 2). One poignant example is the planned expansion of the Ladia Galaska road scheme in northern Sumatra, which will entail around 500 km of roads that will cut through the remote forest interior of the Leuser Ecosystem at 14 different locations -- this is predicted to cause a 25% reduction of the entire Sumatran Orangutan population (Gaveau et al. 2009). In Kalimantan, another environmental crisis in the making is the Balang Island project, which involves the construction of a new road that threatens the survival of at least five endangered mammals (Hance 2010).

Several key lessons can be drawn from the nature of road development in Southeast Asia. For example, people do not always derive the ‘promised’ social benefits from roads (Robichaud et al. 2001). Instead, the construction of roads that are potentially damaging to the environment can be decided by political motivations (Gaveau et al. 2009) and government financial constraints (Sunderlin 2002). Road impacts also vary in different parts of Southeast Asia. In Malaysian Borneo, for example, logging roads can be a more important driver of biodiversity loss compared to paved roads (Curran et al. 2004). Due to varying reasons behind the construction of roads, and their varying degrees of impact, conservation planners and practitioners will face challenges unique to their own landscapes when mitigating road impacts.

One common challenge is the lack of funds to implement mitigation measures for roads over large spatial scales, especially in developing countries where donor support can be limited (see Waldron et al. 2013). If donors or conservation planners have limited resources to work with in a landscape where roads are known to cause negative environmental impacts, it would be advisable to focus on mitigating the impacts of roads that pass through habitats with the most number of species whose populations are at 'tipping points'. The SAFE index (Clements et al. 2011) that I developed provides a heuristic measure of the 'distance' of a species from extinction that incorporates an element of population viability (Fig. 12). This index could help guide decisions on allocating scarce resources for managing endangered mammals or wildlife near problem roads (Table 1). However, an index such as this, which relies on the use of a standard Minimum Viable Population (MVP) target for species, will often be controversial. Such approaches have their limitations, especially when other local factors, such as population connectivity, the degree of habitat fragmentation, source–sink dynamics, and disease susceptibility, can overwhelm extinction risk arising from stochastic disturbances. Empirically based alternatives to a standard MVP might exist and could work equally well under the same 'distance' principle embodied in the SAFE index. Nevertheless, the key point is that species can be assigned a quantifiable index of 'distance from extinction', in order to assess their relative vulnerability more objectively.

Conservation practitioners who wish to oppose a particular road on environmental grounds may also face a dilemma if the road can potentially bring socio-economic benefits. As I learned in Peninsular Malaysia, increasing proximity to roads appears to exert a positive influence on indigenous peoples' support for roads. Therefore, it may be difficult to garner support against undesirable roads from indigenous people living next to roads in important biodiversity areas. Similar scenarios have been reported from other tropical regions where proposed roads would bisect important ecosystems (Fyumagwa et al. 2013). If basic socio-economic needs are not met, hunting pressure can increase markedly near roads (Wilkie et al. 2000). Therefore, before conservation practitioners decide to lobby against undesirable roads, they should attempt to understand the views of resident communities on how planned road projects would affect the people's livelihoods and hunting practices. If the social impacts from a proposed road are potentially positive, a compromise should be reached (e.g. re-routing of the road) that does not jeopardise the surrounding biodiversity, but still manages to bring the desired socio-economic benefits.

Another challenge facing conservation planners in Southeast Asia is the paucity of information on the effectiveness of wildlife-crossing structures for roads. Because such structures can be expensive, there is a need to know whether they are worthwhile investments. Prior to my PhD, the role of road underpasses as crossing structures for large mammals, and the ability of such underpasses to mitigate road impacts, have mainly been investigated outside this region (e.g. Beckmann et al. 2010). Based on my findings at 20 underpasses in two fragmented habitat linkages in Peninsular Malaysia, it appears that these underpasses have been effective crossing structures only for large herbivores so far. One possible reason is that fragmentation effects, such as those arising from the presence of the road, have limited the efficiency at which they are used. Key management interventions to improve efficiency of underpass use include regulating the negative effects from the road (e.g. high vehicle traffic) and reforestation of open areas between forest edges and underpasses. Further, because human activity near underpasses can deter the use of these underpasses by large mammals, measures such as regular law enforcement patrols and warning signs beneath underpasses might be useful to deter human presence.

Such management interventions should be accompanied by long-term monitoring to assess the effectiveness of such interventions. For future studies that require the efficiency of underpasses to be assessed, the Underpass Use Index developed in this thesis can be used as a scientifically defensible alternative to other methods (see Clevenger & Waltho 2005; McCollister & van Manen 2010), mainly because it considers the intensity of forest use by a species within the vicinity of an underpass and strives to account for imperfect animal detection.

Although I found no clear evidence that underpasses mitigated road impacts for larger mammals, it is premature to conclude that these underpasses should no longer be built or used to facilitate animal road crossings in Malaysia. Better protection and management of existing underpasses and surrounding forests, coupled with long-term research to assess the efficacy of these underpasses for wildlife, may eventually improve the potential of underpasses to enhance road permeability.

Given the high extinction rates predicted for Southeast Asian biodiversity (Brook et al. 2003; Wilcove et al. 2013), conservation planners and practitioners need to develop new rules of engagement. For example, it is vital that they increase dialogue with road planners to limit potential threats from roads planned through important biodiversity areas. Further, at existing roads where the probabilities of illegal forest conversion and hunting pressure are high, increased law enforcement is essential. Another needed change is better transparency and accountability of results from environmental and social impact assessments of proposed roads, which have had limited success in mitigating the impacts of these proposed roads. Greater public awareness of the negative impacts of roads on biodiversity, which can be raised via public outreach campaigns, may also help to support opposition against undesirable roads.

Lobbying efforts, however, can only go so far without political will. Where possible, governments should strive to maintain and improve habitat connectivity at forested areas that have been fragmented by roads. Within roadside forests that are being selectively logged, sustainable forest management regimes should be improved to minimise threats from logging roads (Laurance et al. 2001; Meijaard & Sheil 2008). In countries where hunting pressure in roadside forests is high, law enforcement efforts should be strengthened to prevent wildlife poaching and trafficking along roads, particularly those leading to border checkpoints. To minimise the impacts of roads on people's livelihoods, land rights and tenure should be resolved prior to road construction. Wherever possible, projections of economic and biodiversity loss prior to road development should be conducted to guide decision-making involving the construction of new roads. Lastly, governments should explore compensation schemes that can minimise the need for, or impact of, roads in important biodiversity areas.

Research findings and recommendations are worthless if they are not translated into conservation action. As an example, I report on two conservation successes that I achieved during my PhD candidature. In 2010, I co-founded a non-profit research group known as Rimba (www.myrimba.org) that focuses on conservation research on Malaysian biodiversity.

Rimba©

Because we NEED a jungle out there.

Along a road near my project site in the eastern linkage, I witnessed frequent illegal hunting of the Large Flying Fox (*Pteropus vampyrus*), a migratory species that may become locally extinct as early as 2015 due to overhunting (Epstein et al. 2009). My observations supported our research findings that poachers prefer to hunt near roads. After conducting fieldwork in the eastern linkage, I would frequently drive to the road near the roost and deter illegal hunters as well as legal hunters who had exceeded their hunting quotas. This was merely a stop-gap measure, and a longer-term solution was needed to help populations at the roost to recover. Therefore, in December 2011, my colleagues and I at Rimba submitted a proposal to the Chairman of Industry, Trade and Environment Committee, Terengganu State Government (Appendix 19), which oversees environmental conservation and infrastructure development in the state. On 18 January 2012, the government issued a state-wide moratorium on licences to hunt flying foxes (Heng 2012). This was the first conservation success for Rimba.

As discussed above, it is important to engage government agencies that have a mandate to sanction road-building. During the camera-trapping surveys in the eastern linkage, I shared my findings on the rich mammalian diversity and worrying hunting pressure in the eastern linkage with the Terengganu Industry, Trade and Environment Committee. Given the biological importance of selectively logged forests in the eastern linkage (Chapter 5), and in view of the imminent threat of further fragmentation of these forests from infrastructure development, my colleagues and I at Rimba submitted another proposal (Appendix 20) on 24 October 2012 recommending collective gazettement and management of the forests adjoining the road as a protected area known as the Kenyir Wildlife Corridor. This was to help maintain forest connectivity and reduce poaching pressure in the linkage. On 7 November 2012, the state government imposed a moratorium on infrastructure development along the road, pending further assessments to improve the management of the linkage (Hance 2012). This was the second conservation success for Rimba.

Ultimately, roads should always serve as one component of an integrated development plan that has carefully considered the environmental and social implications through stringent impact assessments. Although further research is needed to improve the effectiveness of existing road mitigation measures, it is imperative that research findings are transformed into on-the-ground action in order to limit the negative impacts of roads on biodiversity and people in Southeast Asia.

End of chapter 6

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Appendix 1: *Hierarchically-nested combinations of relevant keywords and wildcards used to search publication titles between 1985 and 2011 in the BIOSIS Previews® database for road-specific biodiversity studies in Southeast Asia.*

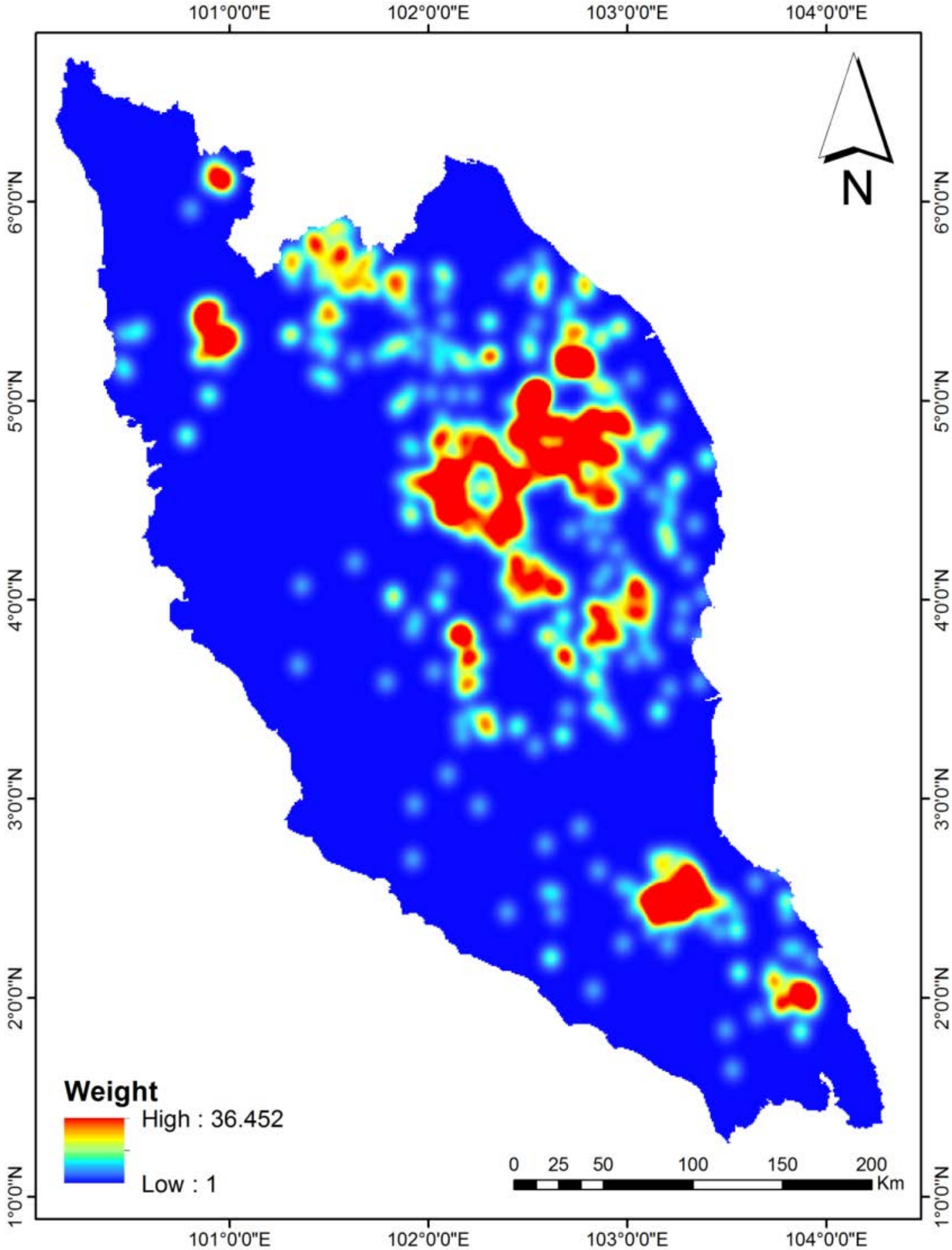
Road-specific biodiversity research worldwide

TI=(Biodiversity OR Conserv* OR Deforest* OR Diversity OR Ecolog* OR Extinction* OR Fauna* OR Flora* OR Forest* OR Fragment* OR Habitat OR Rainforest OR Species OR Wildlife) AND TI=(Road* OR Highway* OR Expressway OR Overpass* OR Over-pass OR Underpass* OR Under-pass* OR Viaduct OR Culvert OR Traffic OR Vehic* OR Road kill OR Roadkill)

Road-specific biodiversity research in Southeast Asia

TI=(Biodiversity OR Conserv* OR Deforest* OR Diversity OR Ecolog* OR Extinction* OR Fauna* OR Flora* OR Forest* OR Fragment* OR Habitat OR Rainforest OR Species OR Wildlife) AND TI=(Road* OR Highway* OR Expressway OR Overpass* OR Over-pass OR Underpass* OR Under-pass* OR Viaduct OR Culvert OR Traffic OR Vehic* OR Road kill OR Roadkill) AND TI= (Southeast Asia OR South East Asia OR SE Asia OR Borneo OR Brunei OR Indo* OR Malay* OR Philippine* OR Indo-China OR Indochin* OR Irian Jaya* Java* OR Kalimantan OR Cambodia* OR Lao* OR Burm* OR Myanmar OR Peninsular Malaysia OR Sabah OR Sarawak OR Singapore* OR Sumatra* OR Thailand OR Vietnam* OR Viet Nam OR East Timor OR Timor Leste OR Timor-Leste)

Appendix 2: *Bias grid for Peninsular Malaysia included in MaxEnt modelling to account for sample selection bias. Intensively sampled areas are indicated in red. Instructions on how to create it are from Supplementary Material in Clements et al. 2012.*



Required software

ArcMap, Geospatial Modelling Environment (http://www.spatialecology.com/gme/), and SPSS.

Instructions

1. Convert your species presence points to a raster. Use one of your Worldclim environment layers for MaxEnt as a reference layer in the environment settings before clicking ‘OK.’

2. Convert this raster back to points (this will remove duplicate points and obtain cell centers).

3. Convert the background raster (i.e. combined presences and pseudo-absences or cells that you want to develop the bias grid for) to points. So you now have 2 sets of points: one of species presence and one of background points.

4. Convert both layers to equi-distant projections (e.g. we used ‘Asian South equidistant conic’ for tapir presence points in Peninsular Malaysia).

5. Using GME, calculate the distance between the background and presence points. The output is a .csv file.

6. Depending on how many presence and background points you have, you can either continue with ArcMap or use SPSS for the calculations of Gaussian kernels. For a dataset with many presence and background points, SPSS is recommended.

ArcMap method

a. Use ArcCatalog and export the .csv file to a .dbf file.

b. Open the dbf file in ArcMap and add a new field (i.e. Options → Add field).

c. In this new field, using the ‘Field calculator’ option, apply the Gaussian function:

exp (- ([d] ^ 2) / (2 * s ^ 2)), where d = distances from GME tools calculations and s = standard deviation depending on your species home range size and/or sampling methods (see Elith et al. 2010). Note: Your Gaussian weights should never be >1. This is because closer points have higher weights, and applying the Gaussian function to a distance of 0 gives you an answer of 1. Now, when you sum the weights for each point, you will get an answer much greater than 1, but for each distance pair it should be <1. That is, your final grid can have values >1, but all of your point-to-point distance weights should be less than one.

d. Open the .dbf file in ArcMap, right-click on your new field, and choose 'Summarise.'

e. From the summarise window, select 'SOURCEUID' as the first entry (the summarise layer), and your new field and SUM as the second entries (you want to sum up the Gaussian weights within each background point, the identifiers of which are stored within the SOURCEUID column). This will create another .dbf file.

SPSS method

a. Open the .csv file in SPSS and calculate the Gaussian weights using the formula above in a new variable column.

b. Use the 'aggregate' function to sum up the Gaussian weights for each background point. Convert .csv file to .dbf file using ArcCatalog

6. The final .dbf file should have the same number of rows as your background grid. If this is not the case, something's gone wrong.

7. Join this second dbf to your original background points (in ArcMap, right-click in table of contents and choose join. Join based on the FID and OID columns of the 2 layers, which should match up). Now export the joined layer (in ArcMap, right-click in table of contents and choose Data →Export Data). Now you have a shapefile of your bias grid.

8. Convert the projection of your shapefile back to the original datum (WGS84 in our case, which is the datum used by Worldclim) before converting this joined point layer to a .asc file using using an environmental layer as a reference layer in environment settings. This is because MaxEnt requires your bias grid to be in .asc format.

9. Now you have a raster with the sum of weights of occurrences or a bias grid. If you want to account for edge effects around coastal areas, you should express the sum of weights of occurrences as a percentage of the sum of weights of all cells without missing data.

Appendix 3: *Confusion matrix used in accuracy analysis of 2010 classified image from Snoul Wildlife Reserve, Cambodia. Classification accuracy for the image was estimated on a per pixel basis using reference datasets of 500 randomly generated points. The reference datasets conisisted of the original Landsat 5 image and a Landsat 7 image obtained from a similar temporal period. These reference datasets were used because there was limited overlap in historical imagery available in Google Earth. As we could not obtain high spatial resolution images for ~1990 and ~2000, we could only assess the accuracy of the 2010 image. However, we expect the accuracy of classified images for 1990 and 2000 to be comparable to that of the 2010 image because it was produced with the same data and methods (e.g. Wijedasa et al. 2012). Overall accuracy was relatively high at 85% and the producer's accuracy was also generally high for all land-cover classes, ranging from 60-91%.*

		Ground Points					**Subtotal (Classified Pixel)**	**User's Accuracy**
		Bare or built-up	Mosaic	Mature Forest	Others	Water		
Classified Pixel	Bare or built-up	**118**	18	0	3	0	139	**84.89%**
	Mosaic	5	**137**	20	2	0	164	**83.54%**
	Mature Forest	0	15	**155**	0	0	170	**91.18%**
	Others	6	5	0	**9**	1	21	**42.86%**
	Water	0	0	0	1	**5**	6	**83.33%**
	Subtotal (Ground Points)	129	175	175	15	6	**500**	
	Producer's Accuracy	**91.47%**	**78.29%**	**88.57%**	**60.00%**	**83.33%**		
	Overall Accuracy	**84.80%**						

Appendix 4: *Google Earth mosaics of roads identified by 36 experts from seven countries and 10 sub-regions in Southeast Asia. Mosaics are not shown for Peninsular Malaysia because the roads are covered by our MaxEnt analysis (Fig. 2), and for Myanmar where data on specific roads were insufficient.*

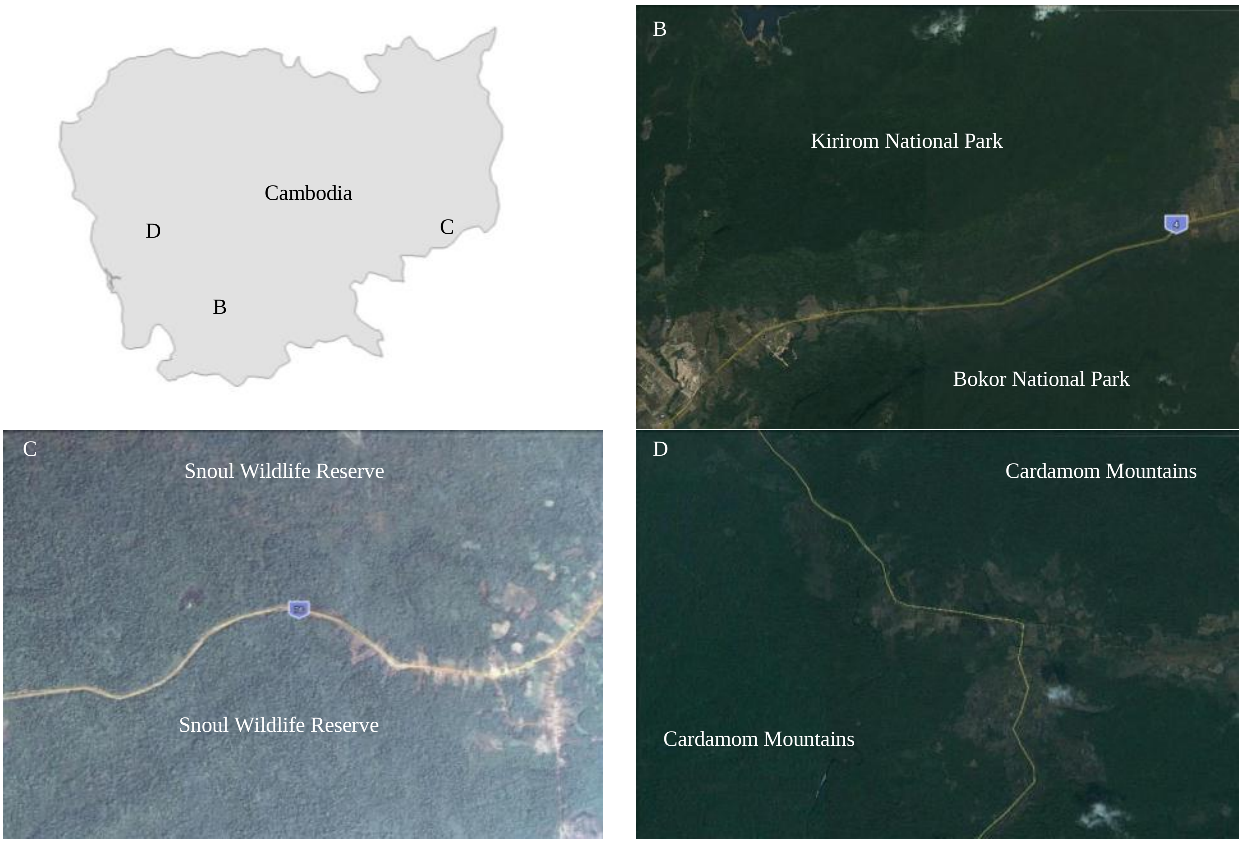


***Fig. 1**. A) Map of Cambodia; B) National Highway 4 bisecting Kirirom and Bokor National Parks with land use change occurring in both protected areas; C) Provincial Road 76 bisecting Snoul Wildlife Reserve with land use change occurring within the protected area; and D) National Road 48 bisecting Cardamom Mountains with land use changes occurring in protected areas. Roads = yellow line, Forest cover = green areas, land use change = brown areas. Images from Google Earth™ Mapping Services, 2010.*

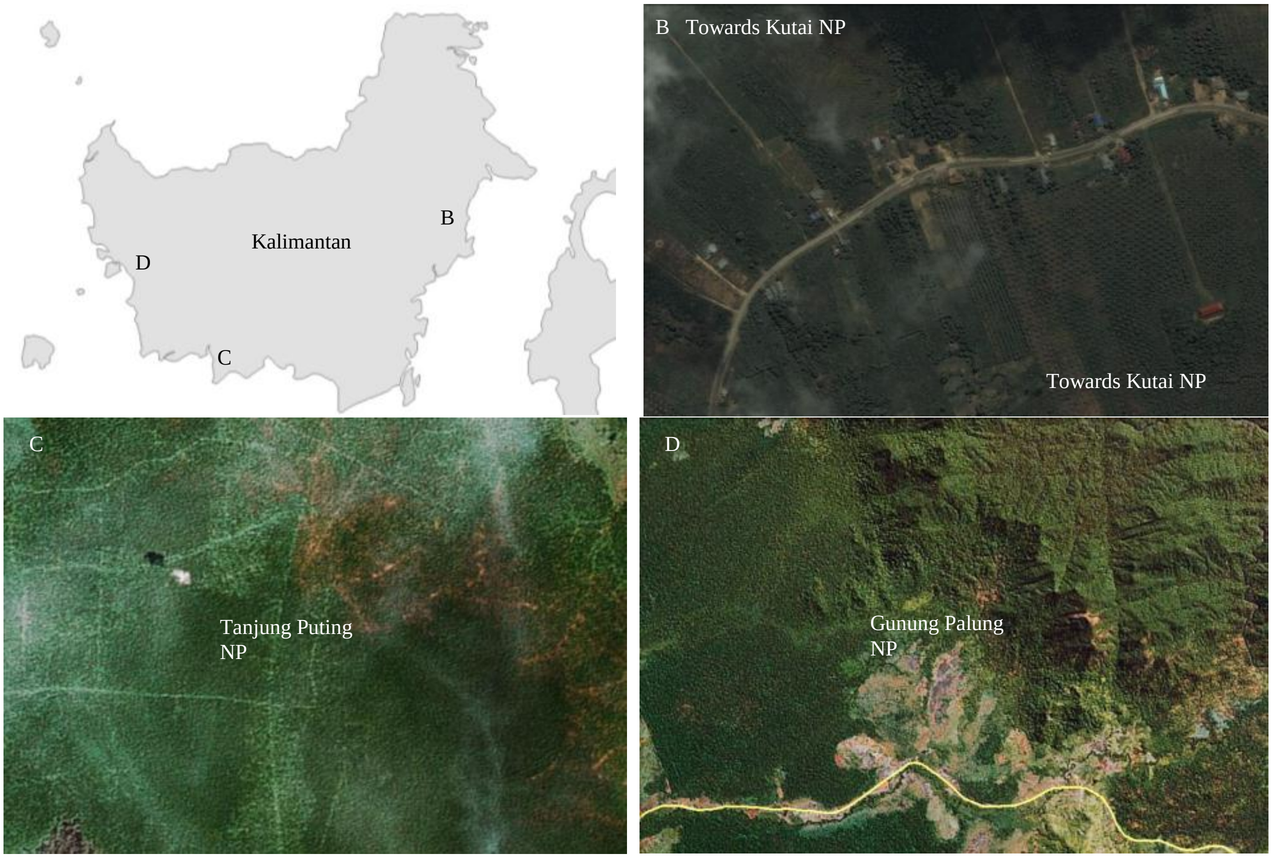


***Fig. 2**. A) Map of Kalimantan; B) Arterial roads and land use change around the Bontang-Sangata Road cutting through Kutai National Park (NP), C) Network of logging roads and land use change within Tanjung Puting NP; D) Land use change around an improved logging road into the southern portion of Gunung Palung NP. Roads = white/yellow lines, Forest cover = green areas, Land use change = brown areas. Images from Google Earth™ Mapping Services, 2010.*

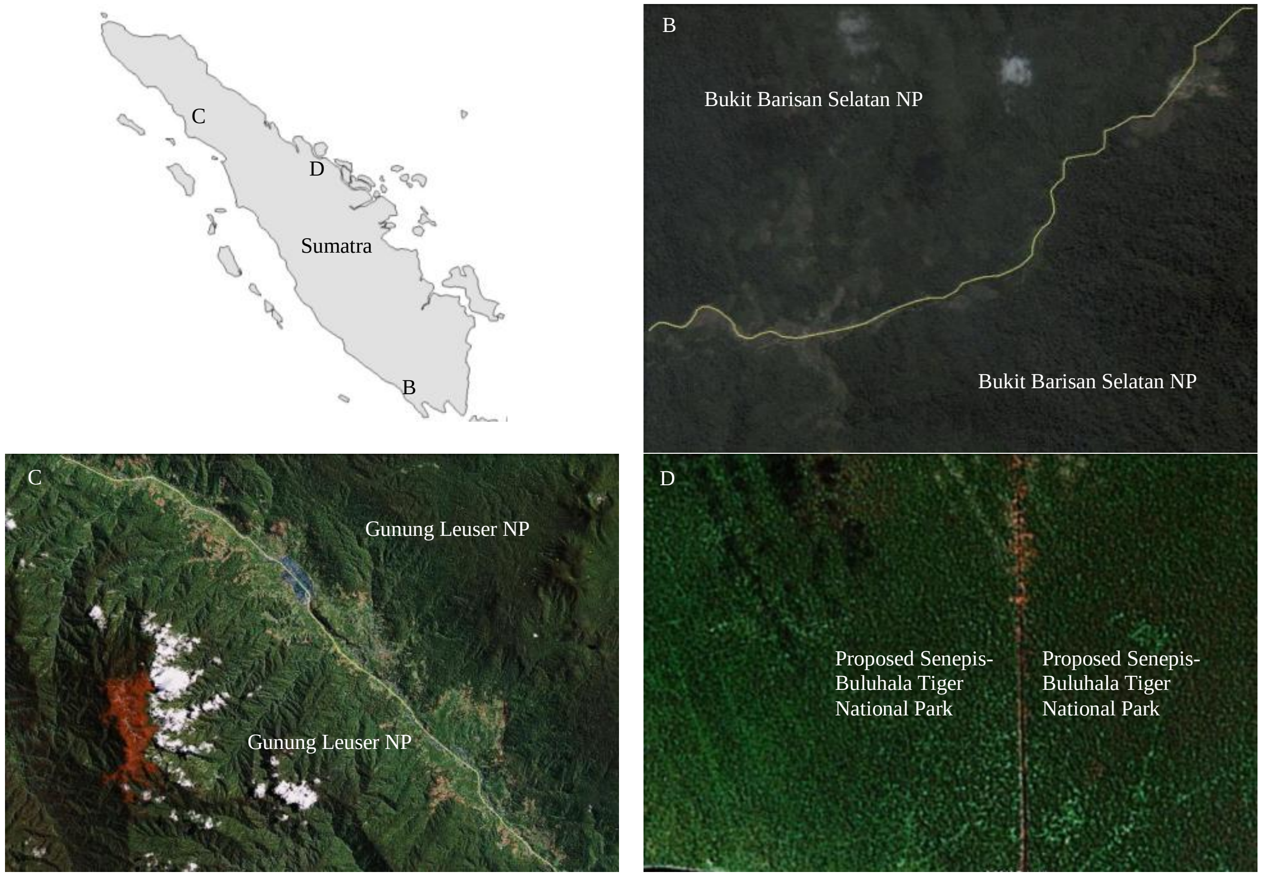


***Fig. 3**. A) Map of Sumatra; B) Land use change along Bengkunat-Sanggi Road cutting through important Sumatran Tiger, Asian Tapir and Asian Elephant habitats of Bukit Barisan Selatan National Park according to MaxEnt models (NP); C) Land use change along Blangkejeren-Kutacane Road cutting the Gunung Leuser National Park into two halves; D) Land use change along a legally questionable logging road cutting through the proposed Senepis-Buluhala Tiger National Park. Road = white/yellow lines, Forest cover = green areas, Land use change = brown areas. Images from Google Earth™ Mapping Services, 2010.*

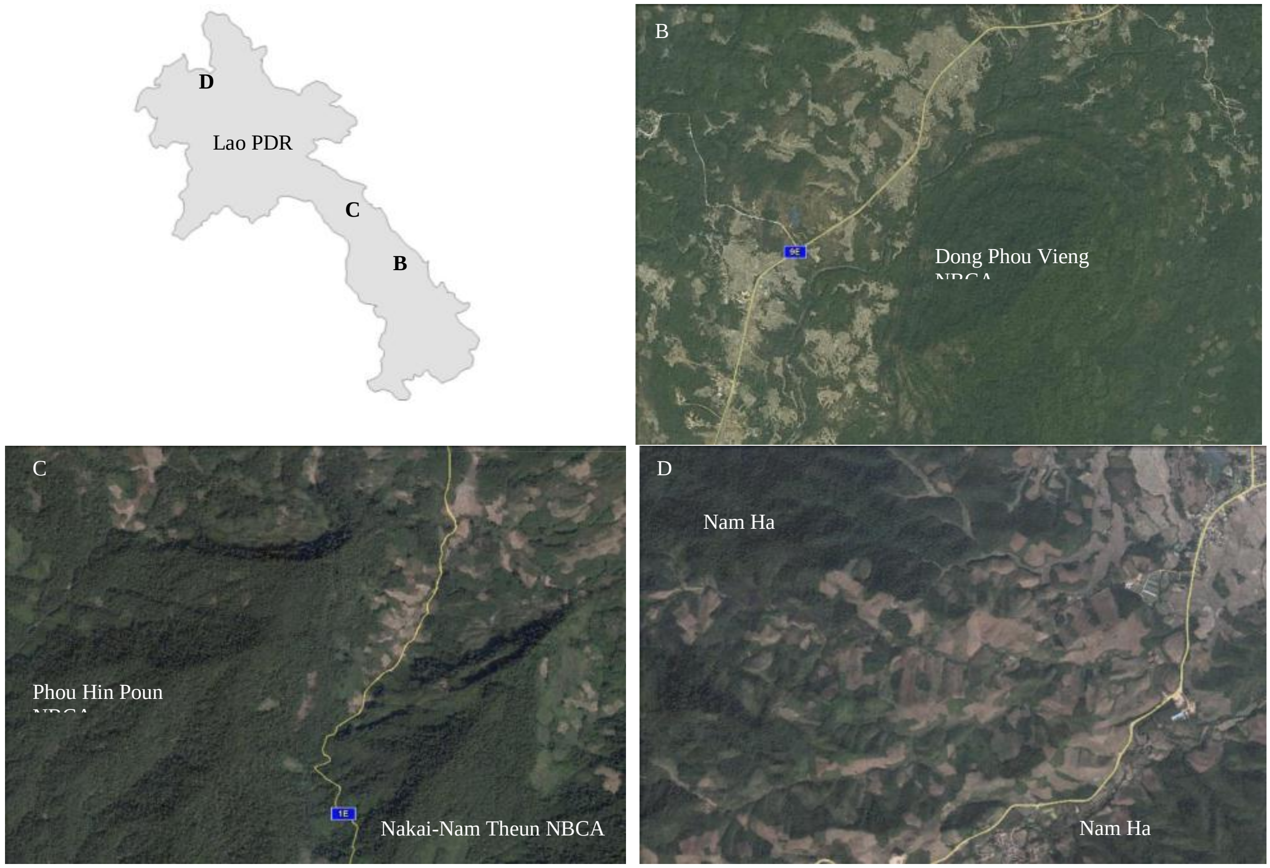


Fig. 4*. A) Map of Lao PDR; B) Route 9E encroaching on northwestern part of Dong Phou Vieng National Biodiversity Conservation Area (NBCA); C) Route 1E severing the corridor between Nakai-Nam Theun and Phou Hin Poun NBCAs; and D) Route Network 17A-3 bisecting Nam Ha National Biodiversity Conservation Area. Roads = yellow line, Forest cover = green areas, Land use change = brown areas. Images from Google Earth™ Mapping Services, 2010.*

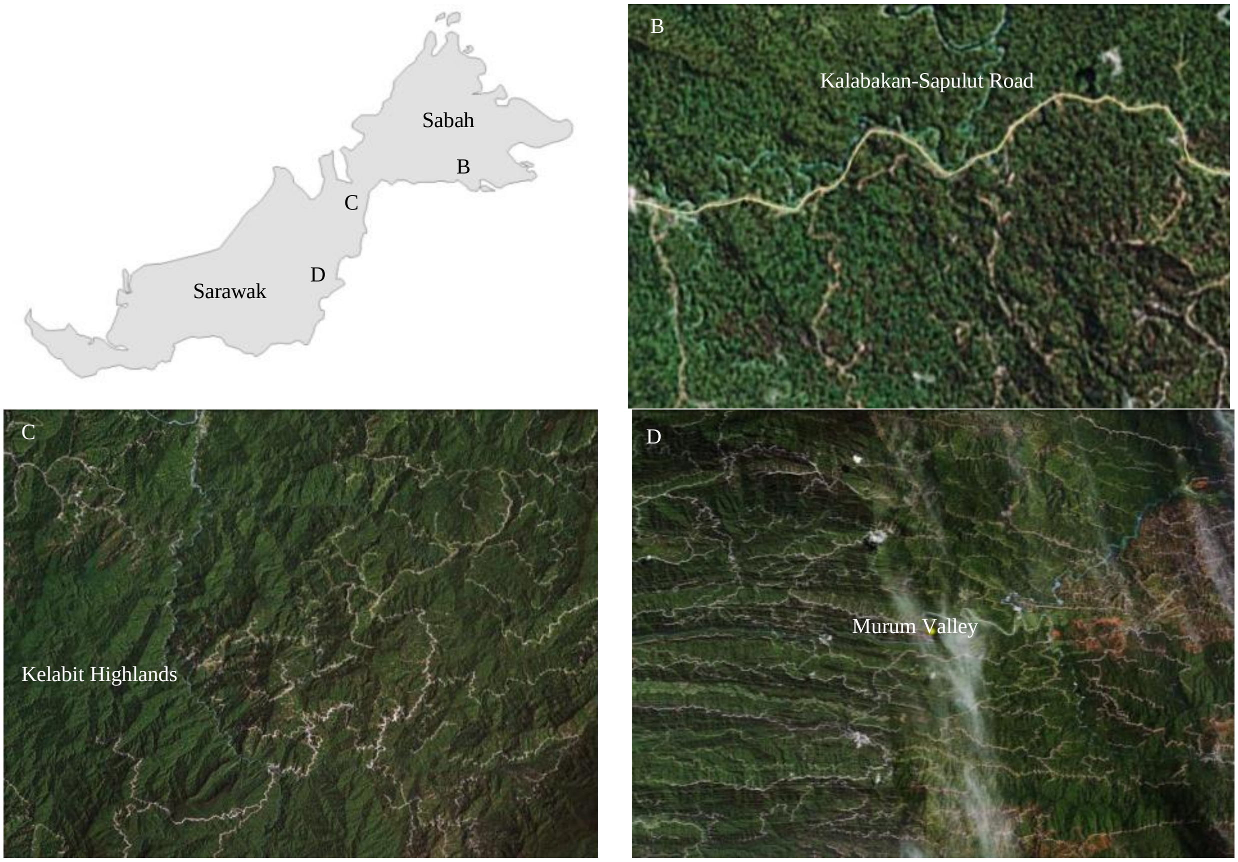


Fig. 5. *A) Map of East Malaysia; B) Logging roads originating from the Kalabakan-Sapulut Road through important elephant habitats towards the East Kalimantan border; C) Kelabit Highlands where another 300 km logging road threatens mammal habitats already fragmented by a sprawling logging road network; and D) Murum Valley where an access road to a new dam will augment the vast network of existing logging roads. Road = white/yellow lines, Forest cover = green areas, Land use change = brown areas. Images from Google Earth™ Mapping Services, 2010.*

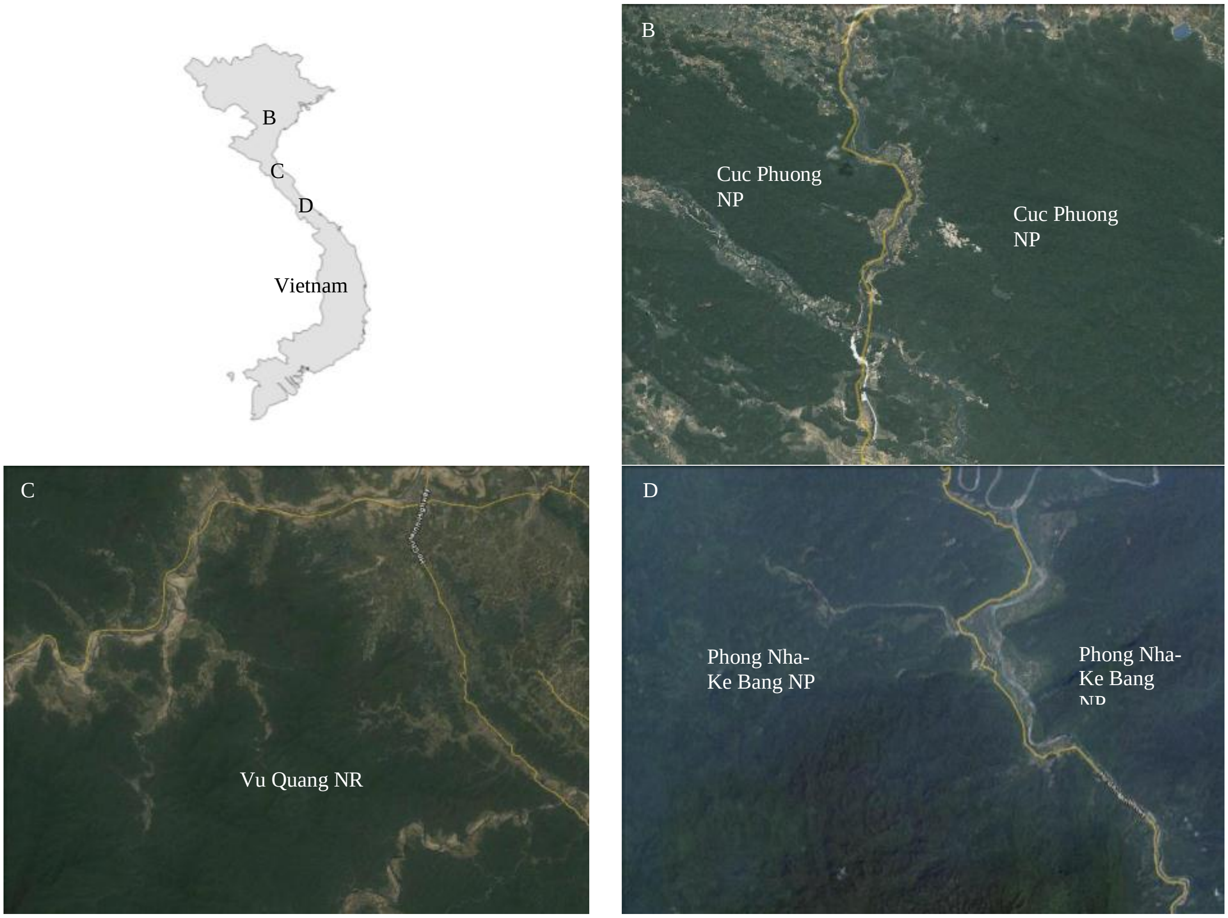


Fig. 6. *A) Map of Vietnam; B) Ho Chi Minh Highway cutting through the country's oldest national park (NP), Cuc Phuong; C) Ho Chi Minh Highway bisecting Phong Nha-Ke Bang NP; and D) arterial roads from the Ho Chi Minh Highway branching westwards past Vu Quang Nature Reserve (NR) into Lao PDR. Roads = yellow line, Forest cover = green areas, Land use change = brown areas. Images from Google Earth™ Mapping Services, 2010.*

Appendix 5: *Summary statistics for 95 mammal species with their associated lower/upper-bound population estimates (International Union for the Conservation of Nature 2013), IUCN threat categories (LC = Least Concern; NT = Near Tthreatened; VU = Vulnerable; EN = Endangered; CE = Critically Endangered; EX = Extinct), percentage range loss (Ceballos & Ehrlich 2002), and the SAFE index values. The SAFE index was calculated using the general formula: $log_{10}(N) - log_{10}(MVP_t)$, where N = lower-bound population estimate and MVP_t = threshold MVP target currently set at 5000 individuals according to Traill et al. (2007). Three other variants of the SAFE index are provided, to represent a range of uncertainty based on the lower and upper 95% confidence limits of mammal-specific MVP thresholds (2261 and 5095, respectively; Traill et al. [2007]): (1) SAFE (low) = log_{10}(lower population estimate bound) $- log_{10}$(5095); (2) SAFE (upp) = log_{10}(upper population estimate bound) $- log_{10}$(2261); and (3) SAFE (med) = median of SAFE (low) and SAFE (upp).*

Common name	**Scientific name**	**Lower pop estimate**	**Upper pop estimate**	**IUCN threat category**	**% Range loss**	**SAFE**	**SAFE (low)**	**SAFE (upp)**	**SAFE (med)**
Addax	*Addax nasomaculatus*	300	300	CE	94.8	–1.22	–1.23	–0.88	–1.06
African Wild Ass	*Equus africanus*	70	600	CE	97.5	–1.85	–1.86	–0.58	–1.22
African Wild Dog	*Lycaon pictus*	3000	5500	EN	84.0	–0.22	–0.23	0.39	0.08
Alice Springs Mouse	*Pseudomys fieldi*	2000	2000	VU	100.0	–0.40	–0.41	–0.05	–0.23
Alpine Ibex	*Capra ibex*	30000	30000	LC	77.7	0.78	0.77	1.12	0.95
Asian Elephant	*Elephas maximus*	41410	52345	EN	80.5	0.92	0.91	1.36	1.14
Baird's Tapir	*Tapirus bairdii*	5500	5500	EN	67.9	0.04	0.03	0.39	0.21
Banded Hare-Wallaby	*Lagostrophus fasciatus*	4300	6700	EN	98.9	–0.07	–0.07	0.47	0.20
Banteng	*Bos javanicus*	5000	8000	EN	87.1	0.00	–0.01	0.55	0.27
Barbary Macaque	*Macaca sylvanus*	15000	15000	EN	90.5	0.48	0.47	0.63	0.55
Beira	*Dorcatragus megalotis*	7000	7000	VU	22.9	0.15	0.14	0.49	0.32

Big-Eared Hopping-Mouse	*Notomys macrotis*	0	0	EX	100.0	–3.70	–3.71	3.35	–3.53
Bilby	*Macrotis lagotis*	10000	10000	VU	84.6	0.30	0.29	0.65	0.47
Bison	*Bison bison*	15000	30000	NT	99.1	0.48	0.47	1.12	0.80
Blackbuck	*Antilope cervicapra*	50000	50000	NT	61.9	1.00	0.99	1.34	1.17
Blue Buck	*Hippotragus leucophaeus*	0	0	EX	100.0	–3.70	–3.71	–3.35	–3.53
Bongo	*Tragelaphus eurycerus*	28000	28000	NT	34.9	0.75	0.74	1.09	0.92
Bridled Nailtail Wallaby	*Onychogalea fraenata*	1100	1100	EN	98.7	–0.66	–0.67	–0.31	–0.49
Broad-Faced Potoroo	*Potorous platyops*	0	0	EX	100.0	–3.70	–3.71	–3.35	–3.53
Brown Bear	*Ursus arctos*	200000	200000	LC	85.3	1.60	1.59	1.95	1.77
Brown Hyaena	*Hyaena brunnea*	5000	8000	NT	55.2	0.00	–0.01	0.55	0.27
Central Hare-Wallaby	*Lagorchestes asomatus*	0	0	EX	100.0	–3.70	–3.71	–3.35	–3.53
Cheetah	*Acinonyx jubatus*	7500	7500	VU	59.6	0.18	0.17	0.52	0.35
Common Hippopotamus	*Hippopotamus amphibius*	125000	148000	VU	82.8	1.40	1.39	1.82	1.61
Crescent Nailtail Wallaby	*Onychogalea lunata*	0	0	EX	100.0	–3.70	–3.71	–3.35	–3.53
Cuvier’s Gazelle	*Gazella cuvieri*	1750	2950	EN	99.3	–0.46	–0.46	0.12	–0.17
Darling Downs Hopping-Mouse	*Notomys mordax*	0	0	EX	100.0	–3.70	–3.71	–3.35	–3.53
Desert Bandicoot	*Perameles eremiana*	0	0	EX	100.0	–3.70	–3.71	–3.35	–3.53
Desert Rat-Kangaroo	*Caloprymnus campestris*	0	0	EX	100.0	–3.70	–3.71	–3.35	–3.53
Dibatag	*Ammodorcas clarkei*	1500	1500	VU	75.3	–0.52	–0.53	–0.18	–0.36
Dibbler	*Parantechinus apicalis*	500	1000	EN	33.5	–1.00	–1.01	–0.35	–0.68
Eastern Hare-Wallaby	*Lagorchestes leporides*	0	0	EX	100.0	–3.70	–3.71	–3.35	–3.53
Ethiopian Wolf	*Canis simensis*	239	239	EN	95.3	–1.32	–1.33	–0.98	–1.16
European Beaver	*Castor fiber*	639000	639000	LC	88.4	2.11	2.10	2.45	2.28
European Bison	*Bos bonasus*	1800	1800	VU	99.5	–0.44	–0.45	–0.10	–0.28
European Mink	*Mustela lutreola*	1500	2000	EN	54.2	–0.52	–0.53	–0.05	–0.29
Gaur	*Bos gaurus*	13000	30000	VU	89.1	0.41	0.41	1.12	0.77
Giraffe	*Giraffa camelopardalis*	80000	80000	LC	88.7	1.20	1.20	1.55	1.38
Golden Bandicoot	*Isoodon auratus*	22000	22000	VU	97.1	0.64	0.64	0.99	0.82
Golden Lion Tamarin	*Leontopithecus rosalia*	1000	1000	EN	99.0	–0.70	–0.71	–0.35	–0.53

Gould’s Mouse	*Pseudomys gouldii*	0	0	EX	100.0	–3.70	–3.71	–3.35	–3.53
Greater Stick-Nest Rat	*Leporillus conditor*	4000	4000	VU	99.3	–0.10	–0.11	0.25	0.07
Grevy’s Zebra	*Equus grevyi*	1996	2447	EN	91.8	–0.40	–0.41	0.03	–0.19
Guanaco	*Lama guanicoe*	535750	589750	LC	73.6	2.03	2.02	2.42	2.22
Hartebeest	*Alcelaphus buselaphus*	362000	362000	LC	69.7	1.86	1.85	2.20	2.03
Hastings River Mouse	*Pseudomys oralis*	10000	10000	VU	93.9	0.30	0.29	0.65	0.47
Iberian Lynx	*Lynx pardinus*	84	143	CE	97.2	–1.77	–1.78	–1.20	–1.49
Indian Rhinoceros	*Rhinoceros unicornis*	2575	2575	VU	95.3	–0.29	–0.30	0.06	–0.12
Javan Rhinoceros	*Rhinoceros sondaicus*	40	60	CE	95.9	–2.10	–2.11	–1.58	–1.85
Jentink’s Duiker	*Cephalophus jentinki*	2000	2000	EN	88.1	–0.40	–0.41	–0.05	–0.23
Kouprey	*Bos sauveli*	50	250	CE	84.6	–2.00	–2.01	–0.96	–1.49
Kowari	*Dasyuroides byrnei*	10000	10000	VU	64.3	0.30	0.29	0.65	0.47
Leadbeater’s Possum	*Gymnobelideus leadbeateri*	2000	2000	EN	74.5	–0.40	–0.41	–0.05	–0.23
Lechwe	*Kobus leche*	98000	98000	LC	82.0	1.29	1.28	1.64	1.46
Lesser Bilby	*Macrotis leucura*	0	0	EX	100.0	–3.70	–3.71	–3.35	–3.53
Lesser Stick-Nest Rat	*Leporillus apicalis*	0	0	CE	100.0	–3.70	–3.71	–3.35	–3.53
Lion	*Panthera leo*	16500	30000	VU	67.7	0.52	0.51	1.12	0.82
Long-Tailed Hopping-Mouse	*Notomys longicaudatus*	0	0	EX	100.0	–3.70	–3.71	–3.35	–3.53
Mountain Nyala	*Tragelaphus buxtoni*	1500	4000	EN	44.0	–0.52	–0.53	0.25	–0.14
Northern Hairy-Nosed Wombat	*Lasiorhinus krefftii*	115	115	CE	95.9	–1.64	–1.65	–1.29	–1.47
Numbat	*Myrmecobius fasciatus*	1000	1000	EN	97.3	–0.70	–0.71	–0.35	–0.53
Okapi	*Okapia johnstoni*	35000	50000	NT	68.4	0.85	0.84	1.34	1.09
Pampas Deer	*Ozotoceros bezoarticus*	20000	80000	NT	22.9	0.60	0.59	1.55	1.07
Philippine Spotted Deer	*Cervus alfredi*	2500	2500	EN	49.5	–0.30	–0.31	0.04	–0.14
Pig-Footed Bandicoot	*Chaeropus ecaudatus*	0	0	EX	100.0	–3.70	–3.71	–3.35	–3.53
Pronghorn Antelope	*Antilocapra americana*	700000	700000	LC	17.9	2.15	2.14	2.49	2.32
Puku	*Kobus vardonii*	130000	130000	NT	86.1	1.41	1.41	1.76	1.59
Pygmy Hippopotamus	*Hexaprotodon liberiensis*	2000	3000	EN	98.7	–0.40	–0.41	0.12	–0.15
Red-Fronted Gazelle	*Gazella rufifrons*	25000	25000	VU	54.2	0.70	0.69	1.04	0.87

Red-Tailed Phascogale	*Phascogale calura*	10000	10000	NT	99.1	0.30	0.29	0.65	0.47
Riverine Rabbit	*Bunolagus monticularis*	500	500	CE	57.5	–1.00	–1.01	–0.66	–0.84
Roan Antelope	*Hippotragus equinus*	40000	76000	LC	34.4	0.90	0.89	1.53	1.21
Rufous Hare-Wallaby	*Lagorchestes hirsutus*	0	0	VU	99.3	–3.70	–3.71	–3.35	–3.53
Sable Antelope	*Hippotragus niger*	54000	75000	LC	50.9	1.03	1.03	1.52	1.28
Scimitar-Horned Oryx	*Oryx dammah*	0	0	EX	97.1	–3.70	–3.71	–3.35	–3.53
Shark Bay Mouse	*Pseudomys praeconis*	2000	2000	VU	88.4	–0.40	–0.41	–0.05	–0.23
Short-Tailed Hopping-Mouse	*Notomys amplus*	0	0	EX	100.0	–3.70	–3.71	–3.35	–3.53
Smoky Mouse	*Pseudomys fumeus*	2500	2500	VU	90.7	–0.30	–0.31	0.04	–0.14
Soemmerring’s Gazelle	*Gazella soemmerringii*	6000	6500	VU	94.3	0.08	0.07	0.46	0.27
Spotted Hyaena	*Crocuta crocuta*	27000	47000	LC	13.6	0.73	0.72	1.32	1.02
Spotted-Tailed Quoll	*Dasyurus maculatus*	20000	20000	NT	15.6	0.60	0.59	0.95	0.77
Springbuck	*Antidorcas marsupialis*	2000000	2500000	LC	52.8	2.60	2.59	3.04	2.82
Sumatran Rhinoceros	*Dicerorhinus sumatrensis*	220	275	CE	92.0	–1.36	–1.36	–0.91	–1.14
Thylancine	*Thylacinus cynocephalus*	0	0	EX	100.0	–3.70	–3.71	–3.35	–3.53
Tiger	*Panthera tigris*	3062	5066	EN	87.5	–0.21	–0.22	0.35	0.07
Toolache Wallaby	*Macropus greyi*	0	0	EX	100.0	–3.70	–3.71	–3.35	–3.53
Tsessebe	*Damaliscus lunatus*	300000	400000	LC	62.4	1.78	1.77	2.25	2.01
Vicuña	*Vicugna vicugna*	347273	347273	LC	83.6	1.84	1.83	2.19	2.01
Western Barred Bandicoot	*Perameles bougainville*	10000	10000	EN	100.0	0.30	0.29	0.65	0.47
Western Quoll	*Dasyurus geoffroii*	10000	10000	NT	98.5	0.30	0.29	0.65	0.47
White Rhino	*Ceratotherium simum*	17480	17480	NT	97.0	0.54	0.54	0.89	0.72
White-Footed Rabbit-Rat	*Conilurus albipes*	0	0	EX	100.0	–3.70	–3.71	–3.35	–3.53
Woolly Spider Monkey	*Brachyteles arachnoides*	1300	1300	EN	89.6	–0.59	–0.59	–0.24	–0.42
Yellow-Footed Rock-Wallaby	*Petrogale xanthopus*	10000	10000	NT	24.2	0.30	0.29	0.65	0.47
Zebra Duiker	*Cephalophus zebra*	10000	15000	VU	59.9	0.30	0.29	0.82	0.56

Appendix 6: *Generalised linear model (GLM) and generalised linear mixed-effect model (GLMM) sets used to examine the relationship between the probability (Pr) of being threatened for 95 mammal species and predictors.*

Model	***k***	***–LL***	**ΔAIC$_c$**	***w*AIC$_c$**	**%DE**
GLM					
Pr(*threat*) ~ *SAFE (low)*	2	–22.58	0.00	1.00	59.5
Pr(*threat*) ~ *% range loss*	2	–46.37	47.57	0.00	16.8
Pr(*threat*) ~ *1*	1	–55.75	64.26	0.00	0.00
Pr(*threat*) ~ *SAFE (upp)*	2	–22.24	0.00	1.00	60.1
Pr(*threat*) ~ *% range loss*	2	–46.37	48.24	0.00	16.8
Pr(*threat*) ~ *1*	1	–55.75	64.93	0.00	0.00
Pr(*threat*) ~ *SAFE (med)*	2	–22.20	0.00	1.00	60.2
Pr(*threat*) ~ *% range loss*	2	–46.37	48.33	0.00	16.8
Pr(*threat*) ~ *1*	1	–55.75	65.02	0.00	0.00
GLMM					
Pr(*threat*) ~ *SAFE (low)* + (1|*ORDER*)	3	–20.96	0.00	1.00	59.7
Pr(*threat*) ~ *% range loss* + (1|*ORDER*)	3	–45.16	48.41	0.00	13.1
Pr(*threat*) ~ *1* + (1|*ORDER*)	2	–51.99	59.93	0.00	0.00
Pr(*threat*) ~ *SAFE (upp)* + (1|*ORDER*)	3	–20.65	0.00	1.00	60.3

Pr(*threat*) ~ *% range loss* + (1|*ORDER*)	3	–45.16	49.03	0.00	13.1
Pr(*threat*) ~ *1* + (1|*ORDER*)	2	–51.99	60.55	0.00	0.00
Pr(*threat*) ~ *SAFE (med)* + (1|*ORDER*)	3	–20.56	0.00	1.00	60.5
Pr(*threat*) ~ *% range loss* + (1|*ORDER*)	3	–45.16	49.20	0.00	13.1
Pr(*threat*) ~ *1* + (1|*ORDER*)	2	–51.99	60.73	0.00	0.00

Notes: *Only single-term models were considered to test the relative ability of three uncertainty variants of the SAFE index versus percentage range loss to predict extinction threat. See Appendix 4 for definitions of SAFE (low), SAFE (upp), and SAFE (med).The analytical theme represented by each model (SAFE, % range loss, the intercept-only model, and ORDER as a random effect), and the information-theoretic ranking of models investigating the predictors of mammal threat status according to Akaike's Information Criterion corrected for small sample size (*AIC_c*) are shown. k = number of parameters, –LL = maximum log-likelihood,* ΔAIC_c *= difference in* AIC_c *for each model from the most parsimonious model,* $wAIC_c$ = AIC_c *weight, and %DE = percent deviance explained in the response variable by the model under consideration. Two data points were removed for the GLMMs because there was only one representative species in its respective Order:* Bunolagus monticularis *and* Elephas maximus.

Appendix 7: *Summary statstics for 25 mammal species and their conservative population estimates compiled from lower-bound figures in IUCN Red List assessments (International Union for the Conservation of Nature 2013), IUCN threat categories (EN = Endangered; CR = Critically Endangered), and SAFE index values. The SAFE index was calculated using the general formula:* $log_{10}(N) - log_{10}(MVP_t)$*, where N = lower-bound population estimate and* MVP_t *= threshold MVP target currently set at 5000 individuals according to Traill et al. (2007).*

No.	Common name	Scientific name	Population estimate	IUCN threat category	SAFE Index
1	Agile Gibbon	*Hylobates agilis*	5,479	EN	0.04
2	Asian elephant	*Elephas maximus*	41,410	EN	0.92
3	Asian Tapir	*Tapirus indicus*	1,550	EN	-0.51
4	Banteng	*Bos javanicus*	5,000	EN	0.00
5	Black-crested Gibbon	*Nomascus concolor*	1,300	CR	-0.59
6	Bornean Gibbon	*Hylobates muelleri*	250,000	EN	1.70
7	Bornean Orangutan	*Pongo pygmaeus*	45,000	EN	0.95
8	Delacour's Langur	*Trachypithecus delacouri*	200	CR	-1.40
9	Dhole	*Cuon alpinus*	2,500	EN	-0.30
10	Douc Langur	*Pygathrix nigripes*	42,609	EN	0.93
11	Eld's Deer	*Rucervus eldii*	1,000	EN	-0.70
12	Giant Muntjac	*Muntiacus vuquangensis*	NA	EN	NA
13	Javan rhinoceros	*Rhinoceros sondiacus*	50	CR	-2.00
14	Otter Civet	*Cynogale bennettii*	NA	EN	NA
15	Northern White-cheeked Gibbon	*Nomascus leucogenys*	NA	CR	NA
16	Pileated Gibbon	*Hylobates pileatus*	33,000	EN	0.82
17	Yellow-cheeked Crested Gibbon	*Nomascus gabriellae*	10,526	EN	0.32
18	Red-shanked Douc Langur	*Pygathrix nemaeus*	171	EN	-1.47
19	Saola	*Pseudoryx nghetinhensis*	750	CR	-0.82
20	Siamang	*Symphalangus syndactylus*	22,390	EN	0.65

21	Sumatran Orangutan	*Pongo abelii*	7,300	EN	0.16
22	Sumatran rhinoceros	*Dicerorhinus sumatrensis*	220	CR	-1.36
23	Sunda Pangolin	*Manis javanicus*	NA	EN	NA
24	Tiger	*Panthera tigris*	3,402	EN	-0.17
25	White-handed Gibbon	*Hylobates lar*	14,000	EN	0.45

Appendix 8: *Questionnaire used to interview Orang Asli with possible (answers) and [notes] from 10 villages in the Belum-Temengor Forest Complex, Perak, Peninsular Malaysia.*

Information group 1 (Demography)

1. What is your ethnic group?
2. What is your village name? [inferred from map]
3. How old are you?
4. Were you born here? (Yes/no)
5. Education level (None/Primary/Secondary/Diploma/University)
6. Does your village have direct access to the highway? [inferred from map]
7. What do you work as?

Information group 2 (Level of support for presence of highway and additional roads to village)

8. Do you support the presence of the Gerik-Jeli highway? (Not supportive/Not sure/Supportive)
9. Do you feel more roads should be built towards your village? (Not supportive/Not sure/Supportive)

Information group 3 (Use of highway for non-hunting livelihood activities)

10. Do you use the highway to get to work? (Yes/no)
11. Do you use the highway to get to market? (Yes/no)
12. Do you use the highway to sell your hunted animals? (Yes/no)

Information group 4 (Perceived negative impacts of highway on livelihoods)

13. Do you feel that the highway brings pollution into your village? (No, Yes, Not Sure)

14. Do you feel that the highway brings diseases to your village? (No, Yes, Not Sure)
15. Do you feel that the highway brings poachers into the forests near your village? (No, Yes, Not Sure)

Information group 5 (Perceived positive impacts of highway on livelihoods)

16. Do you feel that the highway brings health workers into your village? (No, Yes, Not Sure)
17. Do you want that? (No, Yes, Not Sure)
18. Do you feel that the highway brings in donations into your village? (No, Yes, Not Sure)
19. Do you want that? (No, Yes, Not Sure)
20. Do you feel that the highway brings job opportunities into your village? (No, Yes, Not Sure)
21. Do you want that? (No, Yes, Not Sure)

Information group 6 (Perceived state of mammals near highway)

22. What do you think is the level of threat to mammals in forests along the highway compared to forests far from the highway? (Lower/No difference/Higher/Not sure)
23. How abundant are mammals in forests along the highway compared to forests far from the highway? (Lower/No difference/Higher/Not sure). If lower, what animals have been affected?
24. Why are there less mammals in the forests along the highway?
25. Because of hunting from your village (No, Yes, Not Sure)
26. Because of hunting from other indigenous villages (No, Yes, Not Sure)
27. Because of hunting from the locals (No, Yes, Not Sure)
28. Because of hunting from foreigners (No, Yes, Not Sure)

29. Because of roadkills (No, Yes, Not Sure)

30. Because of illegal logging (No, Yes, Not Sure)

31. Because of infrastructure developments (No, Yes, Not Sure)

Information group 7 (Use of forests along highway for hunting) [Interviewee was told that answers were confidential and for research purposes only)

32. Do you hunt in forests along the highway? (No, Yes)

33. If you do hunt, what animals do you hunt in the forests along the highway?

Appendix 9: *Example of an underpass in the eastern linkage, Terengganu, Peninsular Malaysia. Red line indicates a space between underpass columns where mammals can potential use as a crossing point. Blue line indicates a water body where no camera traps were installed.*

Appendix 10*: Global Positioning System (GPS) coordinates of 10 road kills events of six mammal species detected from ad hoc drives along the road bisecting the eastern linkage, Terengganu, Peninsular Malaysia. IUCN status: NT = Near Threatened; LC = Least Concern.*

Common Name (IUCN Status)	**Species**	***x***	***y***
Leopard (NT)	*Panthera pardus*	102.535	5.018
Leopard Cat (LC)	*Prionailurus bengalensis*	102.611	5.082
		102.613	5.117
		102.622	5.163
		102.562	5.032
Large Indian Civet (NT)	*Viverra zibetha*	102.711	5.186
Small-toothed Palm Civet (LC)	*Arctogalidia trivirgata*	102.797	5.178
Wild Pig (LC)	*Sus scrofa*	102.805	5.149
		102.613	5.108
Long-tailed Macaque (LC)	*Macaca fascicularis*	102.744	5.189

Appendix 11: *Example of trails leading to the road (yellow line) and trails leading to the underpass (red line) in the eastern linkage, Terengganu, Peninsular Malaysia. Grass verges (yellow line) prevented trails on one side of the road from directly connecting to an opposable trail – this did not allow cameras to monitor whether animals on trails were crossing to the other side.*

Appendix 12*: List of 37 non-human native mammal species detected over 10,502 camera-trap nights in the forests of the eastern linkage, Terengganu, Peninsular Malaysia. Surveys were conducted between 7 Apr 2011 and16 Mar 2012. N = independent detections (0.5 hr intervals); PCRI = Photographic Capture Rate Index (N/1000 trap nights); Stations = no. of camera trap stations that detected the species. IUCN status: EN = Endangered; VU = Vulnerable; NT = Near Threatened; LC = Least Concern. Note: The White-handed Gibbon* Hylobates lar *(EN) and Smooth-coated Otter* Lutrogale perspicillata *(VU) were photographed by a hand-held camera.*

Common Name (IUCN Status)	**Species**	**N**	**PCRI**	**Stations**
Asian Elephant (EN)	*Elephas maximus*	144	13.71	60
Asian Tapir (EN)	*Tapirus indicus*	282	26.85	100
Bamboo Rat (LC)	*Rhizomys sumatrensis*	1	0.10	1
Banded Linsang (LC)	*Prionodon linsang*	9	0.86	9
Banded Palm Civet (VU)	*Hemigalus derbyanus*	2	0.19	2
Barking Deer (LC)	*Muntiacus muntjak*	1334	127.02	152
Binturong (VU)	*Arctictis binturong*	3	0.29	3
Brush-tailed Porcupine (LC)	*Atherurus macrourus*	32	3.05	11
Clouded Leopard (VU)	*Neofelis nebulosa*	74	7.05	42
Common Palm Civet (LC)	*Paradoxurus hermaphroditus*	4	0.38	4
Crab-eating Mongoose (LC)	*Herpestes urva*	5	0.48	2
Dhole (EN)	*Cuon alpinus*	5	0.48	5
Dusky Leaf Monkey (NT)	*Trachypithecus obscurus*	1	0.10	1
Golden Cat (NT)	*Pardofelis temminckii*	137	13.05	70
Grey-bellied Squirrel (LC)	*Callosciurus caniceps*	1	0.10	1
Large Indian Civet (NT)	*Viverra zibetha*	4	0.38	4
Leopard (NT)	*Panthera pardus*	32	3.05	27
Leopard Cat (LC)	*Prionailurus bengalensis*	14	1.33	10
Long-tailed Macaque (LC)	*Macaca fascicularis*	21	2.00	4
Malay Civet (LC)	*Viverra tangalunga*	2	0.19	2
Malayan Porcupine (LC)	*Hystrix brachyura*	152	14.47	46
Malayan Tiger (EN)	*Panthera tigris jacksoni*	25	2.38	19
Marbled Cat (VU)	*Pardofelis marmorata*	11	1.05	10
Masked Palm Civet (LC)	*Paguma larvata*	5	0.48	4
Moonrat (LC)	*Echinosorex gymnura*	50	4.76	8
Mousedeer (LC)	*Tragulus* spp.	156	14.85	46
Pig-tailed Macaque (VU)	*Macaca nemestrina*	116	11.05	59
Prevost's Squirrel (LC)	*Callosciurus prevostii*	16	1.52	1
Rats	*Rattus* spp.	113	10.76	39

Sambar Deer (VU)	*Rusa unicolor*	2	0.19	2
Serow (VU)	*Capricornis sumatraensis*	12	1.14	11
Sun Bear (VU)	*Helarctos malayanus*	140	13.33	62
Sunda Pangolin (EN)	*Manis javanica*	3	0.29	3
Three-striped Ground Squirrel (LC)	*Lariscus insignis*	11	1.05	8
White-thighed Leaf Monkey (NT)	*Presbytis siamensis*	5	0.48	5
Wild Pig (LC)	*Sus scrofa*	472	44.94	128
Yellow-throated Marten (LC)	*Martes flavigula*	7	0.67	6

Appendix 13*: List of 35 non-human native mammal species detected over 12,063 camera-trap nights in the forests of the western linkage, Perak, Peninsular Malaysia. Surveys were conducted between 12 May 2012 and 17 Feb 2013. N = independent detections (0.5 hr intervals); PCRI = Photographic Capture Rate Index (N/1000 trap nights); Stations = no. of camera trap stations out of 199 camera trap stations that detected the species. IUCN status: EN = Endangered; VU = Vulnerable; NT = Near Threatened; LC = Least Concern.*

Common Name (IUCN Status)	Species	N	PCRI	Stations
Asian Elephant (EN)	*Elephas maximus*	39	3.23	24
Asian Tapir (EN)	*Tapirus indicus*	125	10.36	59
Bamboo Rat (LC)	*Rhizomys sumatrensis*	5	0.41	5
Banded Leaf Monkey (NT)	*Presbytis femoralis*	3	0.25	2
Banded Linsang (LC)	*Prionodon linsang*	2	0.17	2
Banded Palm Civet (VU)	*Hemigalus derbyanus*	5	0.41	5
Barking Deer (LC)	*Muntiacus muntjak*	884	73.28	163
Binturong (VU)	*Arctictis binturong*	19	1.58	18
Brush-tailed Porcupine (LC)	*Atherurus macrourus*	32	2.65	11
Clouded Leopard (VU)	*Neofelis nebulosa*	73	6.05	54
Common Palm Civet (LC)	*Paradoxurus hermaphroditus*	6	0.5	5
Crab-eating Mongoose (LC)	*Herpestes urva*	1	0.08	1
Cream-coloured Giant Squirrel (NT)	*Ratufa affinis*	1	0.08	1
Dusky Leaf Monkey (NT)	*Trachypithecus obscurus*	12	0.99	11
Golden Cat (NT)	*Pardofelis temminckii*	106	8.79	58
Large Indian Civet (NT)	*Viverra zibetha*	2	0.17	2
Leopard Cat (LC)	*Prionailurus bengalensis*	80	6.63	43
Long-tailed Macaque (LC)	*Macaca fascicularis*	3	0.25	3
Malayan Porcupine (LC)	*Hystrix brachyura*	157	13.02	39
Malayan Tiger (EN)	*Panthera tigris jacksoni*	2	0.17	2
Marbled Cat (VU)	*Pardofelis marmorata*	13	1.08	7
Masked Palm Civet (LC)	*Paguma larvata*	9	0.75	7
Moonrat (LC)	*Echinosorex gymnura*	3	0.25	2
Mousedeer (LC)	*Tragulus* spp.	249	20.64	60
Pig-tailed Macaque (VU)	*Macaca nemestrina*	388	32.16	136
Prevost's Squirrel (LC)	*Callosciurus prevostii*	1	0.08	1
Rats	*Rattus* spp.	253	20.97	84
Sambar Deer (VU)	*Rusa unicolor*	20	1.66	7
Serow (VU)	*Capricornis sumatraensis*	70	5.8	35
Sun Bear (VU)	*Helarctos malayanus*	130	10.78	57
Sunda Pangolin (EN)	*Manis javanica*	4	0.33	4

Three-striped Ground Squirrel (LC)	*Lariscus insignis*	17	1.41	6
White-handed Gibbon (EN)	*Hylobates lar*	1	0.08	1
Wild Pig (LC)	*Sus scrofa*	2079	172.35	184
Yellow-throated Marten (LC)	*Martes flavigula*	40	3.32	24

Appendix 14: *Photographic evidence of 17 IUCN-threatened mammal species (EN-VU) detected in the eastern and western linkages, Peninsular Malaysia.*

1. Dhole *Cuon alpinus* (EN) **–** Eastern linkage

2. Asian Elephant *Elephas maximus* (EN) – Eastern linkage

3. Sunda Pangolin *Manis javanica* (EN) – Eastern linkage

4. Malayan Tiger *Panthera tigris jacksoni* (EN) – Eastern linkage

5. Asian Tapir *Tapirus indicus* (EN) – Eastern linkage

6. Binturong *Arctictis binturong* (VU) – Western linkage

7. Gaur *Bos gaurus* (VU) – Eastern linkage

8. Serow *Capricornis sumatraensis* (VU) – Western linkage

9. Sun Bear *Helarctos malayanus* (VU) – Eastern linkage

10. Banded Palm Civet *Hemigalus derbyanus* (VU) – Eastern linkage

11. Pig-tailed Macaque *Macaca nemestrina* (VU) – Eastern linkage

12. Clouded Leopard *Neofelis nebulosa* (VU) with *Arctictis binturong* prey – Eastern linkage

13. Marbled Cat *Pardofelis marmorata* (VU) – Eastern linkage

14. Sambar *Rusa unicolor* (VU) – Eastern linkage

15. Asian Small-clawed Otter *Aonyx cinerea* (VU) **–** Eastern linkage

16. White-handed Gibbon *Hylobates lar* (EN) – Eastern linkage

17. Smooth-coated Otter *Lutrogale perspicillata* (VU) **–** Eastern linkage

Appendix 15*: List of 18 non-human native mammal species detected over 11,278 camera-trap nights beneath 10 underpasses in the eastern linkage, Terengganu, Peninsular Malaysia. Surveys were conducted between 23 May 2011 and 17 Mar 2012. N = independent detections (0.5 hr intervals); PCRI = Photographic Capture Rate Index (N/1000 trap nights); Stations = no. of camera trap stations that detected the species. IUCN status: EN = Endangered; VU = Vulnerable; NT = Near Threatened; LC = Least Concern.*

Common Name (IUCN Status)	**Species**	**N**	**PCRI**	**Stations**
Asian Elephant (EN)	*Elephas maximus*	100	8.87	24
Asian Tapir (EN)	*Tapirus indicus*	186	16.49	23
Bamboo Rat (LC)	*Rhizomys sumatrensis*	1	0.09	1
Banded Palm Civet (VU)	*Hemigalus derbyanus*	1	0.09	1
Barking Deer (LC)	*Muntiacus muntjak*	680	60.29	30
Clouded Leopard (VU)	*Neofelis nebulosa*	1	0.09	1
Golden Cat (NT)	*Pardofelis temminckii*	4	0.35	3
Large Indian Civet (NT)	*Viverra zibetha*	28	2.48	11
Leopard (NT)	*Panthera pardus*	1	0.09	1
Leopard Cat (LC)	*Prionailurus bengalensis*	85	7.54	23
Long-tailed Macaque (LC)	*Macaca fascicularis*	525	46.55	39
Malay Civet (LC)	*Viverra tangalunga*	1	0.09	1
Malayan Porcupine (LC)	*Hystrix brachyura*	1	0.09	1
Masked Palm Civet (LC)	*Paguma larvata*	1	0.09	1
Pig-tailed Macaque (VU)	*Macaca nemestrina*	7	0.62	3
Rats	*Rattus* spp.	2	0.18	2
Sun Bear (VU)	*Helarctos malayanus*	10	0.89	8
Wild Pig (LC)	*Sus scrofa*	191	16.94	32

***Appendix 16**: List of 18 non-human native mammal species detected over 13,841 camera-trap nights beneath 10 underpasses in the western linkage, Perak, Peninsular Malaysia. Surveys were conducted between 18 Jun 2012 and 28 Jan 2013. N = independent detections (0.5 hr intervals); PCRI = Photographic Capture Rate Index (N/1000 trap nights); Stations = no. of camera trap stations that detected the species. IUCN status: EN = Endangered; VU = Vulnerable; NT = Near Threatened; LC = Least Concern.*

Common Name (IUCN Status)	**Species**	**N**	**PCRI**	**Stations**
Asian Elephant (EN)	*Elephas maximus*	27	1.95	12
Bamboo Rat (LC)	*Rhizomys sumatrensis*	25	1.81	18
Banded Leaf monkey (NT)	*Presbytis femoralis*	7	0.51	3
Barking Deer (LC)	*Muntiacus muntjak*	60	4.33	8
Binturong (VU)	*Arctictis binturong*	2	0.14	2
Black Giant Squirrel (NT)	*Ratufa bicolor*	7	0.51	3
Clouded Leopard (VU)	*Neofelis nebulosa*	5	0.36	3
Crab-eating Mongoose (LC)	*Herpestes urva*	1	0.07	1
Leopard Cat (LC)	*Prionailurus bengalensis*	122	8.81	36
Long-tailed Macaque (LC)	*Macaca fascicularis*	166	11.99	36
Malayan Porcupine (LC)	*Hystrix brachyura*	14	1.01	4
Masked Palm Civet (LC)	*Paguma larvata*	11	0.79	8
Mousedeer (LC)	*Tragulus spp.*	1	0.07	1
Pig-tailed Macaque (VU)	*Macaca nemestrina*	14	1.01	9
Rats	*Rattus spp.*	10	0.72	7
Serow (VU)	*Capricornis sumatraensis*	71	5.13	16
Sunda Pangolin (EN)	*Manis javanica*	4	0.29	4
Wild Pig (LC)	*Sus scrofa*	2089	150.93	46

Appendix 17: *List of 34 non-human native mammal species detected over 16,066 camera-trap nights on trails leading to the underpass or the road in the eastern linkage, Terengganu, Peninsular Malaysia. Surveys were conducted between 20 Apr 2012 and 17 Mar 2013. N = independent detections (0.5 hr intervals); PCRI = Photographic Capture Rate Index (N/1000 trap nights); Stations = no. of camera trap stations that detected the species. IUCN status: EN = Endangered; VU = Vulnerable; NT = Near Threatened; LC = Least Concern.*

Common Name (IUCN Status)	**Species**	**N**	**PCRI**	**Stations**
Asian Elephant (EN)	*Elephas maximus*	148	9.21	28
Asian Tapir (EN)	*Tapirus indicus*	394	24.52	44
Bamboo Rat (LC)	*Rhizomys sumatrensis*	3	0.19	3
Banded Linsang (LC)	*Prionodon linsang*	2	0.12	2
Banded Palm Civet (VU)	*Hemigalus derbyanus*	1	0.06	1
Barking Deer (LC)	*Muntiacus muntjak*	907	56.45	48
Binturong (VU)	*Arctictis binturong*	2	0.12	2
Brush-tailed Porcupine (LC)	*Atherurus macrourus*	11	0.68	1
Clouded Leopard (VU)	*Neofelis nebulosa*	7	0.44	5
Common Palm Civet (LC)	*Paradoxurus hermaphroditus*	20	1.24	11
Dhole (EN)	*Cuon alpinus*	1	0.06	1
Dusky Leaf Monkey (NT)	*Trachypithecus obscurus*	5	0.31	1
Gaur (VU)	*Bos gaurus*	3	0.19	3
Golden Cat (NT)	*Pardofelis temminckii*	40	2.49	20
Large Indian Civet (NT)	*Viverra zibetha*	18	1.12	13
Leopard (NT)	*Panthera pardus*	18	1.12	12
Leopard Cat (LC)	*Prionailurus bengalensis*	75	4.67	29
Long-tailed Macaque (LC)	*Macaca fascicularis*	75	4.67	14
Malay Civet (LC)	*Viverra tangalunga*	7	0.44	4
Malayan Porcupine (LC)	*Hystrix brachyura*	87	5.42	16
Malayan Tiger (EN)	*Panthera tigris jacksoni*	30	1.87	18
Masked Palm Civet (LC)	*Paguma larvata*	3	0.19	2
Moonrat (LC)	*Echinosorex gymnura*	2	0.12	1
Mousedeer (LC)	*Tragulus* spp.	78	4.85	17
Oriental Small-clawed Otter (VU)	*Aonyx cinereus*	1	0.06	1
Pig-tailed Macaque (VU)	*Macaca nemestrina*	30	1.87	13
Rats	*Rattus* spp.	154	9.59	20
Sambar Deer (VU)	*Rusa unicolor*	7	0.44	5
Serow (VU)	*Capricornis sumatraensis*	5	0.31	5
Sun Bear (VU)	*Helarctos malayanus*	74	4.61	33
Sunda Pangolin (EN)	*Manis javanica*	17	1.06	12

White-thighed Leaf Monkey (NT)	*Presbytis siamensis*	2	0.12	1
Wild Pig (LC)	*Sus scrofa*	510	31.74	47
Yellow-throated Marten (LC)	*Martes flavigula*	7	0.44	7

Appendix 18. *Summary statistics for human encroachers, all humans, and non-human/non-native mammal species detected in forests and underpasses in and around eastern and western linkages, Peninsular Malaysia. N = independent detections (0.5 hr intervals); PCRI = Photographic Capture Rate Index (N/1000 trap nights); Stations = no. of camera trap stations that detected the species.*

No.	Common Name	Species	N	PCRI	Stations
		Eastern linkage forests (10,502 trap nights)			
1	Human Encroachers	*Homo sapiens*	19	1.81	13
2	All Humans	*Homo sapiens*	145	13.81	37
3	Domestic Dog	*Canis lupus familiaris*	0	0	0
4	Domestic Cow	*Bos primigenius*	0	0	0
		Eastern linkage underpasses (11,278 trap nights)			
1	Human Encroachers	*Homo sapiens*	13	1.15	8
2	All Humans	*Homo sapiens*	1253	111.10	40
3	Domestic Dog	*Canis lupus familiaris*	0	0	0
4	Domestic Cow	*Bos primigenius*	0	0	0
		Western linkage forests (12,063 trap nights)			
1	Human Encroachers	*Homo sapiens*	30	2.49	19
2	All Humans	*Homo sapiens*	464	38.46	42
3	Domestic Dog	*Canis lupus familiaris*	7	0.58	2
4	Domestic Cow	*Bos primigenius*	13	1.08	1
		Western linkage underpasses (13,841 trap nights)			
1	Human Encroachers	*Homo sapiens*	60	4.33	19
2	All Humans	*Homo sapiens*	3123	225.63	66
3	Domestic Dog	*Canis lupus familiaris*	1497	108.16	49
4	Domestic Cow	*Bos primigenius*	693	50.07	15

***Appendix 19**: Proposal (in Malay) submitted to the Terengganu state government to issue a moratorium on hunting licenses for Flying Foxes* (Pteropus spp.).

Kertas Cadangan
'Lindungi Keluang, Lindungi Buahan'
Perlindungan Sepenuhnya untuk Keluang (*Pteropus* spp.) sebagai Produk Pelancongan Negeri Terengganu Darul Iman

Latar belakang

Keluang (*Pteropus vampyrus, Pteropus hypomelanus*) yang dikenali sebagai 'Flying Fox' dalam Bahasa Inggeris adalah sejenis kelawar gergasi yang boleh didapati di Negeri Terengganu Darul Iman. Keluang Malaya/Keluang Besar *Pteropus vampyrus* merupakan spesis kelawar yang terbesar di dunia, dengan bukaan sayap yang boleh mencapai 6 kaki (1.8 m). Beratnya pula boleh melebihi 1 kg. Habitatnya merupakan kawasan hutan dan paya, dan makanannya adalah bunga, madu bunga, debunga, daun dan buah-buahan. Ia terbang berhijrah antara Malaysia dan Negara-negara lain di Asia Tenggara seperti Thailand dan Indonesia (Sumatra dan Kalimantan). Ia dilindungi oleh Akta Pemuliharaan Hidupan Liar 2010 tetapi tidak mempunyai perlindungan sepenuhnya, di mana lesen boleh dipohon daripada Jabatan Perlindungan Hidupan Liar dan Taman Negara (PERHILITAN) untuk memburunya.

Keluang memakan bunga pokok kapok.

Keluang menjilat madu bunga.

Kepentingan keluang untuk manusia

Keluang merupakan haiwan frugivor (pemakan buah) yang sangat penting di hutan tropika. Ia memainkan peranan yang penting dalam proses **penyebaran biji benih**, yang secara tidak langsung membantu mengekalkan kesihatan hutan serta menggalakkan pemulihan hutan. Ia juga membantu **pendebungaan** pokok-pokok buah, termasuklah pokok durian dan petai yang mempunyai nilai ekonomi yang sangat penting di Asia Tenggara. Nilai penjualan durian pernah dianggarkan sekurang-kurangnya RM380 juta setahun, manakala penjualan petai di Semenanjung Malaysia adalah sekurang-kurangnya RM47 juta setahun. Pendapatan ekonomi ini sebenarnya adalah hasil kerja pendebungaan oleh keluang.

Ancaman terhadap keluang

Ancaman utama yang dihadapi oleh keluang adalah **pemburuan** dan **pemusnahan hutan**. Kini, pemburuan sama ada secara sah atau haram merupakan ancaman yang paling besar sekali terhadap keluang kerana ia sangat digemari oleh pemburu bagi tujuan makanan, dagangan dan juga rekreasi. Selain daripada itu, keluang juga dibunuh kerana dianggap sebagai perosak tanaman. Disebabkan oleh aktiviti manusia ini, populasi keluang di Semenanjung Malaysia kini hanya boleh didapati di kawasan-kawasan paya yang sukar diterokai. Pemburuan dan pembunuhan keluang yang berleluasa secara besar-besaran akan memudaratkan populasinya dan mengakibatkan kepupusan kerana ia merupakan spesis yang mengambil masa yang lama untuk membiak, dan melahirkan anak hanya sekali setahun. Jika pemburuan keluang pada tahap yang sedia ada tidak dihentikan segera, ada kemungkinan spesis ini boleh pupus di Semenanjung Malaysia sekitar tahun 2015.

Keluang *Pteropus vampyrus* yang telah ditembak mati oleh pemburu berlesen di negeri Terengganu.

Flying foxes facing extinction

But Wildlife Department still mulling hunting ban

By DARSHINI KANDASAMY
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RECOGNISING that the race is on to save the flying fox, the Malaysian wildlife authority said it would consider recommendations put forth by wildlife groups on the preservation of the species — even that of a temporary hunting ban.

This was the response from the Malaysian Wildlife and National Parks Department (Perhilitan) deputy director-general 1, Misliah Mohamad Basir, when contacted by Malay Mail on claims the world's largest fruit bat could very well be extinct by 2015 in Peninsular Malaysia.

She said: "We will take into consideration all recommendations made especially that which will benefit the wildlife." However, she was unable to furnish details on a hunting ban, explaining that the wildlife act was currently being ammended.

A BBC News article on Aug 25 reported the extinction could occur if urgent steps are not taken to curb or reduce the current level of hunting being carried out.

The article 'Extinction threat to flying fox' on news.bbc.co.uk reported scientists writing in the Journal of Applied Ecology stating that some 22,000 large flying foxes, or 'Pteropus vampyrus', are legally hunted each year — and more illegally.

The article said that researchers have estimated the extinction of this species, vital in their role as seed dispersal and pollination agents, could arrive between six to 81 years time if current hunting rates continue — following a study conducted on the animal and government data on hunting licenses.

Apart from a hunting ban, BBC News quoted the journal's lead author — Dr Jonathan Epstein of Wildlife Trust — that a "co-ordinated protection management" is needed between the countries where the bats live such as Malaysia, Thailand and Indonesia.

The article also suggested a temporary hunting ban be considered by the Malaysian wildlife department, a partner to the study, to allow the flying foxes' population to recover and provide time for a complete assessment of the species' survival.

Wildlife trade monitoring network Traffic Southeast Asia senior programme officer, Noorainie Awang Anak, agreed that a temporary ban on the hunting of flying foxes would allow the government to identify if hunting alone is the cause of this animal's population decline.

"These animals are also hunted for medicinal reasons or sometimes killed by fruit farmers as they eat their crops. There is a need to study if hunting is indeed the cause of its population decline."

Noorainie suggested that Perhilitan beef up enforcement and monitoring on the ground. While a license is required to hunt the flying fox, she felt more intensive monitoring was needed to ensure hunters do not abuse the license by killing more animals than necessary or by straying out of the allocated locality.

Wildlife Conservation Society Malaysia country programme director, Dr Melvin Gumal, noted that the number of flying foxes is lower in Peninsular Malaysia in comparison to areas such as Sarawak, where there are some 200,000 flying foxes.

Gumal said this was because of the higher human population on the peninsula and the "landscape conversion" problem which reduces the number of trees, fruits and nectar available for the bats.

He explained that flying foxes only reach sexual maturity after 15 to 18 months and the young's gestation period can last up to six months. A flying fox would on average only have one young a year.

Gumal said: "They are not like rats which give birth to multiple young. When the hunting or harvesting rate of the animal exceeds its breeding capacity, a decline in the animal population will occur, such as what we are observing now."

Keratan akhbar The Malay Mail bertarikh 2 September 2009 yang memberi amaran tentang kepupusan keluang disebabkan oleh aktiviti pemburuan yang berleluasa di Semenanjung Malaysia. Tetapi sehingga kini, keluang masih tidak dilindungi sepenuhnya oleh Akta Pemuliharaan Hidupan Liar 2010.

Melindungi keluang sepenuhnya sebagai produk pelancongan

Kami ingin mencadangkan supaya pihak kerajaan Negeri Terengganu Darul Iman memberi **status perlindungan sepenuhnya** kepada keluang. Ini adalah kerana keluang boleh dijadikan satu **produk pelancongan** negeri yang membawa manfaat yang banyak. Kini, populasi keluang boleh dilihat berterbangan di kawasan kampung berhampiran Kenyir pada setiap senja, dan pelancong boleh dibawa ke kawasan tersebut untuk menyaksikan kejadian ini. Ini akan meningkatkan lagi tarikan kawasan Kenyir sebagai satu destinasi eko-pelancongan yang terkemuka.

Keluang-keluang boleh dilihat keluar berterbangan mencari makan pada waktu senja di negeri Terengganu. Gambar ini telah diambil di suatu lokasi rahsia di Kuala Berang, dan dapat dijadikan satu tarikan pelancong yang sangat menarik.

Selain daripada itu, keluang juga memainkan peranan yang penting untuk mengekalkan hutan-hutan semulajadi di kawasan Kenyir. Pendebungaan dan penyebaran biji benih yang dilakukan oleh keluang akan memastikan bahawa struktur dan komposisi pokok di dalam hutan dikekalkan secara semulajadi. Jika kawasan hutan Kenyir dapat kekal sihat, maka habitat hidupan liar akan terpelihara dan pelancong akan tertarik untuk mengunjungi kawasan ini untuk menikmati keindahan alam semulajadi. Tambahan pula, pendebungaan oleh keluang juga mengekalkan kesihatan pokok-pokok buah yang sangat penting untuk ekonomi negara, pendapatan masyarakat tempatan dan juga industri agro-pelancongan. Justeru itu, status perlindungan sepenuhnya untuk keluang secara tidak langsung akan membantu kesinambungan industri pelancongan di negeri Terengganu Darul Iman.

Cadangan-cadangan
Berikut adalah senarai cadangan kami untuk melindungi keluang secara sepenuhnya di negeri Terengganu:

1) Menghentikan untuk sementara (**moratorium**) semua jenis pemburuan dan pembunuhan semua spesis keluang supaya kajian terperinci boleh dilakukan ke atas status populasinya dan kesan-kesan pemburuan. Moratorium sebegini pernah diwujudkan oleh pihak PERHILITAN bagi rusa (*Rusa unicolor*) dan kijang (*Muntiacus muntjak*) sebagai usaha melindungi harimau Malaya (*Panthera tigris jacksoni*) dan haiwan makanannya. Usaha sebegini wajar dicontohi dengan menghentikan pengeluaran lesen untuk pemburuan keluang.

2) Wartakan secara serta-merta kawasan paya yang telah dikenalpasti merupakan habitat keluang, sebagai **kawasan perlindungan** sama ada taman negeri ataupun santuari hidupan liar.

Kesimpulan
Keluang merupakan spesis rantauan Asia Tenggara yang amat penting untuk manusia dan alam sekitar, tetapi kini ia sangat terancam disebabkan oleh aktiviti manusia. Disebabkan oleh ini, adalah sangat penting untuk mewujudkan kerjasama serantau supaya spesis ini dapat dilindungi. Keluang telah pun dilindungi sepenuhnya di Thailand, serta negeri Sarawak dan Sabah. Negeri Johor pula telah mengharamkan semua jenis pemburuan hidupan liar. Usaha Negeri Terengganu Darul Iman untuk melindungi secara sepenuhnya spesis yang penting ini dapat membantu meningkatkan ekonomi negeri dalam jangka panjang. Kami amat berharap bahawa pihak kerajaan negeri dapat mempertimbangkan cadangan-cadangan yang telah dikemukakan di sini, dan mengambil tindakan serta-merta untuk melindungi keluang secara sepenuhnya. Kami dengan segala hormatnya ingin memohon untuk mengadakan satu perjumpaan dengan pihak tuan/puan untuk membincangkan isu ini.

Untuk maklumat lanjut, sila hubungi:
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Mengenai Rimba:
Rimba adalah sekumpulan ahli biologi yang menjalankan kajian mengenai spesis dan ekosistem terancam di Malaysia. Kami ditubuhkan pada November 2010. Kami bukan NGO, syarikat perunding ataupun persatuan – kami merupakan kumpulan penyelidikan bukan keuntungan (*non-profit*) yang berdaftar di Malaysia (002085549-T). **Rimba** tidak menentang usaha-usaha pembangunan ekonomi seperti pembalakan atau pembangunan infrastruktur, tetapi kami percaya bahawa kesan-kesannya dapat dikurangkan. Di samping penyelidikan asas kami juga menjalankan penyelidikan gunaan yang dapat menghasilkan cadangan pengurusan berdasarkan jumpaan saintifik yang akan membantu ahli pentadbir untuk mengurangkan (atau menghentikan)

ancaman-ancaman terhadap ekosistem dan spesis di Malaysia. Ini adalah misi kami dan kami semangat menjalankannya kerana kita semua memerlukan rimba di luar sana!

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Appendix 20: *Proposal submitted to the Terengganu state government to gazette the eastern linkage as part of the Kenyir Wildlife Corridor.*

ESTABLISHING THE KENYIR WILDLIFE CORRIDOR TO SAFEGUARD TERENGGANU'S ECONOMY AND NATURAL HERITAGE

"*Today… the tiger is in grave danger. Its endangered status is an indicator of ecosystems in crisis. Let us not be proud of a tiger economy without real tigers in the forest.*" - Datuk Douglas Uggah Embas, Minister, Ministry of Natural Resources and Environment

Prepared by:

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Prof. Emeritus Dr Faizah Shahariom, Universiti Malaysia Terengganu

For:

Dato' Toh Chin Yaw, Terengganu Chairman of Industry, Trade and Environment Committee, Terengganu State Government

RINGKASAN EKSEKUTIF

Cadangan kami adalah untuk mewujudkan Koridor Hidupan Liar Kenyir bagi melindungi ekonomi dan khazanah semulajadi negeri Terengganu.

Koridor Hidupan Liar Kenyir yang dicadangkan ini sangat berpotensi untuk meningkatkan hasil industri pelancongan negeri Terengganu. Menurut satu kajian ekonomi di Malaysia baru-baru ini, ekonomi negeri Terengganu berpeluang menjana sebanyak RM61 juta setahun jika satu mekanisme jangka panjang dapat diwujudkan untuk mengutip hasil daripada penduduk di sekitar Kuala Lumpur yang sanggup membayar untuk melindungi hutan sebesar hutan koridor yang dicadangkan ini. Secara keseluruhan, nisbah faedah-kos untuk melindungi hutan koridor ini adalah 3:1.

Koridor Hidupan Liar Kenyir yang dicadangkan adalah antara kawasan yang paling kaya dalam kepelbagaian mammalia dan burung. Sekurang-kurangnya 40 spesies mammalia telah direkodkan semasa tinjauan kamera perangkap di koridor ini, termasuklah 18 spesies mammalia terancam seperti gajah, harimau dan tapir – tahap kepelbagaian ini adalah sama tinggi seperti yang pernah direkodkan semasa tinjauan kamera perangkap di Taman Negara. Selain daripada itu, sekurang-kurangnya 290 spesies burung telah direkodkan di Kenyir, termasuklah 9 daripada 10 spesies enggang di Malaysia, dan juga burung yang terpantas di dunia, iaitu *peregrine falcon*. Malangnya, hampir ¼ daripada spesies burung di sini adalah terancam (bersamaan dengan 66 spesies). Akhir sekali, 2 spesies bunga yang terbesar di dunia, *Rafflesia*, boleh didapati di koridor ini. Justeru itu, Koridor Hidupan Liar Kenyir yang dicadangkan ini mengandungi sebahagian besar daripada khazanah semulajadi negeri Terengganu, dan harus dilindungi untuk generasi akan datang.

Walaupun hutan-hutan simpan yang terdapat di Koridor Hidupan Liar Kenyir telah pun dikenalpasti sebagai penghubung utama oleh Pelan Induk Rangkaian Ekologi *Central Forest Spine* dan juga perunding-perunding Wilayah Ekonomi Pantai Timur (*East Coast Economic Region* atau ECER), ia masih diancam oleh: 1) penukaran hutan kepada ladang getah secara sah; 2) pembersihan hutan semulajadi untuk pembinaan empangan hidroelektrik Puah; 3) pengambilan tumbuhan dan pemburuan haram oleh penceroboh, yang kebanyakannya datang dari negara-negara Indochina.

Untuk menangani ancaman-ancaman ini, kami mencadangkan yang berikut:

1) Wartakan sebanyak 415 km^2 hutan di Kenyir sebagai kawasan perlindungan untuk mengurangkan ancaman penukaran hutan – kawasan ini akan dikenali sebagai Koridor Hidupan Liar Kenyir, iaitu Koridor Hidupan Liar yang pertama sekali diwartakan secara rasmi di negeri Terengganu dan juga Semenanjung Malaysia.

2) Adakan dua sekatan jalan raya yang tetap pada dua tapak di Koridor Hidupan Liar Kenyir untuk mencegah pemburuan haram.

3) Tubuhkan satu pasukan pencegahan pemburuan haram yang terdiri daripada agensi-agensi penguatkuasa yang berkenaan untuk menjalankan rondaan semasa dan menangkap penceroboh di Koridor Hidupan Liar Kenyir.

Jika Koridor Hidupan Liar Kenyir diwartakan, negeri Terengganu berpeluang diiktiraf sebagai pelindung khazanah alam pada peringkat negara dan antarabangsa. Ini dapat menarik lebih banyak pengunjung dan meningkatkan hasil industri pelancongan negeri.

EXECUTIVE SUMMARY

We propose the establishment of the Kenyir Wildlife Corridor to help safeguard Terengganu's economy and natural heritage.

The proposed Kenyir Wildlife corridor has tremendous potential to increase tourism revenue for Terengganu. Based on estimates from a recent economic study in Malaysia, the Terengganu economy may receive a boost of RM61 million/yr if a long-term mechanism is created to collect revenue from the willingness of Malaysian residents in and around Kuala Lumpur to pay for the protection of forests the size of the proposed corridor. Overall, the benefit-cost ratio of protecting the forests in this corridor is 3:1.

The proposed Kenyir Wildlife Corridor contains one of Malaysia's richest diversity of mammals and birds. At least 40 mammal species have been recorded in camera trap surveys of this corridor, including 18 threatened mammal species such as elephants, tigers, tapirs – this diversity is as high as that recorded by previous camera trap surveys in Taman Negara. Also, at least 290 bird species have been documented in Kenyir, including 9 out 10 of Peninsular Malaysia's hornbill species, and the world's fastest bird, the peregrine falcon. Unfortunately, almost a quarter of the total, or 66 bird species, are threatened. Finally, two species of the world' largest flower, the *Rafflesia*, can be found in this corridor. The proposed Kenyir Wildlife Corridor therefore contains a large proportion of Terengganu's natural heritage that must be protected for future generations.

Although the forest reserves in the proposed Kenyir Wildlife Corridor have been identified as a primary linkage in the Central Forest Spine Master Plan for Ecological Linkages and by consultants from the Eastern Corridor Economic Region (ECER), they are still threatened by: 1) legal conversion of natural forests to rubber plantations; 2) clear-felling of natural forests for the proposed Puah hydroelectric dam; and 3) poaching of plants and animals by encroachers, largely made up of foreigners from Indo-China.

In order to address these threats, we propose the following:

1) Gazette around 415 km^2 of forest in Kenyir as a protected area to minimize the threat of forest conversion - this will be known as the Kenyir Wildlife Corridor, Terengganu's and Peninsular Malaysia's first officially gazetted Wildlife Corridor.

2) Establish two permanent road blocks at two sites within the proposed Kenyir Wildlife Corridor to deter poaching.

3) Establish an anti-poaching task force made up of relevant enforcement agencies to conduct regular forest patrols in the proposed Kenyir Wildlife Corridor to capture foreign encroachers.

If the Kenyir Wildlife Corridor is gazetted, Terengganu stands to gain local and international recognition as a steward of the environment – this is likely to attract more visitors to Terengganu and increase the revenue for its thriving tourism industry.

WHY KENYIR IS IMPORTANT FOR TERENGGANU'S ECONOMY AND NATURAL HERITAGE

1) Because of the forests

While production forests generate income, preserved forests also generate income by attracting **eco-tourism** activities, the fastest growing sector in tourism today.

In 2009, Terengganu's tourism revenue was estimated to be around **RM2.6 billion**[1].

Preserved forests are worth more that logged forests based on recent economic studies. ©Paul Henry

Based on an economic study[2], Malaysian residents are willing to pay as much as **RM252.7 million/yr** to prevent logging of forests as large as 3,000 km^2 in Perak.

The opportunity costs, such as revenue from timber extraction and job creation in the timber industry forgone from not logging a forest of that size, were estimated at **RM89.4 million/yr**. Therefore, there is a net benefit from protecting forests.

Based on these estimates, the Terengganu economy may receive a boost of **RM61 million/yr** if a long-term mechanism is created to collect revenue from the willingness of Malaysian residents in and around Kuala Lumpur to pay for the protection of forests the size of the proposed corridor. Overall, the benefit-cost ratio of gazetting the Kenyir Wildlife Corridor can be **3:1**.

[1] Zolkepli, F. 2011. Kuala Terengganu City Centre expected to double RM2.6 bil tourism revenue. The Star. Oct 20.
[2] Economics team of the UNDP/GEF/FRIM CBioD Project 2011. Valuing the protection of Belum-Temengor to the Malaysian public. 20pp.

2) Because of the wildlife

There is tremendous **potential** for Kenyir to attract tourists to appreciate its wildlife.

Since 2010, at least **40 mammal species** have been recorded by camera trap surveys by Rimba in the proposed Kenyir Wildlife Corridor, including **18 threatened mammal species** such as elephants, tigers, tapirs – this diversity is as high as that recorded from camera trap surveys conducted in Taman Negara more than 10 years ago. Of these, 18 are threatened, including 5 endangered species according to the International Union for the Conservation of Nature Red List.


A chance to spot elephants swimming in Lake Kenyir is a potential eco-tourism draw. ©Paul Henry

6 out of Peninsular Malaysia's 8 wild cat species have been recorded by researchers from Rimba in the proposed Kenyir Wildlife Corridor.


People want to come to forests where the 'king of the jungle' still roams. This is just one of six tigers recorded in the proposed Kenyir Wildife Corridor. ©Rimba

At least **290 bird species** have been recorded by the Malaysian Nature Society in the forests around Kenyir, including the world's fastest bird – the peregrine falcon. Of these, almost a quarter, or 66 species, have been classified as **threatened** by the International Union for the Conservation of Nature Red List.

Kenyir is also one of the best sites in Malaysia for observing hornbills. **9 out of Peninsular Malaysia's 10 hornbill species** are regularly found in the proposed Kenyir Wildlife Corridor — more than the whole of Sabah and Sarawak!

There is a tremendous potential for Kenyir to generate eco-tourism revenue from bird enthusiasts, who will come to see birds such as this Rhinoceros hornbill in the proposed Kenyir Wildife Corridor. ©Anuar McAfee

The **first animal** to be named after Kenyir, a land snail known as *Kenyirus sodhii*, as well as two species of the world's **largest flower**, the *Rafflesia*, can also be found in the proposed Kenyir wildlife corridor.

Sections of the proposed Kenyir Wildlife Corridor where two species of *Rafflesia* have been found can be designated as viewing areas to attract tourists. ©Rimba

HOW THE KENYIR WILDLIFE CORRIDOR CAN PROTECT TERENGGANU'S ECONOMY AND NATURAL HERITAGE

1) By minimizing forest conversion

The forests in the proposed Kenyir wildlife corridor lie in four production **forest reserves**: Hutan Simpan Tembat, Hulu Telemong, Hulu Nerus and Petuang. While we support sustainable forest management, production forest reserves can still be degazetted at any time and cleared for **hydroelectric dams** such as the Tembat and Puah dams, or legally converted to **rubber** plantations without degazettement – this is happening in Hutan Simpan Lebir (Kelantan) at the Kelantan-Terenggnau border.


A stark contrast of Hutan Simpan Lebir (Kelantan) cleared for rubber plantations with the beautiful forests of Hutan Simpan Tembat (Terengganu). © William Laurance

We do not wish to see this happen to forest reserves in the proposed Kenyir Wildlife Corridor as this will have a **negative effect** on wildlife, forests and water sources, which provide important ecosystem services for the people of Terengganu. Continued clearing of forests for dams and plantations will consequently have an **adverse effect** on Terengganu's tourism and economy

2) By minimizing poaching

Terengganu's wildlife requires urgent protection from **poachers**.

As of 2012, at least **111** encroachment camps, most of which were made by foreigners from Indo-China, have been detected by researchers from Rimba in the proposed Kenyir Wildlife Corridor.

A foreign encroachment camp for at least 19 people found in the proposed Kenyir Wildlife Corridor. ©Rimba

Rimba scientists have also met **6 Indo-Chinese poachers** in the proposed Kenyir Wildlife Corridor – this is a national security issue. Not only are they after Terengganu's trees...

Two Indo-Chinese poachers detected by camera traps in the proposed Kenyir Wildlife Corridor. ©Rimba

...they are also setting traps for endangered wildlife. More than **155 snare signs** have been found so far in the proposed Kenyir Wildlife Corridor.

A young elephant found snared (on the right front leg) in the proposed Kenyir Wildlife Corridor. ©Rimba

3) By providing the last place for animals to move freely from Taman Negara to the rest of Terengganu's forests

The proposed Kenyir Wildlife Corridor, which is fragmented by the Kuala Berang highway, is the **last place in Terengganu** where animals can move freely from Taman Negara to the rest of the forests in Terengganu.

This area has also been identified as Primary Linkage 7 in the **Central Forest Spine** Master Plan for Ecological Linkages and important linkage by consultants from the **Eastern Corridor Economic Region (ECER)** currently working on an ecotourism project for Taman Negara**.**

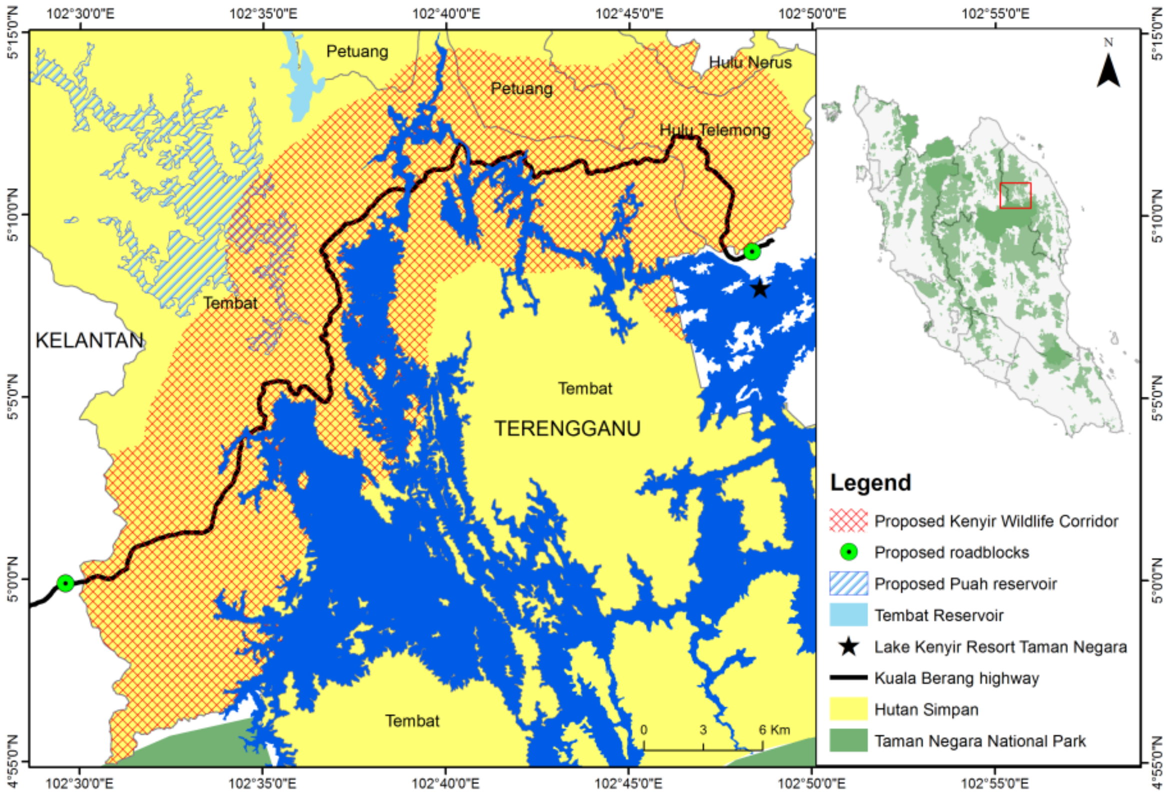
Without this corridor, Terengganu's wildlife may soon be **isolated** from Taman Negara.

A rare leopard seen crossing the Kuala Berang highway in the proposed Kenyir Wildlife Corridor. ©Rimba

HOW DO WE CREATE THE KENYIR WILDLIFE CORRIDOR?

1) Gazette a corridor[1] containing **415 km²** of forest as a: 1) Wildlife Reserve or Wildlife Sanctuary under the Wildlife Conservation Act 2010; or a 2) State Park or Forest Sanctuary for Wild Life under the National Forestry Act 1984 - this protected area will be known as the **Kenyir Wildlife Corridor** (see map).

2) Establish two **permanent road blocks** (see map) to deter poaching and other illegal smuggling activities: 1) Kelantan-Terengganu border and 2) T-junction leading to Jetty Gawi. These road blocks will involve checking of suspicious cars passing through the corridor, with rotation of personnel from the nine enforcement agencies listed below.



3) Establish an anti-poaching **task force** made up of nine enforcement agencies to coordinate regular forest patrols to apprehend foreign encroachers: Polis Diraja Malaysia (PDRM), Polis Gerakan Am (PGA), Jabatan Perlindungan Hidupan Liar (PERHILITAN), Jabatan Perhutanan Negeri Terengganu (JPNT), Unit Pencegahan Penyeludupan (UPP), Askar Watahniah, Jabatan Immigresen, RELA and Pertahanan Awam. This task force should also have a dedicated administrative unit that oversees the implementation of the road blocks and ensures wildlife-related offences are prosecuted in the Green Courts.

By implementing these actions, Terengganu would be the **first state** in Malaysia to meet the requirements of the federal government's Central Forest Spine Master Plan for Ecological Linkages to protect a primary linkage. Therefore, Terengganu will be recognized as the creator of **Peninsular Malaysia's first wildlife corridor.**

[1] The size of the corridor is based on estimates that forests 5 km from a road must be protected to prevent mammal abundance from being negatively affected: Benítez-López, A., Alkemade, R., and Verweij, P. A. 2010. The impacts of roads and other infrastructure on mammal and bird populations: a meta-analysis. Biological Conservation 143: 1307-1316.