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Conserving tigers in Malaysia: A science-driven approach for eliciting conservation policy change

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ABSTRACT

The unprecedented economic growth occurring across Southeast Asia is causing large tracts of rainforest to be logged, converted to plantations or fragmented by infrastructure development. It also opens up forest to poachers which, in combination, places acute pressure on the region's large carnivores. Here, we focus on one of Malaysia's three priority tiger landscapes that illustrate these regional conservation challenges. The Royal Belum State Park (RBSP) and Temengor Forest Reserve (TFR) are connected by a strip of unprotected forest with portions assigned for conversion to monoculture plantations. To support government in setting aside wildlife corridors, we assessed: the abundance of tiger and principle prey under two different forest management regimes in RBSP and TFR; and, tiger habitat use in the unprotected forest strip, from which a spatially-explicit habitat model was produced to identify priority points of forest connectivity. Camera trapping revealed a threefold higher tiger density in the protected area (RBSP) than the forest reserve subjected to selective logging (TFR), which was likely explained by the higher relative abundance of its principal prey, seemingly lower levels of poaching as indicated from an independent study and presence of armed forces that may have deterred poachers. Two forest corridors were identified as being important for maintaining landscape connectivity and these findings were used to successfully lobby state government in affording them protection. This research offers an urgently needed approach for better managing Malaysian tiger habitat within forest reserves, which are predominantly designated for logging and have weak or non-existent wildlife protection measures.

1. Introduction

The unparalleled pace of economic growth in Asia is pushing numerous species to the edge of extinction. High demand for wildlife products, such as rhino horn and tiger bone, is increasing poaching pressures, while rapid infrastructure development, including a proliferation of road networks, is fragmenting forest habitat (Bennett, 2011; Laurance et al., 2009). In addition, fragmentation of habitat caused by selective logging particularly in terms of the edge effect (certain sensitive species avoid boundaries) poses a threat to the long-term survival of wildlife populations, particularly large mammals (Grieser Johns, 1997). These forces of change are most adversely affecting large-bodied mammals because of their prized status, relatively slow reproduction rates, wide range requirements and naturally low population densities (Clements et al., 2010).

Scientific research into the possible impact of infrastructure development and the identification of alternative options that are compatible with wildlife management are urgently needed. For these studies to be meaningful, they need to translate into on-the-ground action. This requires conservation scientists closely engaging with policy makers and conservation managers to greatly increase the likelihood of their recommendations being adopted (Knight et al., 2008; Laurance et al., 2009). Such an approach is pertinent for managing Peninsular Malaysia's forest. Here, 80% of its 59,230 km^2 forest estate is primarily designated for selective logging within Permanent Reserved Forests. Yet, of the presumed

Abbreviations: RBSP, Royal Belum State Park; TFR, Temengor Forest Reserve; BT-SLF, Belum-Temengor State Land Forest; PCRI, photo capture rate index; FR, Forest Reserve; RSF, resource selection functions; SECR, spatially explicit capture recapture.

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500 tigers (*Panthera tigris*) living in Malaysia, 85% of their habitat is located within these forests (Kawanishi et al., 2003). Best evidence now suggests that there are between 250 and 340 adult tigers in the country (DWNP and MYCAT, 2014).

Malaysia contains three large forest landscapes that are designated as both national and global tiger conservation priorities (Dinerstein et al., 2006; DWNP, 2008). All three areas span multiple land use types and maintaining their forest connectivity is critical for the long-term viability of tigers, as well as other wide-ranging mammals. However, beyond estimating tiger population density, no research on how spatial planning affects tigers has yet to be published from any of these landscapes (Kawanishi and Sunquist, 2004; Lynam et al., 2007; Rayan and Mohamad, 2009).

The challenges involved with managing these landscapes are exemplified by the Belum-Temengor Forest Complex in northern Peninsular Malaysia. It consists of a primary state park (Royal Belum State Park; RBSP) that is connected to a Permanent Reserved Forest (Temengor Forest Reserve; TFR) via state land forest that lacks a formal protection status (Belum-Temengor State Land Forest; BT-SLF) and contains a two lane public highway (Gerik-Jeli East–West Highway). In 2009, the susceptibility of BT-SLF to conversion became evident when several forest patches were clear-felled as part of a state government highland cash crop project. This happened despite existing national government plans that stressed the importance of maintaining forest integrity in the landscape (DTCP, 2005, 2009) and plans to build a viaduct, but during this time a lack of spatial planning expertise within government, readily available wildlife data and inter-agency and inter-sectoral coordination meant that wildlife concerns were not fully considered. Carnivores are known to cross roads in locations that vary with passage characteristics, road-related attributes, surrounding habitat characteristics, and human disturbance levels and therefore, understanding the habitat use patterns of a species on both sides of a highway is critical to identify a matrix of suitable patches surrounding proposed viaduct locations, so that these are protected (Clevenger and Waltho, 2000).

Where tigers occur and how they move within the Belum-Temengor Forest Complex, and indeed other such mosaic landscapes, is poorly understood in Malaysia and therefore a conservation research priority. Recognising this, we carried out wildlife surveys with an explicit aim of providing science-based management recommendations to the Perak State authorities to identify corridor and viaduct placements that closely matched high quality large mammal habitat (Rayan et al., 2012). In this paper, we conduct the first comprehensive tiger and prey assessment in a global priority Tiger Conservation Landscape in Malaysia. We aim to investigate species abundance in two forest estates under different management regimes, tiger habitat use within the BT-SLF forest strip and then use these results to work with government to set aside critically important corridors and identify potential viaduct locations. Finally, we describe our approach in engaging different government agencies and its outcome in influencing policy.

2. Material and methods

2.1. Study areas

The Belum-Temengor Complex consists of three study areas; RBSP, BT-SLF and TFR, which are all located in the state of Perak (101°15′0″–101°46′0″E and 5°55′0″N–5°0′0″N; Fig. 1) and cover 2922 km^2. RBSP is home to an estimated 740 indigenous people (*Orang Asli*) and TFR an estimated 5000 indigenous people (Department of *Orang Asli* Affairs, unpublished data). Within the BT-SLF, there are about 570 indigenous people (DTCP, 2009). The *Orang Asli* primarily hunt small mammals and primates, and rear poultry, but do not keep larger livestock that might be attractive to tigers (Mark Rayan, unpublished data). Over the past decade, there have been only two incidents of people being attacked by tigers in the Complex (Mark Rayan, unpublished data).

The 1175 km^2 RBSP was officially gazetted in 2007 as a strictly protected area with only non-exploitive commercial activities, such as tourism by the edge of a lake. It ranges from 260 m to 1533 m a.s.l and consists of lowland dipterocarp (5.6%), hill dipterocarp (71.5%), upper dipterocarp (20.9%) and montane (2.0%) forests. The 1489 km^2 TFR was officially gazetted in 1991 as a Permanent Reserved Forest and is an active production forest undergoing selective logging through a second cycle. The forest ranges from 260 m to 2160 m a.s.l and consists of lowland dipterocarp (4.2%), hill dipterocarp (34.4%), upper dipterocarp (41.7%) and montane (19.7%) forests.

RBSP and TFR are intersected by the 131 km^2 BT-SLF forest strip (2.4 × 33.9 km) that is bisected by the Gerik-Jeli East–West Highway, which provides access to poachers and is a potential barrier to tiger movement (Clements et al., 2010; WWF-Malaysia, 2011). At the time of the study, BT-SLF was classified as 'state land forest' and therefore had no protection status, making it vulnerable to conversion for agriculture, plantations or infrastructure by state authorities. The BT-SLF ranges from 260 to 1265 m a.s.l. and consists of lowland dipterocarp (0.2%), hill dipterocarp (43.5%), upper hill dipterocarp (55.9%) and montane (0.4%) forests.

2.2. Data collection and field methods

Two field survey designs were applied from October 2009–January 2011. For the larger areas of RBSP and TFR, 2 × 2 km grid cell sampling units were used to guide the placement of camera traps for investigating tiger density and relative prey abundance (Karanth et al., 2008). To assess finer-scale tiger habitat use in the BT-SLF corridor, 1 × 1 km grid cell sampling units were surveyed for indirect sign. Camera trapping was not able to be conducted during sign surveys in BT-SLF as there were not enough camera traps due to problems with theft and damage by wildlife.

Camera trapping was first conducted for about nine months in TFR and then RBSP, with a sampling area of about 400 km^2 for each of the tiger density surveys. In 2 × 2 km grid cells, 35 fixed location paired camera traps were set to record both flanks of a passing tiger. To maximise study area coverage, as part of a separate study on habitat use, another 70 single placement camera traps were set in neighbouring grid cells. To increase spatial coverage within grid cells, these 70 traps were moved once within the same cell after 3–4 months of operation, corresponding to a total of 140 single placements. These provided a total of 175 camera placements with about an average inter-trap distance of 1 km for tiger density surveys. To increase species detection rates, placements were made along forest trials, ridge trails and inactive logging roads. Camera traps were checked every 2–3 months to retrieve data and replace batteries.

In the BT-SLF study area, indirect species signs were recorded through three repeat surveys (each defined as a sampling occasion) in a grid cell over five months. Surveys were conducted by six teams, each of two personnel (a field biologist and an indigenous guide). Each team was rotated in sequence to minimise observer bias between temporal replicates. Repeat surveys in a grid cell were conducted at 1–2 month intervals. Each team was required to survey a minimum of 1 km per 1 km^2 grid cell containing tiger habitat. Each team intensively searched areas considered to have the highest likelihood of tiger and prey sign (i.e. tracks <1 month old), such as forest trails, ridges, sand beds, river banks, saltlicks and logging roads.

Presence of tiger, prey species: sambar (*Rusa unicolor*), muntjac (*Muntiacus muntjak*), wild pig (*Sus scrofa*) and gaur (*Bos gaurus*),

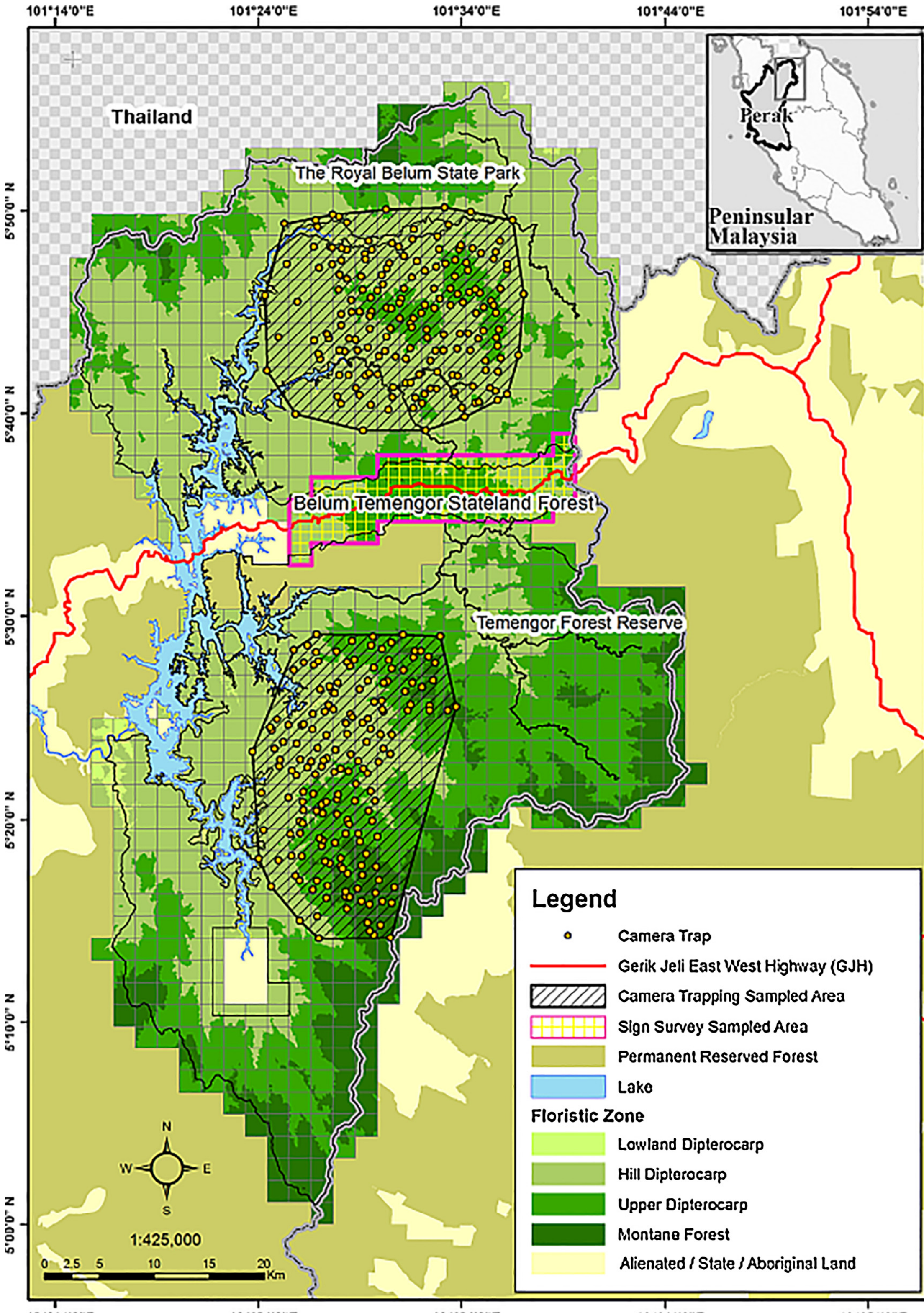

Fig. 1. Location of the three study areas; Royal Belum State Park, Temengor Forest Reserve and Belum Temengor State Land Forest in Perak State.

different encroachment types (camp sites and foreign tree markings), poaching (cage traps, bullet cartridges and snare traps) and weather condition (rain or no rain within a prior 24-h period) were recorded within each sampling unit. It should be noted that encroachment and poaching signs were recorded opportunistically during observer searches for animal signs and that no dedicated anti-poaching patrols were conducted for these species. For subsequent verification, a photograph of each sign detected was taken.

To increase the likelihood of independence between repeat surveys, the start of each survey route began with a 100 m walk following a random bearing at the edge of a 1 × 1 km grid cell.

2.3. Data analysis and dissemination

For each camera placement, a photo capture rate index (PCRI) for tiger, individual prey species, humans and vehicles were calculated as the number of independent photographs (detections) per 100 trap-nights. These estimates were averaged across all camera placements within each study area to produce respective mean PCRIs and standard errors. An independent photograph was defined as a photograph being taken at least 30 min apart at the same camera trap location for the same indicator (O'Brien et al., 2003). Differences in the study area PCRIs were assessed using a Wald test (Morgan, 2008), whereby Z values larger than 1.96 (critical value at $\alpha = 0.05$) were considered significant.

A maximum likelihood-based spatially explicit capture recapture (SECR) framework was used to estimate tiger density using the *secr* package (Efford, 2011) in R software environment v2.10.1 (R Development Core Team, 2010). Camera traps were treated as proximity detectors that allowed the animal to be detected at multiple traps for any occasion. Once all 35 paired camera traps were set in the field, only data (inclusive of the 140 single traps) from the first three months were used, so that the likelihood of violating the demographic closed population assumption was minimised (Karanth and Nichols, 1998). From this data set, detection histories were constructed according to a 24-h period of camera trapping which represented a unique occasion and whether the camera trap was functioning (1) or not (0).

To approximate a buffer width, the root pooled spatial variance, which is a measure of the two dimensional dispersion of locations where individual animals are detected, was calculated for each study area using the *secr* package. Based on the recommendations by Efford (2004), the variance values were multiplied by a factor of four to approximate a buffer width, as an animal outside such a buffer width has a low probability (<0.001) of being photographed and therefore unlikely to influence the density estimate. The buffer width was used to extend the area of integration incorporating a habitat mask, whereby a 0.336 km^2 resolution grid was constructed following Royle et al. (2009) for forest habitat excluding the lake and human settlements. A closure test (Otis et al., 1978) was conducted within the same *secr* package (Efford, 2011).

Tiger density was estimated using conditional likelihood SECR with a half-normal detection function from a Poisson distribution, with the constant default models; magnitude of the detection function ($g0[.]$) and spatial scale parameter ($\sigma[.]$). SECR-based density estimates and corresponding 95% confidence intervals were used to estimate the expected number of tigers within each study area. Differences in density estimates between study areas were assessed using a Wald test, whereby Z values larger than 1.96 (critical value at $\alpha = 0.05$) were considered significant.

Tiger habitat use inside the BT-SLF corridor was estimated using indirect sign (track) survey data within a likelihood-based sampling approach (MacKenzie, 2006). Tiger detection histories were constructed for each site over three temporal sampling occasions, each comprising an independent sign survey. Next, a spatial data set was constructed by extracting the following habitat and anthropogenic disturbance covariates for each site: (i) distance to settlement (permanent human presence, indigenous settlements and logging camps); (ii) mean elevation; (iii) mean Normalized Difference Vegetation Index (NDVI; used as a proxy to describe the bio-structural changes in vegetation caused by logging (e.g. Hamel et al., 2009; Krishna et al., 2008); and, (iv) distance from the Gerik-Jeli East–West Highway. The elevation data were originally derived from 20 m interval contour lines on L7030 Topographic Map and a Digital Elevation Model (DEM) was generated from the contour lines and re-sampled to 50 m spatial resolution. Normalized Difference Vegetation was analysed using ASTER satellite images with 15 m spatial resolution. Due to the effect of cloud cover, a series of ASTER images in different years were masked and mosaic-ed. The images were closely matched to the sampling period in each study area, which consisted of images for 2008, 2009, 2010 and 2011. NDVI was computed by using ERDAS Imagine Version 9.0. Evidence of collinearity between the continuous covariates was tested in SPSS v16.0 software (SPSS Inc., Chicago, IL., USA) using a Spearman's rank correlation. Covariates were treated as independent and included within the same model, if their correlation coefficient was <0.6.

To explicitly account for variation in detection probability (p), three covariates were modelled: (i) survey effort per grid cell (distance walked adjusted for varying topography by overlaying two-dimensional GPS unit track-logs onto a three-dimensional digital elevation model); (ii) a binary variable for whether it rained (1) or not (0) within a 24-h period within a unit prior to surveying; and, (iii) categorical variables (1–5) assigned for each observer (excluding the reference observer) in order to account for observer bias.

All continuous covariates were transformed into standardized z-scores and the combination of covariates that best explained tiger habitat use and detection probability were investigated using PRESENCE v4.0 software (Hines, 2006) under the single-species, single-season framework. A two-step modelling approach was used. First, detection probability was modelled where the parameter was either assumed constant or allowed to vary with individual or additively combined covariates. For each model, a global model for the probability of habitat use (ψ) was maintained (MacKenzie, 2006). Subsequently, the influence of covariates on habitat use was modelled where the parameter was either assumed constant or allowed to vary with individual or additively combined covariates, whilst maintaining the top ranked model for detection probability as derived from the first step.

Models were ranked using the small-sample correction to Akaike's information criterion (AICc; Burnham and Anderson, 2010) by changing the effective sample size (defined as the number of sites). Model fit was evaluated by comparing the observed Pearson chi-square statistic from the global model with chi-square statistics from 10,000 simulated parametric bootstrap data sets (MacKenzie and Bailey, 2004). Poor model fit (i.e. $\hat{c} > 1.0$) was accounted for by estimating an over-dispersion factor ($\hat{c}$) and inflating the corresponding standard errors by a factor ($\sqrt{\hat{c}}$) and by using a quasi-likelihood over-dispersion parameter (QAICc) for model selection (Burnham and Anderson, 2010). Model selection uncertainty was based on the evidence ratio between the top ranked model and the next best model (Burnham and Anderson, 2010).

Covariates that were likely to affect detection and habitat use probabilities were identified based on the covariates that were contained in the top ranked model and relative summed Akaike weights ($\sum w_i$) of models that contained a particular covariate; where $\sum w_i > 0.50$ were considered indicative of a strong habitat use response to a covariate (Kalies et al., 2012). To assess spatial autocorrelation in the response variables that were not accounted for by the predictor variables, Moran's I statistic (Cliff and Ord, 1981) was calculated for residuals from occupancy models following Moore and Swihart (2005) for the top ranked model using the Crime-Stat v1.1 software (N Levine and Associates, Annadale, VA).

Within a GIS (ArcGIS v9.3, ESRI 2008), the predicted intensity of habitat use from the top ranked models were used to spatially identify tiger habitat use intensity in the 156 sites that covered the BT-SLF study area. The intensity of habitat use estimates were categorised as; 'Very high', 'High', 'Moderate' and 'Low' using a natural breaks classification function within a GIS. The first two intervals were considered to represent the most important tiger habitat and used to delineate suitable viaduct locations and forest corridors for protection.

Finally, from January 2011–May 2013, WWF-Malaysia actively engaged with various state and national government planning, wildlife and policy development agencies to initially develop this research and then disseminate its findings, in particular the status of protected wildlife in the landscape and the importance of maintaining connectivity between RBSP and TFR (Rayan et al., 2012). Here, discussions focused on the importance of the forest for tigers and their prey, and where corridors and viaducts should be placed. In parallel, WWF-Malaysia highlighted the need on protecting the corridors through the local media to raise this issue and to foster strong public support.

3. Results

3.1. Photo capture rates

A total of 1105 photos of tigers, representing 3.6% of the 30,282 wildlife photos, over 33,727 trap-nights were obtained from both study areas. A total of 153 and 458 tiger detections were obtained from 15,969 and 17,758 trap-nights from TFR and RBSP, respectively. Eight individual adult tigers (five males, two females and one of unknown sex), as well as four juveniles, were recorded in TFR exclusively, whilst 21 individual adult tigers (two males, 16 females and three of unknown sex), as well as five juveniles were recorded exclusively in RBSP. One additional adult male was detected once in TFR and twice within a 24-h period in RBSP in camera traps close to the adjoining study area boundaries and towards the end of the sampling period.

In RBSP, the PCRI was significantly higher than in TFR for tiger (2.7 times higher), sambar (77.5), muntjac (3.7) and wild pig (1.6; Table 1). The Southern serow (*Capricornis sumatraensis*) was only detected in TFR and although gaur PCRIs were not significantly different between study areas, herds were only recorded in RBSP, indicating a greater biomass. The most common type of human activity recorded by camera traps was that of *Orang Asli*, which was similar in both study areas (Table 1). Vehicular traffic was only recorded in TFR, as RBSP does not contain roads. Army personnel were only photographed in RBSP due to its proximity to a fenceless border with neighbouring Thailand. Foreign encroacher detections in RBSP ($n = 44$) was higher compared to that in TFR ($n = 2$). Two detections of local encroachers with firearms were obtained in TFR and none in RBSP. Nine and three active wire snares were discovered in BT-SLF and RBSP respectively, whereas no snares were found in TFR. However, a hunting platform and bullet cartridges in another location were found in TFR and not in RBSP and BT-SLF.

3.2. Tiger population density

Restricting the camera trap data to three months yielded trap night efforts of 5386 (TFR) and 7771 (RBSP) with 55 and 232 independent tiger detections, respectively. These represented four individual adult tigers (two males and two females) from TFR and 17 individual adult tigers (two males, 13 females and two of unknown sex) from RBSP. A root pooled spatial variance value of 3688 m (TFR) and 3767 m (RBSP) was recorded that corresponded to 14.8 km for TFR and 15.1 km for RBSP with a multiplication of four, and so a 15 km buffer width was used to extend the area of integration for both study sites which was characterised by suitable habitat by a grid mesh of 6313 cells (TFR) and 6709 cells (RBSP) over the buffered area. The SECR statistical closure test

Table 1
Detections (N) and photo capture rate index (PCRI) of tiger, tiger prey species with a body mass of >2 kg (excluding carnivores and species with less than 10 detections) and human activity in Temengor Forest Reserve (TFR) and Royal Belum State Park (RBSP).

Category	TFR		RBSP	
	N	PCRI ± SE	N	PCRI ± SE
Tiger[*] (*Panthera tigris*)	153	0.86 ± 0.17	458	2.40 ± 0.23
Large prey				
Gaur (Bos gaurus)	16	0.15 ± 0.08	50	0.25 ± 0.07
Sambar[*] (*Rusa unicolor*)	6	0.04 ± 0.02	546	3.10 ± 0.42
Southern serow[*] (*Capricornis sumatraensis*)	21	0.13 ± 0.04	0	0.0
Medium prey				
Wild pig[*] (*Sus scrofa*)	646	4.24 ± 0.46	1289	6.78 ± 0.68
Muntjac[*] (*Muntiacus muntjak*)	1119	6.73 ± 0.77	4251	25.02 ± 2.14
Potential small prey				
Banded leaf monkey[#] (*Presbytis femoralis*)	45	0.21 ± 0.12	4	0.02 ± 0.01
Malayan porcupine (*Hystrix brachyura*)	166	0.94 ± 0.29	164	0.98 ± 0.19
Pig-tailed macaque (*Macaca nemestrina*)	257	1.78 ± 0.29	263	1.52 ± 0.18
Great argus pheasant (*Argusianus argus*)	344	2.38 ± 0.90	332	1.92 ± 0.40
Mouse deer[a] (*Tragulus* sp)	37	0.22 ± 0.08	81	0.37 ± 0.12
Brush-tailed porcupine (*Atherurus macrourus*)	21	0.14 ± 0.06	23	0.12 ± 0.05
Other large mammals				
Asian tapir (Tapirus indicus)	256	1.67 ± 0.22	274	1.59 ± 0.38
People				
Indigenous people (*Orang Asli*)	147	0.89 ± 0.21	187	0.87 ± 0.19
Army	0	0	61	0.22 ± 0.12
Foreign encroachers	2	0.41 ± 0.41	44	0.27 ± 0.06
Local encroachers	2	0.01 ± 0.01	0	0.0
Vehicles[*]	72	0.46 ± 0.16	0	0.0
Other humans[b]	6	0.05 ± 0.02	27	0.14 ± 0.07

[a] Both mouse deer species (lesser and greater mouse deer) were pooled together due to difficulty in differentiating between the species from photos.
[b] Includes: tourists, loggers, and government staff involved in managing the area.
[#] Camera-traps are not an optimal method for detecting arboreal species such as banded leaf monkeys.
[*] Significant differences based on Wald test (critical value at $\alpha = 0.05$).

supported the assumption that the two sub-populations were demographically closed during the study (TFR: $z = -1.56$, $P = 0.06$; and, RBSP: $z = -0.97$, $P = 0.16$). The SECR produced a density estimate ($\widehat{D} \pm$ SE; adult tigers/100 km^2) of 0.61 ± 0.31 (TFR) and 1.95 ± 0.48 (RBSP), which were significantly different (Wald test: critical value at $\alpha = 0.05$; $P < 0.05$). The 95% density estimate ($\widehat{D} \pm$ SE; adult tigers/100 km^2) confidence intervals for TFR and RBSP were (0.23–1.56) and (1.21–3.15) respectively, corresponding to an estimated adult tiger abundance of 9 (3–23) in TFR and 24 (15–39) in RBSP.

3.3. Tiger corridor identification in BT-SLF

From 651 km surveyed over three sampling occasions, tiger presence was detected in 30 of the 156 1×1 km grid cells. The constant model for detection probability ($p = 0.16 \pm 0.05$, SE) was the top ranked model and used in the subsequent habitat suitability analyses (Table 2). The estimated tiger detection probability (±SE; 95% CI) was 0.16 (±0.05; 0.07–0.28) and there was evidence of over-dispersion in the data ($p = 0.15$, $\hat{c} = 15$). The top ranked model was found to be affected by spatial autocorrelation (Moran's $I = 0.02$, $P < 0.01$). The quasi-likelihood over-dispersion parameter (QAICc) for model selection showed that tiger habitat use was negatively influenced by elevation ($\hat{\beta} \pm \mathrm{SE} = -0.52 \pm 0.29$; Table 3) as this covariate had the highest percentage of relative summed model weight (54.7%) compared to the other covariates (NDVI; 27.1%, Settlement; 31.4% and Road; 31.9%). Collectively, the results indicated that tigers primarily preferred sites at lower elevation.

The predictive tiger habitat model identified 14.7% of the cells with 'Very high' ($\psi = 0.78$–0.65) suitability and 34.0% of the cells with 'High' (0.64–0.51) suitability for a corridor. From their geographical spread, eight cells in the west and seven cells in the east each connected RBSP–TFR and were deemed as suitable locations for potential viaduct placements (Fig. 2). As about 80% of the two committed land developments on the west end of the study area overlap the proposed core corridor habitat and therefore these development projects were proposed to be relocated elsewhere (Fig. 2).

These findings and a set of accompanying management recommendations were presented to 23 Federal and State authorities. Four meetings and two workshops were held over 26 months so that the study could continually inform the policy drafting process until the state land forest was gazetted as Amanjaya Forest Reserve (Perak State Government Gazette, No. 786) on 9 May 2013. This new status protected the proposed forest corridors from further conversion and allowed the government to finalise plans for constructing a viaduct. The final Permanent Reserved Forest corridor covered 100% of the recommended area.

4. Discussion

The effect of two distinctly different forest management regimes, particularly their wildlife protection measures, on tigers in Malaysia was dramatic. Tiger density was three fold higher in the primary forest of RBSP than in the logged forest of TFR, which was related to a significantly higher relative abundance of several preferred prey species in RBSP, especially sambar. While poaching by snares appeared to be low in both study areas, with only a few snare traps recorded, an independent study from a similar period recorded about 140 tiger and focal prey species individuals as being annually poached from the wider Belum-Temengor landscape (TRAFFIC, unpublished data). As our field surveys did not include any dedicated anti-poaching patrols, the information from TRAFFIC is considered to better reflect the poaching pressure

Table 2
Tiger detection probability (p) models ($w_i > 0$), with ψ (NDVI + Settlements + Elevation + Road), in the Belum Temengor State Land Forest.

No	Candidate models for detection probability with a global (ψ) model	K	ΔAICc	w_i
1	p(.)	6	0.00	0.387
2	p(Rain)	7	1.04	0.229
3	p(Distance walked)	7	1.71	0.164
4	p(Rain + distance walked)	8	2.91	0.090
5	p(Observer)	11	3.96	0.053
6	p(Observer + rain)	12	4.67	0.037
7	p(Observer + distance walked)	12	5.61	0.023
8	p(Observer + rain + distance walked)	13	6.54	0.015

Note: ΔAICc = difference in AICc values between each model and the model with the lowest AICc. w_i = AICc model weight, K = number of parameters within the model.

Table 3
Tiger habitat use (ψ) models, with p(.), in the Belum Temengor State Land Forest.

No	Model	K	ΔQAICc	w_i	Evidence ratio
1	ψ(Elevation)	3	0.00	0.174	1.00
2	ψ(.)	2	0.05	0.169	0.97
3	ψ(Elevation + road)	4	1.29	0.091	0.52
4	ψ(Elevation + settlements)	4	1.49	0.083	0.48
5	ψ(Settlements)	3	1.82	0.069	0.40
6	ψ(Road)	3	1.90	0.067	0.39
7	ψ(Elevation + NDVI)	4	1.92	0.066	0.38
8	ψ(NDVI)	3	2.12	0.060	0.35
9	ψ(Elevation + road + settlements)	5	2.63	0.046	0.26
10	ψ(Elevation + NDVI + road)	5	3.25	0.034	0.20
11	ψ(Elevation + NDVI + settlements)	5	3.27	0.033	0.19
12	ψ(Road + settlements)	4	3.62	0.028	0.16
13	ψ(NDVI + settlements)	4	3.92	0.025	0.14
14	ψ(NDVI + road)	4	4.00	0.024	0.13
15	ψ(Elevation + NDVI + road + settlements)	6	4.48	0.019	0.11
16	ψ(NDVI + Road + settlements)	5	5.75	0.009	0.05

Note: ΔQAICc = difference in QAICc values between each model and the model with the lowest AICc. w_i = AICc model weight, K = number of parameters within the model.

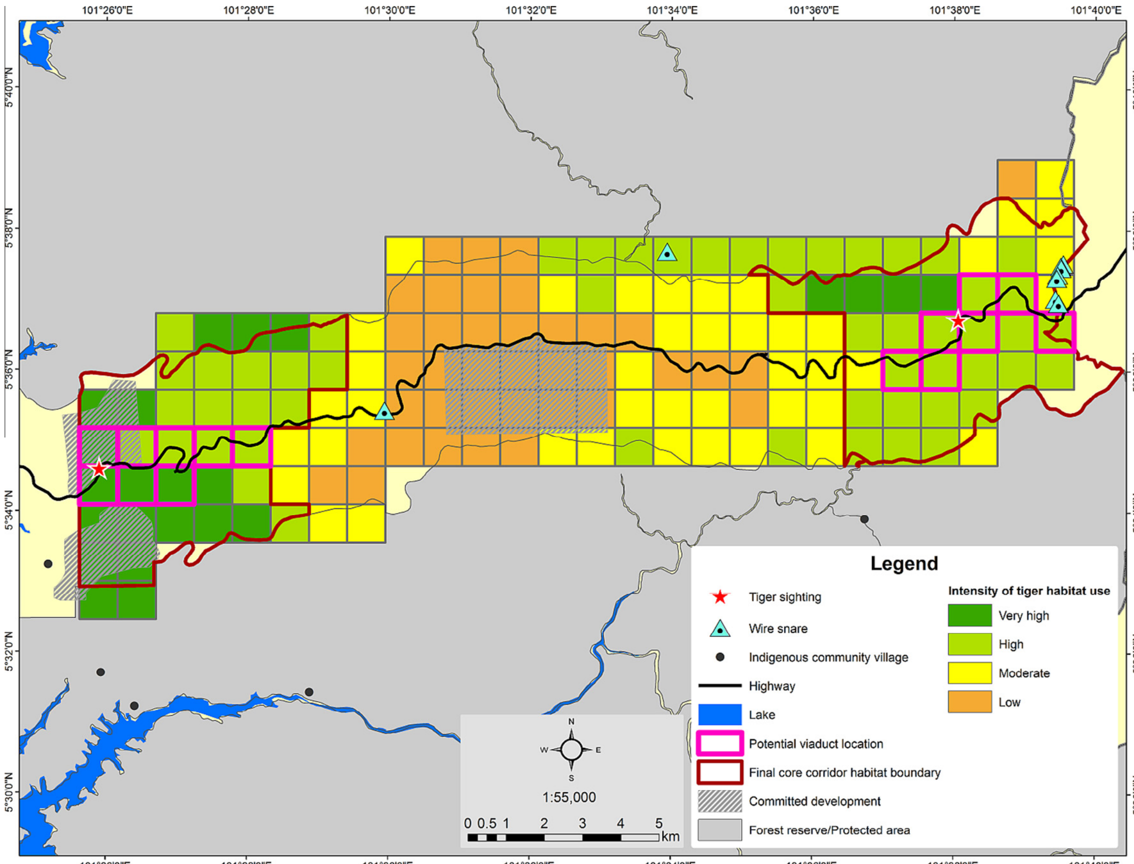

Fig. 2. Final core corridor habitat boundary and potential viaduct locations for tigers in the Belum Temengor State Land Forest.

within the study sites. Nevertheless, whether due to differences in poaching or vegetation characteristics all findings suggest that TFR did not provide suitable conditions that would enable a tiger population to flourish in. This is cause for concern because 85% of the Malaysian tiger population is known to occur in Forest Reserves (FRs) (Kawanishi et al., 2003) and its future greatly depends on them being properly managed for wildlife, which would include specific measures for tackling poaching through a robust law enforcement strategy.

Wildlife protection and accessibility markedly differed between the two study areas. RBSP had an average capacity of 1.45 forest rangers/100 km^2, an army checkpoint through the lake and no access through roads, whereas TFR had no anti-poaching patrols, no check point through the lake and direct access through unmanned logging roads (WWF-Malaysia, 2011). The relatively high forest use by *Orang Asli* in both study areas seemed to cause no apparent disruption to tigers or their prey. Presumably, this is due to their limited impact on the forest through extracting non-timber forest products, fishing and hunting of primates, small mammals and birds using blowpipes (Aziz et al., 2013; Azrina et al., 2011). A higher number of foreign encroachers were recorded inside RBSP, most likely searching for agarwood, who are also thought to opportunistically poach forest ungulates for sustenance (Lim and Noorainie, 2010). Based on encroachment signs such as litter left behind and tree marking signs as well as past records (Abdul Kadir, 1998), these foreigners are likely from Thailand, Laos, Cambodia and Vietnam. Overall poaching using snare traps was low in both study areas. Yet despite this, an independent assessment that relied on data provided from local informants found a starkly different situation.

Using a well-established local informant network, TRAFFIC conducted a widespread survey of the main poachers operating in and around the Belum-Temengor Forest complex from January 2009 to December 2011. Over three years, 20 adult tigers, 64 sambar and 326 muntjac were recorded as having been poached. The majority of poachers (50%) stated that these animals came from TFR, then surrounding FRs (39%) and then RBSP (11%; TRAFFIC, unpublished data). Thus this information coupled with evidence on the use of firearms in TFR suggests a higher hunting pressure in TFR and other FRs than in the better protected and less accessible RBSP. Although we were unable to independently verify the TRAFFIC findings, our tiger abundance estimates would fit the poaching pattern described. It raises the question of how tigers are faring in the Malaysia's other FRs, as no recent (after 2009) studies on tiger population status exist for these.

The poaching findings, add extra importance to maintaining connectivity between RBSP and TFR. Based on the extrapolation of the SECR population size range (95% CI: 20–70 adult tigers), the larger Belum-Temengor Forest Complex (encompassing 3385 km^2) may harbour at least 15% of Malaysia's estimated 250–340 tigers (DWNP and MYCAT, 2014). However, the distinct differences in tiger density estimates imply that TFR may represent a sink population in which its connectivity to the source population of RBSP is of critical importance. It is likely that strict trans boundary protection by Thai paramilitary border police in the Hala-Bala Wildlife Sanctuary directly adjacent to RSBP, in Thailand (Lynam, 2005) as well as patrolling by the Malaysian army on the fenceless border has served as a deterrent to poachers. This has possibly contributed to safeguarding tigers and their prey as well as aided in ensuring that RBSP remains as an important source population for tigers. Although our extrapolation is scientifically defensible based on the representation of the proportion of floristic habitats within each study area and indicates that protected areas may have higher densities compared logged forests, this generalisation may not hold true across other sites due to uneven anthropogenic differences that may not be accounted for such as higher poaching pressure at the edges of the forest.

In the BT-SLF area, tigers preferred lower elevation forest, as this is most likely preferred by its prey (Lynam et al., 2012; Steinmetz et al., 2008). In Chitwan National Park, Nepal, displaced tigers or transients often occupied edge environments (Sunquist, 1981). Tigers in the BT-SLF may therefore have included transients, displaced individuals or even resident females with cubs that predominantly focused activity budgets on maximising hunting opportunities. This appears to be supported by camera trap data collected several months later (May–August 2011) that recorded three females, with at least two cubs each, using the BT-SLF (WWF-Malaysia, unpublished data). Further, our sampling design for corridors involved a substantial indirect sign survey effort to account for imperfect detection that produced sufficiently high levels of precision, psi(SE) < 0.18, and sufficient information for guiding government policy development. As the government had allocated USD18.8 million for viaduct construction and management, our relatively inexpensive corridor surveys provided timely and reliable information.

Wildlife crossings and corridors have been previously identified by using expert opinion, GIS-based least-cost path analysis, resource selection functions (RSFs), habitat suitability (Chetkiewicz and Boyce, 2009; Clevenger et al., 2002) and, more recently, GPS-telemetry data and RSFs analysis under a Bayesian framework (Colchero et al., 2011). RSFs are defined as any function that is proportional to the probability of use of a resource unit (Manly et al., 2002). The occupancy modelling approach used in our study is a form of RSF analysis (as sampling units are treated as resource units). However, it differs from many of the above approaches by explicitly accounting for unequal detection probabilities (Gu and Swihart, 2004; MacKenzie, 2006; Sunarto et al., 2012). Owing to the sampling design of adjoining 4 km^2 grid cells and the high mobility of tigers, habitat use by tigers was spatially autocorrelated (i.e. the state of habitat use at a given site is influenced by the habitat use state of the neighbouring sites). Thus the resulting interpretation of the habitat use of tigers from this study is considered biologically meaningful but needs to be treated with caution. The presence of spatial autocorrelation and the non-inclusion of other important covariates that could not be included but are likely influential in describing the habitat use of tigers such as prey abundance should be noted. On the whole, the findings in our study offer a new approach to overcome data collection and modelling limitations with respect to resource selection in identifying fine-scale corridors and viaduct placements especially when telemetry-based movement data are not available.

In Malaysia's Main Range, which represents the largest (c. 17,900 km^2) of the three priority landscapes for tiger, there are already at least 12 main roads of which seven bisect large swathes of intact forest. Plans to expand and widen existing roads have been detailed in the country's first extensive spatial planning blue print (DTCP, 2005). This is therefore likely to negatively affect the localised population dynamics of tigers (Kerley et al., 2002; Linkie et al., 2006, 2008) through restricted movement for tigers and increased access for hunters. Hence, apart from applying mitigation measures that minimise road fragmentation effects, it is critical that robust measures for protecting tigers be adopted and implemented in Malaysia's FRs especially in lower elevation forest (<1000 m a.s.l.) as indicated from our study.

The spatial modelling and government engagement used in our study would go a long way to supporting spatial planners and decision makers to incorporate wildlife concerns into their future development plans across the tropics. For Malaysia, this was recently bolstered by research findings that brought about a hunting moratorium on tiger prey (Kawanishi et al., 2013). These approaches, whilst integral to a comprehensive tiger conservation strategy, will not succeed unless specialist anti-poaching patrols are conducted regularly. Our study is the first to assess tiger and prey population status inside a FR and was revealing because biological monitoring teams were unable to detect a principal threat (poaching), despite their putative widespread presence in the field. Yet, if TFR is representative of other FRs in Malaysia then the supposedly high tiger and prey poaching, supported by a low tiger density, would point to an impending extinction crisis because none of these presumed tiger strongholds have active tiger law enforcement units. Future efforts to constructively engage government agencies must be focused here. In addition to this, it is imperative that selectively logged forests should not be considered as 'degraded' habitat and thereby used as an excuse for it to be converted to monoculture timber plantations (Aziz et al., 2010). We urge against the establishment of monoculture plantations in FRs and state land forest corridors within Peninsular Malaysia especially since the status of tigers and other endangered species remains largely unknown in these forest habitats.

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