Exploring potential range connectivity of sun bear (Carnivora: Ursidae: Ursinae)

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Abstract. In this era of deforestation and human-dominated landscapes, fragmentation disrupts once continuous wildlife ranges into unnatural subpopulations. In the tropics, where habitat is changing rapidly, there are limited data on activity patterns of large mammals and on their tolerance to habitat disturbance. Therefore, to inform habitat and wildlife management, biologists must work with imperfect datasets in innovative ways. I explored potential range connectivity for the forest dependent sun bear, using density of tree cover (%) as a proxy for habitat condition to classify land area into non-habitat, and marginal, sub-optimal, and core habitat. Potential range fractures were visualised using non-habitat, habitat quality, and severity of human disturbance. Global sun bear range was divided into seven potential subpopulations, within which habitat fragmentation occurs in a continuum of severity. Almost 90% of core sun bear habitat fell outside protected areas, and large areas of core habitat fell in areas where sun bear are considered Extirpated by the IUCN. Habitat fragmentation likely restricts sun bear from functioning as continuous populations; instead, subpopulations may require inter-area movements, either naturally or through human-assisted translocation. The resulting maps can be used as a visual guide for those researching and managing sun bears in order to steer priorities. Further ground studies will determine if the habitat classifications and assumptions in this analysis are justified.

Key words. *Helarctos malayanus*, habitat fragmentation, remote sensing, tree density, habitat selection

INTRODUCTION

Human-caused forest fragmentation is one of the biggest threats to terrestrial biodiversity (Tilman et al., 1994; Fahrig, 2003). Interior forest has diminished globally in the past few decades and more than 70% of the worlds' forest now occurs within 1 km of the forest edge (Wade et al., 2003; Haddad et al., 2015). Habitat fragmentation often increases rates of mortality and extirpation in forest dependent wildlife, due to cumulative and interacting factors related to the short and long-term impacts of genetic and demographic isolation (Krause et al., 2008; Staddon et al., 2010; Riitters et al., 2016). Understanding a species-level response to habitat fragmentation is often hindered by lack of data on key parameters, such as dispersal behavior and tolerance to sub-optimal habitat and non-habitat. Data are especially scarce within the tropical forests of Southeast Asia, where many endangered species are undergoing rapid declines, and where resources to research, manage and conserve species are typically limited (Sodhi et al., 2010).

An example of a species affected by this dilemma is the forest dependent sun bear *Helarctos malayanus*, for which two sub-species exist, one distributed across mainland Southeast Asia, small parts of adjacent China, India and Bangladesh, and the island of Sumatra (*H. m. malayanus*), and the second on the island of Borneo (*H. m. euryspilus*; Meijaard, 2004). The sun bear is estimated to have declined globally by between 30–50% over 30 years, largely due to habitat loss and poaching (Scotson et al., 2017a, b). The extent of the sun bear range was once almost continuous (Erdbrink, 1953). However, Southeast Asia has experienced some of the highest rates of deforestation globally, and habitat fragmentation is considered a major threat to sun bear (Miettinen et al., 2011; Hansen et al., 2013; Margono et al., 2014; Scotson et al., 2017a). Other bear species have been assessed on a subpopulation level, including brown bear *Ursus arctos*, Andean bear *Tremarctos ornatus* and Polar bear *Ursus maritimus* (Kattan et al., 2004; Wiig et al., 2015; McLellan et al., 2016), but this has yet to be attempted for sun bear (Scotson et al., 2017a).

Subpopulations — geographically isolated groups between which there is little demographic or genetic exchange (IUCN, 2017a) — are an important entity to monitor given the cumulative threats associated with population fragmentation. Assessing the connectivity of sun bear range faces several methodological problems because data scarcity rules out traditional data-intensive methods. Movement patterns of sun bear have been monitored in very few individuals relative to American and European bears (McLellan & Hovey,

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2001; Dahle & Swenson, 2003; Kendall et al., 2008). Radio collaring has proven to be labor intensive, with trapping effort high relative to success rates; only 14 wild sun bears have been monitored with radio collars, in two sites in Malaysia (Normua et al., 2004; Wong & Servheen, 2004; Cheah, 2013), one in Indonesian Borneo (Fredriksson, 2012). Use of DNA from scat and hair samples has worked with bears in temperate climates (e.g., Kendall et al., 2008; Swaisgood et al., 2016), but less so with sun bear populations in the tropics, where scat studies are limited, and hair snaring so far unsuccessful (Fredriksson, 2012; Ngoprasert et al., 2015). Consequently, sun bear home range, natal dispersal, and tolerance to moving through non-forested areas are poorly understood. Moreover, limited resources (i.e., time, money) and limited dedicated sun bear research make it unlikely that such data will become available soon. With the rapid rate of forest cover change, habitat loss and fragmentation, and increasing human populations within sun bear range, it is urgent to use what data are available to identify sun bear subpopulations and to steer priorities for broadscale population management.

Globally available high-resolution tree cover data (Hansen et al., 2013) have advanced the spatial understanding of estimated sun bear range extent, and the understanding of how sun bear distribution responds to changes in tree cover (Scotson et al., 2017b). By using tree cover as a proxy for habitat suitability, I investigated probable habitat connectivity within the global sun bear range with the following objectives: 1) create a habitat suitability index using the modelled relationship between sun bear occurrence and % tree cover, 2) identify areas of non-habitat and evaluate other factors that may restrict movement within sun bear range, and 3) assess the connectivity of global sun bear range in terms of structural connectivity and the potential for demographic and genetic movement. Finally, to improve understanding of the threats posed by fragmented habitat to sun bear at the species and subpopulation level, I suggest a number of site-level priorities for research and management of sun bear subpopulations and habitat. This analysis is a hypothesis on the potential connectivity (and fragmentation) of sun bear range, intended to steer future research and to generate baseline information that can be improved upon once the appropriate ecological data become available.

MATERIAL & METHODS

Investigating the potential connectivity of the sun bear landscape using existing data sources required several almost arbitrary decisions — drawing from existing knowledge where possible — on viable patch size and the distance that sun bear can move through non-habitat. To create a map of the current connectivity status of sun bear range, I created a habitat suitability index from the previously modelled relationship between sun bear presence and tree cover (Scotson et al., 2017b), removing patches of forest assumed to be not viable for supporting a breeding bear population. Maps were visually inspected to identify possible subpopulations

and to qualitatively evaluate habitat connectivity between and within subpopulations. Finally, I quantified the spatial extent and composition of habitat, compared viable habitat with the IUCN sun bear range map, and calculated the extent that fell inside protected areas within IUCN categories I–V (IUCN, 2017b).

Tree cover processing. This analyses used two satellite-based tree cover rasters that were available at the time of writing from open source Global Forest Watch; i) tree cover for the year 2000 (pixels valued from 0 to 100% tree cover), and ii) tree cover loss from between 2000 and 2014 (pixels of loss rasters were valued 1 [100% loss of tree cover within pixel] or 0 [no loss]) (www.globalforestwatch.org, accessed 14th Feb 2017). Tree cover (%) was any vegetation (nature and non-natural) above 5-m height, and reflects differences in habitat assemblages, being highest in tropical evergreen forests (> 70%), lower in secondary degraded forest, and lowest in dry dipterocarp forest. Tree cover within non-natural vegetation (e.g., agriculture, rubber, and palm oil plantations) usually falls below 20% (Scotson et al., 2017b). Rasters were trimmed to the geographic extent of sun bear range within the last 500 years (Erdbrink, 1953), excluding China, where the sun bear is thought to be almost extirpated and it's not clear how far north sun bear range once extended (Scotson et al., 2017a). A tree cover raster for 2014 was created by masking out tree cover that was lost by 2014 by i) multiplying all loss pixels by 100 to transform pixel values of 1 to become 100 and on the same scale as the tree cover layer and ii) subtracting the transformed loss raster from the year 2000 tree cover raster. All negative values, when 100% loss was subtracted from a cell with < 100% tree cover, were transformed to zero. Tree cover gain was not incorporated (also available from Global Forest Watch), because it includes an unknown amount of planted forest, which tends to be intensively managed single species plantations (e.g., eucalyptus, teak, rubber; Keenan et al., 2015) assumed to be not viable for sustaining sun bear populations (however, sun bears may move through some types of non-natural vegetation when it is contiguous to patches of natural forest [Normua et al., 2004; Guharajan et al., 2018]).

Creating a Habitat Suitability Index. To create a Habitat Suitability Index, I first smoothed the 2014 tree cover raster to the same scale used by Scotson et al., 2017b (6 km^2), and used the slope coefficient from their model to calculate the relative probability of sun bear occurrence in each map pixel in the year 2014 of > 20% tree cover:

$$p[i] = \exp(\ *\ \%\ \text{tree cover})/\exp(\ *\%\text{tree cover})$$

All areas < 20% tree cover were marked as zero, because sun bears were never detected within this level of tree cover (Scotson et al. 2017b). To smooth out patches of < 20% tree cover areas that were considered unlikely to restrict sun bear movement, I broadened the scale to that of the mean home range size of sun bear by averaging all relative probability values over a 11 km^2 circular area (Nomura et al., 2004; Fredriksson, 2012; Cheah, 2013).

To distinguish habitat quality, I created a habitat suitability index by grouping all relative probabilities into four categories, non-habitat (assumed not suitable for sun bear), marginal (low relative probability of use), sub-optimal (moderate relative probability of use), and core (high relative probability of use). The values were grouped using Natural Jenks classification in ArcGIS 10.2. Natural Jenks minimised variability within classes and maximised variability between classes (North, 2009). All of these probabilities were assigned as if hunting and other non-habitat actors were not constricting current sun bear use of the landscape; in parts of the species' range, notably Lao PDR and Vietnam, there are sizeable areas of prime sun bear habitat from which the species has been extirpated by recent industrial hunting. It was assumed that non-habitat restricts sun bear movement to some degree, as it does with other bear species (e.g., Proctor et al., 2012), except for when moving through short distances (i.e., distances less than average daily movement) between patches of habitat. Sun bear may or may not travel through marginal habitat, depending on the composition of that habitat (i.e., plantations versus secondary shrub); this remains unknown until better GIS land cover data become available. Sub-optimal and core habitat was expected to be largely natural forest.

Assessing the connectivity of the sun bear landscape. Forest patches smaller than 16 km² (the average size of a male sun bear home range (Normua et al., 2004; Wong & Servheen, 2004; Cheah, 2013) were considered not viable for sustaining sun bear; a male sun bear home range can overlap several female home ranges, therefore a patch > 16 km² could theoretically support a breeding population. Patches were also classed as non-viable when smaller than 100 km² and more than 4.5 km away from an adjacent patch (i.e., the maximum recorded daily movement of sun bear, Fredriksson, 2012); using 4.5 km this as a measure of how far a sun bear might be able to move between patches of habitat through non-habitat (however this is guesswork in the absence of data on ability of bears to move through non-habitat, and does not consider the composition of the non-habitat). Finally, the remaining polygons were compared with the IUCN sun bear range map and areas where sun bear is known not to occur removed. Some patches of tree cover that the IUCN classified as Extirpated were not removed, when they occurred adjacent or contiguous to known range (assessed visually) and could theoretically be used by bears (i.e., between 16–100 km² and within 4.5 km of a neighboring viable patch, or > 100 km²). To test the sensitivity of this analysis to how far a sun bear was assumed to be able to travel through non-habitat, I repeated the process of identifying non-viable habitat using the median daily movement (2.3 km) and maximum daily movement × 2 of sun bear (9 km; Wong & Servheen, 2004; Fredriksson, 2012; Cheah, 2013).

Validating the habitat classifications. To check if the classification of habitat met with expectations, I compared the resulting categories with the most up to date data available on landcover and human influence levels. I generated a set of 15,000 random points within non-habitat, and another set of 15,000 random points within habitat, divided equally between marginal, sub-optimal, and core (i.e., 5,000 points in each). At each random point, I calculated i) the percent tree cover in 2014, ii) the relative probability of bear occurrence (based on tree cover), iii) Human Influence Index (Sanderson et al., 2002), and iv) Land Cover according to the Tropical Ecosystem Environment Observations by Satellites (TREES; Stibig et al., 2003). I evaluated the dissimilarity of Human Disturbance Index values between the habitat and non-habitat categories with density plots, simple linear regression, and a Welch Two Sample t-test. I used frequency plots to compare the distribution of TREES land cover classification values within the mosaic habitat classes.

Identifying subpopulations and patches most threatened by fragmentation. I overlaid the habitat mosaic with i) areas of high Human Influence Index (defined as areas with a higher than average value of Human Influence in non-habitat), and ii) the road network (https://urs.earthdata.nasa.gov, accessed 16 February 2017); both assumed to reduce or prevent sun bear movement. Possible subpopulations, and 'At Risk' areas within those subpopulations that may be vulnerable to becoming isolated, were identified by visually inspecting all core and sub-optimal habitat patches within the context of the surrounding landscape features that may restrict movement. 'At Risk' areas were areas of core and sub-optimal habitat appearing, on visual inspection, to be highly fragmented within a mosaic of concentrated potential barriers movement. To quantify the structure of habitat versus non-habitat and the composition marginal, sub-optimal, and core habitat, I calculated the proportion of each within each potential subpopulation. I also calculated the areas of each habitat class that fell within protected areas.

Data were analysed in ArcGIS 10.2.4 and R Version 3.2.2 (R Core Team, 2016).

RESULTS

Landscape metrics. Varying the distance used to identify non-viable habitat patches (i.e., how far a sun bear may be able to move through non-habitat) had little effect on the total area of viable habitat; 2.3 km = 2,320,138 km² (mean = 6,057, SD = 70,049), 4.5 km = 2,324,799 km² (mean = 4,485, SD = 60,938), and 9 km = 2,327,333 km² (mean = 4,040, SD = 57,191). All subsequent results are based on the middle value of 4.5 km (maximum recorded daily movement of a sun bear). Therefore, viable habitat is above 20% tree cover, with a patch size > 16 km², or between 16–100 km² and closer than 4.5 km to the nearest viable patch.

In 2014, 55% of viable sun bear habitat was classified as core, 27% as sub-optimal, and 18% as marginal. Only 18% of forest classed as viable habitat fell within protected areas, the bulk of which was classified as sub-optimal habitat (65%) with a smaller amount of core (12%) and the remainder being marginal (6%) and non-viable habitat (16%). Almost 90% of core sun bear habitat fell outside protected areas.

Validating the habitat classes. Comparing values of the Human Influence Index and TREES land classification within habitat (all classes grouped together) and non-habitat supported the assumption that area classified as viable habitat was different from non-habitat. Human Influence Index values were on average 13.8 points higher in areas classified as non-habitat ($t = -95.2$, df = 29658, $p < 0.001$, $\bar{x}$ within non-habitat = 36.7, SD = 13.1, $\bar{x}$ within habitat = 23, SD = 11.8). When marginal habitat was grouped with non-habitat, the average Human Influence Index was on average 14.2 points higher than sub-optimal and core grouped together ($t = -100.3$, df = 24971, $p < 0.001$, $\bar{x}$ within non-habitat & marginal grouped = 34.6, SD = 13.5, $\bar{x}$ within sub-optimal and core grouped = 20.4, SD = 10.4). Bear habitat was heavily skewed to lower Human Influence Index values (< ~40), however it fell to some extent within all values of Human Influence (Fig. 1). TREES land classification values within non-habitat were more often classified as non-viable bear habitat (i.e., cropland, shrub, bare land, rock). Second to non-habitat, marginal habitat had the highest proportion of points within non-viable bear habitat categories. In sub-optimal and core habitat, land classification tended to be
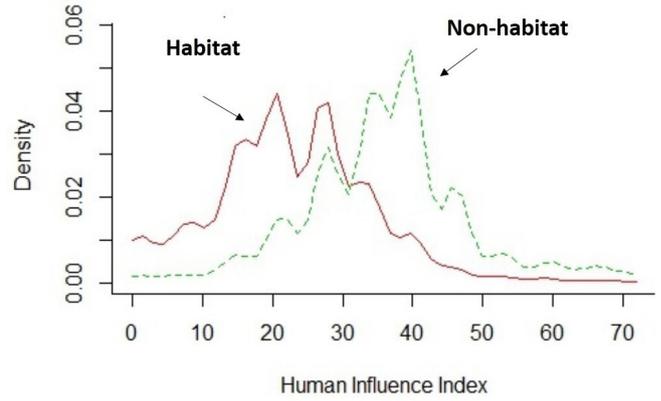

Fig. 1. Density plots of Human Influence Index values within areas classified as habitat and non-habitat within sun bear range. Human Influence Index values (Sanderson et al., 2002) were calculated to 30,000 random points generated equally within areas of non-habitat and habitat. Human Influence Index values were on average 13.8 points higher in areas classified as non-habitat ($t = -95.2$, $df = 29658$, $p < 0.001$, $\bar{x}$ within non-habitat = 36.7, SD = 13.1, $\bar{x}$ within habitat = 23, SD = 11.8) supporting our assumption that habitat is different from non-habitat.
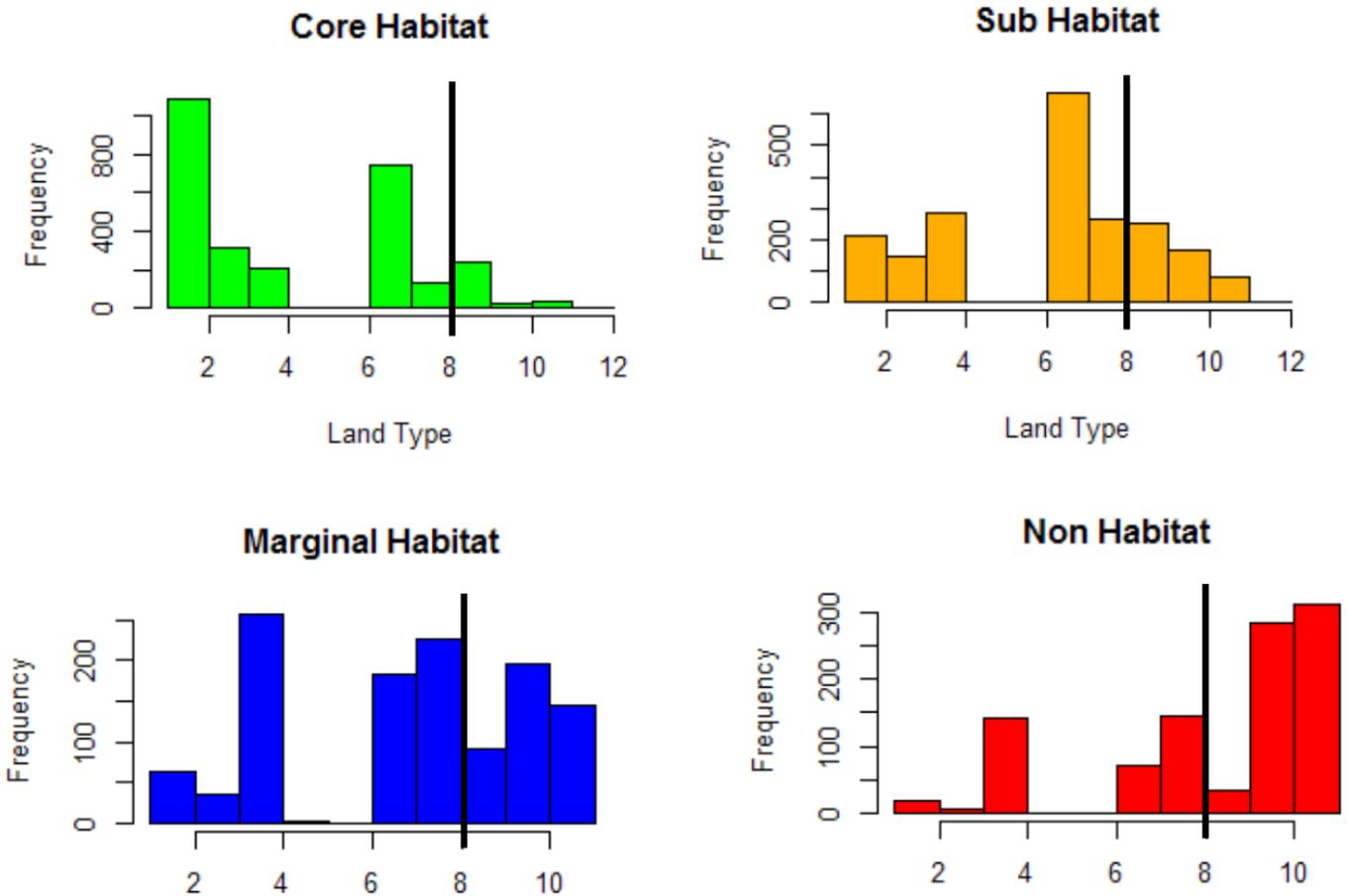

Fig. 2. Frequency plots of land type values from the Terrestrial Ecosystem Environment Observation by Satellites (TREES; Stigbig et al., 2003). TREES land type values were calculated to 20,000 random points generated equally between non-habitat, marginal, sub-optimal, and core habitat. Land type was reclassified into 12 categories; 1–8 are categories in which bear use has been previously detected (1–3 = evergreen; 4 = deciduous, woodland; 5 = mangrove; 6 = swamp, woodland; 7–8 = mosaic of woodland, secondary, evergreen and cropland), and 9–12 are considered non-habitat (9 = cropland, shrub; 10 = cropland, bare land; 11 = rock, limestone; 12 = water). To correct for errors on the TREES map due to deforestation since 2000, % tree cover in 2014 was extracted for each random point, and points with no tree cover in 2014 reclassified as falling in non-viable habitat. TREES land classification values within non-habitat were more often classified as areas considered as non-viable bear habitat (i.e., cropland, shrub, bare land, rock). Second to non-habitat, marginal habitat had the highest proportion of points within non-viable bear habitat categories. In sub-optimal and core habitat, land classification tended to be areas of potential bear habitat (i.e., evergreen, deciduous forest, and other forms of mosaic forest).
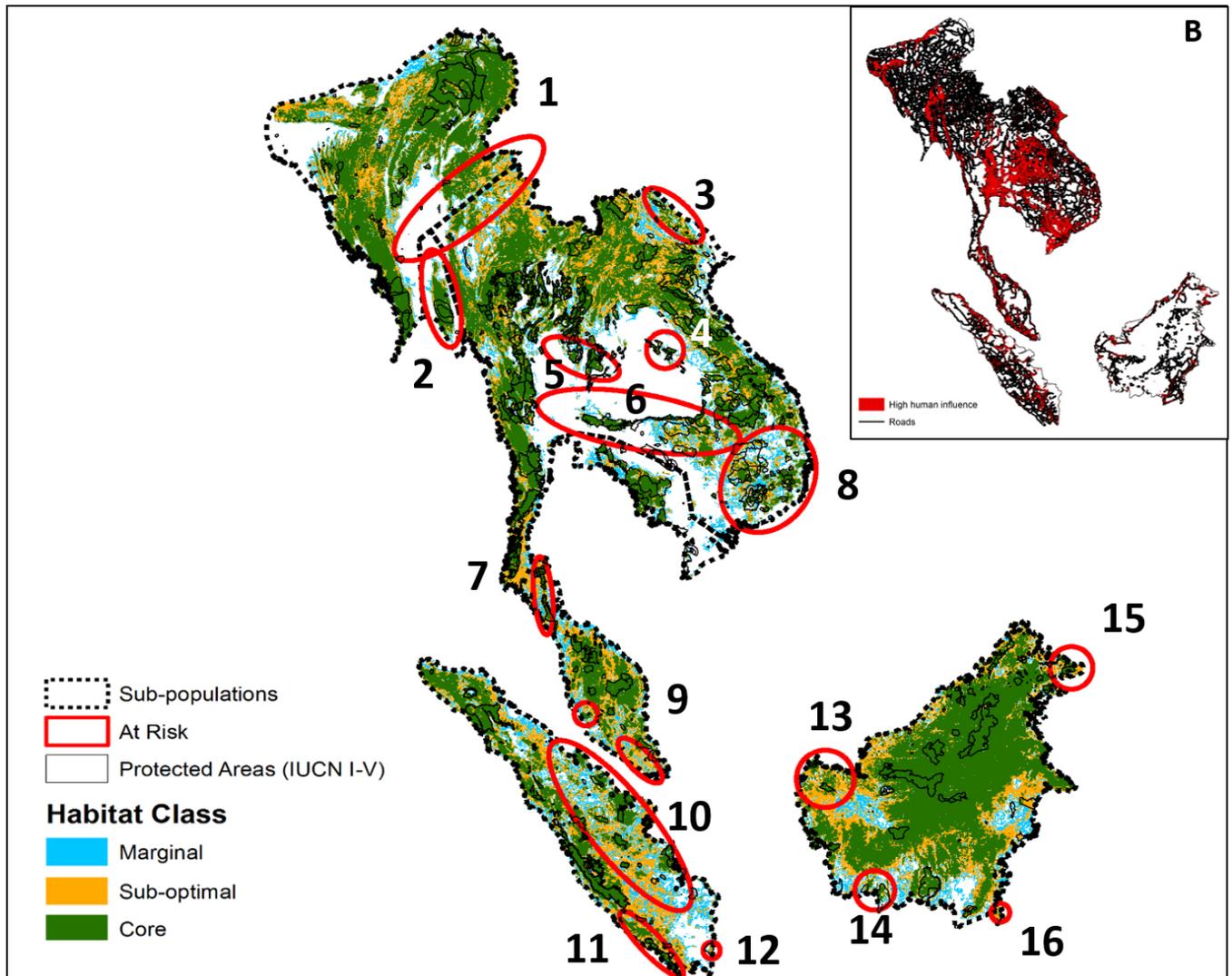
Fig. 3. Sun bear landscape fragmentation and connectivity in Southeast Asia, India and Bangladesh. A) Core and sub-optimal contiguous range is assumed to positively impact bear movement (i.e., connectivity), although dependent on associated levels of human influence and roads. Visual analysis identified seven potential subpopulations of sun bears; i) northern Mainland, ii) Central Myanmar, iii) Central SE Asia, iv) South-central SE Asia, v) Thai-Malay peninsula, vi) Sumatra, vii) Borneo (divided by dashed lines). Within these potential subpopulations there were many 'At Risk' areas where sun bears may be vulnerable to becoming isolated due to potential barriers to movement including habitat fragmentation, high human influence and roads (identified by red ovals and numbers 1–16 correspond with the IDs listed in Table 2). B) High Human Influence and road network are assumed to be significant barriers to bear movement across the sun bear landscape.

areas of potential bear habitat (i.e., evergreen, deciduous forest, and other forms of mosaic forest; Fig. 2).

Sun bear subpopulations and areas most at risk from isolation. The sun bear range is broken up naturally by ocean into three main populations: mainland Southeast Asia including Peninsular Malaysia, the island of Sumatra, and that of Borneo. A mosaic of core, sub-optimal, and marginal habitat creates potential habitat connectivity throughout much of the range. However, further population fractures seem likely, due to diminishing tree cover, high levels of human influence, and road networks. In total, global sun bear range was grouped into seven discrete blocks: i) northern Mainland, ii) Central Myanmar, iii) Central SE Asia, iv) South-central SE Asia, v) Thai-Malay peninsula, vi) Sumatra, vii) Borneo (divided by dashed lines in Fig. 3; Table 1). Within these seven population blocks, I identified 16 areas likely to be

at risk from further isolation where sun bear face further barriers to movement due to habitat fragmentation, human influence and roads (red ovals in Fig. 3; Table 2). This visual analysis is not considered exhaustive.

DISCUSSION

The sun bear was an ideal test case with which to assess range connectivity using tree cover. Existing research shows that the sun bear has large area needs, occurs in low densities, depends on forest for survival, and that density may be affected by changes in % tree cover (Wong & Servheen, 2004; Ngoprasert et al., 2015; Guharajan et al., 2018, Scotson et al., 2017b). These characteristics make the sun bear particularly vulnerable to the fast-changing tree cover across Southeast Asia. Key parameters required to quantitively measure range

Table 1. Potential sub-populations of sun bears identified by visually inspecting a landscape mosaic of suitable habitat in 2014 and potential barriers to bear movement.

| Subpopulation | Countries | Landscape metrics | | Habitat composition | | Barriers to movement (intra-pop/inter-pop)[2] | | | |
		Area (km²)[1]	% Range	% Core	% Sub-optimal	Road density	Human Influence[3]	Marginal	Non-Habitat
Northern Mainland	Bangladesh, NE India, northern Myanmar	364,599	18%	27%	73%	High/High	Low/High	Low/Low	Low/Low
Central Myanmar	Myanmar	680,032	<1%	40%	60%	High/High	Low/High	Low/Low	Low/High
Central SE Asia	South Myanmar, Lao PDR, Vietnam, Thailand, north Cambodia	14,425	33%	20%	80%	High/High	High/High	Low/Low	High/High
South-central SE Asia	South Cambodia, southwest Thailand	27,941	1.30%	28%	72%	Moderate/High	Low/High	Low/Low	Low/High
Thai-Malay Peninsula	South Thailand, Malay Peninsula	126,855	6%	43%	57%	Moderate/Moderate	Moderate/Moderate	Moderate/Low	Low/Low
Sumatra	Indonesia	265,302	13%	48%	52%	High/NA	Moderate/NA	Moderate/NA	Moderate/NA
Borneo	Malaysia, Indonesia, Brunei	598,032	29%	26%	74%	Low/NA	Low/NA	Moderate/NA	Moderate/NA

Footnotes

[1] Area of core and sub-optimal habitat

[2] Visually ranked by comparing the intensity of barriers to movement between sub-populations. Interpopulation areas surrounding Sumatra and Borneo is ocean and marked as NA.

[3] Visually ranks the intensity of areas above 32.7, which is the mean value of Human Influence within Non-habitat (32.7).

Table 2. Sun bear research priorities within areas perceived to be at the most risk from un-naturally fragmented habitat in Southeast Asia, India and Bangladesh in 2014.

| ID | Location | Status | Research Priorities | | | | |
			Confirm presence / absence	Population viability	Inter sub-population movement	Intra sub-population movement	Possible research Methods[5]
1	Sub-population divide between the northern-mainland and Central SE Asia	Unknown[1]	×			×	SS, I, G, T, CT
2	Central Myanmar Sub-population	Unknown[1,2]	×	×			SS, I, G
3	Northeast Vietnam range limit	Unknown[1,3]	×	×	×		SS, I, G, T, CT
4	Eastern Thailand	Confirmed[1]		×	×		T, G, CT
5	Central Thailand	Confirmed[1]		×	×		T, G, CT
6	Southern Thailand - northern Cambodia	Confirmed[1]		×	×		T, G, CT
7	Sub-population divide between southern Thailand and Peninsular Malaysia	Confirmed[1]				×	T, G, CT
8	Southern Vietnam	Mix of confirmed & unknown[1]	×	×		×	SS, I, G, T, CT
9	Peninsular-Malaysia (several)	Unknown[1,4]	×	×	×		T, G, CT
10	Eastern Sumatra	Confirmed[1]		×	×		T, G, CT
11	South-west Sumatra	Confirmed[1]		×	×		T, G, CT
12	Southern Sumatra	Confirmed[1]		×	×		T, G, CT
13	West Borneo	Confirmed[1]		×	×		T, G, CT
14	South Borneo	Confirmed[1]		×	×		T, G, CT
15	North-east Borneo	Confirmed[1]		×	×		T, G, CT
16	Laut Island, South-east Borneo	Unknown[1]	×	×	×		SS, I, G

Footnotes:
The ID number identifies the position of each area in Figure 4.
[1]IUCN Sun bear range map (Scotson et al., 2017a).
[2]Classified as definite in 2006 IUCN range map edition.
[3]Rural interviews reported bear presence in 74% of interviews, but there is no distinction between Asiatic black bear and sun bear (Crudge et al., 2016).
[4]Maxent habitat suitability models predicted that the only highly suitable habitat in the south of Peninsular Malaysia occurs within Endau Rompin National Park (Nazeri et al., 2012). The IUCN 2016 range map marks several patches in this region as definite.
[5]T = radio telemetry, G = genetics, CT = camera traps, SS = Sign surveys, I = rural interviews

connectivity (i.e., dispersal distance, tolerance to non-habitat, what 'non-habitat' really is) of sun bear are lacking. The speed at which forest cover is changing throughout the sun bear range exceeds the speed at which data on these key parameters are likely to become available. Using tree cover as a proxy for habitat suitability allowed a coarse insight into potential global subpopulations, and the potential status of sun bear within those subpopulations. The resultant maps of potential sun bear range connectivity give insights into research priorities for improving scientific understanding of sun bear range connectivity, and priority areas for sun bear conservation.

An overview of global sun bear range. Assuming that non-viable habitat was accurately classified, the global non-Chinese sun bear range can be divided into seven potential subpopulation blocks; two created by oceanic barriers between the islands of Sumatra and Borneo, and the mainland; and the five others a result of human-made fragmentation. Around 50% of forest was classified as core and sub-optimal habitat, which almost entirely over-lapped with sun bear range depicted by the IUCN (Scotson et al., 2017a) and extended into other contiguous areas. Therefore, if core and sub-optimal habitat is an accurate indication of viable sun bear habitat, the current sun bear IUCN range map may be underestimating the extent of sun bear range. Inaccuracies in the IUCN range map is a given for such a large and under surveyed region, but the differences in range extent may also reflect errors in the habitat suitability model. Determining the cause of these discrepancies requires on the ground-surveys.

Many core and sub-optimal areas appear to be in a highly fragmented state with high levels of human disturbance, suggesting that many areas within the defined subpopulations are at risk from further geographic isolation. Moreover, core sun bear range is poorly represented within the protected area network, which appears to primarily contain sub-optimal bear habitat, with almost 90% of core sun bear habitat falling outside protected areas. Sun bear conservation and subpopulation persistence in fragmented landscapes is likely to require active protection in many areas, to reduce poaching, protect and restore habitat, and in some cases, by augmenting populations.

Limitations. The interpretation of how habitat configuration affects the sun bear was largely speculative due to scarcity of data on the movement patterns of sun bear, especially on tolerance to moving through non-habitat. If bears are more resilient to moving through non-habitat than expected, then I may have overestimated the level of fragmentation of global range. Conversely, if sun bear is more sensitive than expected, range fragmentation has been underestimated. The extent of viable habitat is assumed to be overly optimistic; a weakness was in being unable to distinguish between marginal habitat that bears can cross (i.e., shrub, crop fields, plantations), versus that which they cannot (i.e., large highways, urban areas), or at least do not do so with

sufficient frequency to allow population linkage. Road traffic and human settlements reduce dispersal in other bear species (McLellan, 1989; Gibeau et al., 2002), and there are several records of road kill of sun bears crossing highways (Cheema, 2015). Moreover, as sun bear is heavily poached throughout their range (Scotson et al., 2017a), the risk of encountering humans when moving through non-natural vegetation likely reduces the suitably of such habitat. The sensitivity test, using minimum and maximum distances between neighboring patches to identify non-viable patches, caused little difference in the result. Therefore, the influence of the choice of distance between patches (4.5 km) on the conclusions of this analysis is minimal. Data intensive methods (e.g., FRAGSTATS, graph theory) may have allowed an objective model-based analysis of range connectivity, however data to inform such methods are not yet available. A further limitation is that there is no understanding of whether tree cover gained from single species plantations influences the connectivity between subpopulations—it presumably does in some cases, allowing sun bears to transition between contiguous patches of natural forest. Ground surveys are required to clarify this discrepancy.

Conservation implications. The long-term viability of sun bear populations depend on either habitat blocks being big enough to support healthy breeding populations even if totally isolated from all other populations, or on smaller blocks of habitat having exchange of bears from other areas. In mainland Southeast Asia, India, and Bangladesh, there are several potential range fractures. For example, in northern Myanmar and central Southeast Asia, a fracture might be caused by a break in core habitat, replaced by patchy sub-optimal, marginal and non-habitat, and a heavy road network. The IUCN sun bear range map supports the existence of this fracture, classifying the area directly south of the northern population limit as unknown bear range (Scotson et al., 2017a). Similar habitat transitions are evident in north-eastern Vietnam, southern Thailand, southern Peninsular Malaysia, east and west Sumatra, and around coastal Borneo, where breaks in core habitat into patchy lower quality and non-habitat are causing further subdivision of global sun bear range. Potential human fragmented island subpopulations exist in central Thailand, northern Cambodia and southern Vietnam, where relatively small patches of core habitat are almost surrounded by non-habitat, with very little potential for movement between small patches and larger neighboring blocks. In southern Cambodia, a moderate sized subpopulation is encompassed by non-habitat and roads. Several much smaller, isolated patches are located in southern Myanmar (where it is unclear if bears remain), the southern tip of Sumatra, in Way Kambas National Park, which is currently occupied by sun bears (Scotson et al., 2017a), and on a small island of the south-east coast of Borneo (if bears are present; Fig. 4). This visual analysis is non-exhaustive. For instance, there are many more fragments of forest found on Sumatra and Borneo, and many more neighboring islands where sun bear presence is uncertain (e.g., Banka, Belitung).

CONCLUSIONS

Sun bear researchers and managers can use these results as a guide to potential sun bear habitat status within their area of focus. In the absence of data to inform a full quantitative analysis of sun bear range connectivity, the modelled relationship between bear occurrence and percent tree cover allowed creation of a coarse overview of the global sun bear landscape. Results indicate that core habitat is poorly represented in the Protected Area network, and that sun bear may be mistakenly thought to be extinct within large areas of core habitat in which focused research has not yet been undertaken. While this is somewhat encouraging from a habitat availability standpoint, industrial hunting, particularly hunting with cable snares, has likely suppressed or extirpated bears from many of these areas (Gray et al., 2017; O'Kelly et al., 2018). Eventually, as researchers collect data on population estimates and movement patterns within and between subpopulations, the conservation needs of those subpopulations and the validity of these conclusions will become clearer.

LITERATURE CITED

Cheah C (2013) The Ecology of Malayan Sun Bears (*Helarctos malayanus*) at the Krau Wildlife Reserve, Pahang, Malaysia and Adjacent Plantations. Unpublished PhD dissertation, University Putra Malaysia, Malaysia, 275 pp.

Cheema MS (2015) Some insights into the sun bears of Brunei Darussalam. International Bear News, 24(2): 18–19.

Crudge B, Wilkinson NM, Thoai D Van, Dung CT, Trung CT, Weegenaar A & Hunt M (2016) The Status and Distribution of Bears in Vietnam. Free the Bears, Vietnam, 44 pp.

Dahle B & Swenson JE (2003) Home ranges in adult Scandinavian brown bears (Ursus arctos): Effect of mass, sex, reproductive category, population density and habitat type. Journal of Zoology, 260(4): 329–335.

Erdbrink DP (1953) A Review of the Fossil and Recent Bears of the Old World with Remarks on Their Philogenese Based Upon Their Dentition. Deventer: Drukkerij Jan de Lange, 596 pp.

Fahrig L (2003) Effects of Habitat Fragmentation on biodiversity. Annual Review of Ecology, Evolution, and Systematics, 34: 487–515.

Fredriksson GM (2012) Effects of El Niño and Large-scale Forest Fires on the Ecology and Conservation of Malayan Sun Bears (*Helarctos malayanus*) in East Kalimantan, Indonesian Borneo. Institute for Biodiversity and Ecosystem Dynamics. Unpublished PhD dissertation, University of Amsterdam, Netherlands, 122 pp.

Gibeau ML, Clevenger AP, Herrero S & Wierzchowski J (2002) Grizzly bear response to human development and activities in the Bow River Watershed, Alberta, Canada. Biological Conservation, 103(2): 227–236.

Gray TNE, Lynam AJ, Seng T, Laurance WF, Long B, Scotson L & Ripple WJ (2017) Wildlife-snaring crisis in Asian forests. Science, 355(6322): 225–256.

Guharajan R, Arnold TW, Bolongon G, Dibden GH, Abram NK, Teoh SW, Magguna MA, Goossens B, Wong ST, Nathan SKSS & Garshelis DL (2018) Survival strategies of a frugivore, the sun bear, in a forest-oil palm landscape. Biodiversity and Conservation, 27(14): 3657–3677.

Haddad NM, Brudvig LA, Clobert J, Davies KF, Gonzalez A, Holt RD, Lovejoy TE, Sexton JO, Austin MP, Collins CD, Cook WM, Damschen EI, Ewers RM, Foster BL, Jenkins CN, King AJ, Laurance WF, Levey DJ, Margules CR, Melbourne BA, Nicholls AO, Orrock JL, Song D & Townshend JR (2015) Habitat fragmentation and its lasting impact on Earth's ecosystems. Science Advances (March): 1–9. DOI: 10.1126/sciadv.1500052

Hansen MC, Potapov PV, Moore R, Hancher M, Turubanova SA, Tyukavina A, Thau D, Stehman SV, Goetz SJ, Loveland TR, Kommareddy A, Egorov A, Chini L, Justice CO & Townshend JRG (2013) High-resolution global maps of 21st-century forest cover change. Science, 342(6160): 850–853.

IUCN (2017a) Guidelines for Using the IUCN Red List Categories and Criteria, Version 13. Prepared by the Standards and Petitions Sub-committee. International Union for the Conservation of Nature, Gland, Switzerland, 108 pp. https://doi.org/http://www.iucnredlist.org/documents/RedListGuidelines.pdf. (Accessed 29 December 2018).

IUCN (2017b) IUCN Protected Areas. https://www.iucn.org/theme/protected-areas/about/protected-areas-categories. (Accessed 7 June 2017).

Kattan G, Hernández OL, Goldstein I, Rojas V, Murillo O, Gómez C, Restrepo H & Cuesta F (2004) Range fragmentation in the spectacled bear *Tremarctos ornatus* in the northern Andes. Oryx, 38(2): 155–163.

Keenan RJ, Reams GA, Achard F, Freitas JV De, Grainger A & Lindquist E (2015) Dynamics of global forest area: Results from the FAO Global Forest Resources Assessment 2015. Forest Ecology and Management, 352: 9–20.

Kendall KC, Stetz JB, Roon DA, Waits LP, Boulanger JB & Paetkau D (2008) Grizzly bear density in Glacier National Park, Montana. Journal of Wildlife Management, 72(8): 1693–1705.

Krause J, Unger T, Noçon A, Malaspinas AS, Kolokotronis SO, Stiller M, Soibelzon L, Spriggs H, Dear PH, Briggs AW, Bray SCE, O'Brien SJ, Rabeder G, Matheus P, Cooper A, Slatkin M, Pääbo S & Hofreiter M (2008) Mitochondrial genomes reveal an explosive radiation of extinct and extant bears near the Miocene-Pliocene boundary. BMC Evolutionary Biology, 8(1): 220.

Margono BA, Potapov P V, Turubanova S, Stolle F & Hansen MC (2014) Primary forest cover loss in Indonesia over 2000–2012. Nature Climate Change, 4: 1–6.

McLellan BN (1989) Dynamics of a grizzly bear population during a period of industrial resource extraction. I. Density and age–sex composition. Canadian Journal of Zoology, 67(8): 1856–1860.

McLellan BN & Hovey FW (2001) Natal dispersal of grizzly bears. Canadian Journal of Zoology, 79: 838–844.

McLellan BN, Proctor MF, Huber D & Michel S (2016) *Ursus arctos* Brown bear. IUCN Red List of Threatened Species. http://www.iucnredlist.org/details/summary/41688/0. (Accessed 29 December 2018).

Meijaard E (2004) Craniometric differences among Malayan sun bears (*Ursus malayanus*); Evolutionary and taxonomic implications. The Raffles Bulletin of Zoology, 52(2): 665–672.

Miettinen J, Shi C & Liew SC (2011) Deforestation rates in insular Southeast Asia between 2000 and 2010. Global Change Biology, 17(7): 2261–2270.

Nazeri M, Jusoff K, Madani N, Mahmud AR, Bahman AR & Kumar L (2012) Predictive modeling and mapping of Malayan sun bear (*Helarctos malayanus*) distribution using maximum entropy. PLoS ONE, 7(10): e48104.

Ngoprasert D, Gale G., Vichitsoonthonkul T & Vaeokhaw S (2015) Population Demographics and Genetics of Asiatic Black Bear and Sun Bear in the World Heritage Dong Phayayen - Khao Yai Forest Complex. Report to International Bear Association, AK, USA, 38 pp.

Normua F, Higashi S, Ambu L & Mohamed M (2004) Notes on oil palm plantation use and seasonal spatial relationships of sun bears in Sabah, Malaysia. Ursus, 15(2): 227–231.

North MA (2009) A method for implementing a statistically significant number of data classes in the Jenks algorithm. 2009 Sixth International Conference on Fuzzy Systems and Knowledge Discovery, Tianjin, 2009, 1: 35–38.

O'Kelly HJ, Rowcliffe JM, Durant S & Milner-Gulland EJ (2018) Experimental estimation of snare detectability for robust threat monitoring. Ecology and Evolution, 8(3): 1778–1785.

Proctor MF, Paetkau D, McLellan BN, Stenhouse GB, Kendall KC, MacE RD, Kasworm WF, Servheen C, Lausen CL, Gibeau ML, Wakkinen WL, Haroldson MA, Mowat G, Apps CD, Ciarniello LM, Barclay RMR, Boyce MS, Schwartz CC & Strobeck C (2012) Population fragmentation and inter-ecosystem movements of grizzly bears in Western Canada and the Northern United States. Wildlife Monographs, (180): 1–46.

R Core Team (2016) R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria. http://www.R-project.org. (Accessed 29 December 2018).

Riitters K, Wickham J, Costanza JK & Vogt P (2016) A global evaluation of forest interior area dynamics using tree cover data from 2000 to 2012. Landscape Ecology, 31(1): 137–148.

Sanderson EW, Jaiteh M, Levy M, Redford KH, Wannebo AV & Woolmer G (2002) The human footprint and the last of the wild. BioScience, 52(10): 891–904.

Scotson L, Fredriksson G, Augeri D, Chea C, Ngoprasert D & Wong WM (2017a) *Helarctos malayanus* Sun bear. IUCN Red List of Threatened Species. www.iucnredlist.org. (Accessed 29 December 2018).

Scotson L, Fredriksson G, Ngoprasert D, Wong W-M & Fieberg J (2017b) Projecting range-wide sun bear population trends using tree cover and camera-trap bycatch data. PLoS ONE, 12(9).

Sodhi NS, Posa MRC, Lee TM, Bickford D, Koh LP & Brook BW (2010) The state and conservation of Southeast Asian biodiversity. Biodiversity and Conservation, 19(2): 317–328.

Staddon P, Lindo Z, Crittenden PD, Gilbert F & Gonzalez A (2010) Connectivity, non-random extinction and ecosystem function in experimental metacommunities. Ecology Letters, 13(5): 543–552.

Stibig H-J, Beuchle R & Achard F (2003) Mapping of the tropical forest cover of insular Southeast Asia from SPOT4-Vegetation images. International Journal of Remote Sensing, 24(18): 3651–3662.

Swaisgood R, Wang D & Wei F (2016) *Ailuropoda melanoleuca* (Giant Panda). IUCN Red List of Threatened Species. The IUCN Red List of Threatened Species. http://www.iucnredlist.org/details/summary/712/0. (Accessed 29 December 2018).

Tilman D, May RM, Lehman CL & Nowak MA (1994) Habitat destruction and the extinction debt. Nature, 371(6492): 65–66.

Wade TG, Riitters KH, Wickham JD & Jones KB (2003) Distribution and causes of global forest fragmentation. Ecology and Society, 7(2): 7 (online) URL: http://www.consecol.org/. (Accessed 29 December 2018).

Wiig Ø, Amstrup S, Atwood T, Laidre K, Lunn N, Obbard M, Regehr E & Thiemann G (2015) *Ursus maritimus*, Polar bear. IUCN Red List of Threatened Species. http://www.iucnredlist.org/details/summary/22823/0. (Accessed 29 December 2018).

Wong ST & Servheen CW AL (2004) Home range, movement and activity patterns, and bedding sites of Malayan sun bears *Helarctos malayanus* in the Rainforest of Borneo. Biological Conservation, 119: 169–181.