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Deforestation and forest fragmentation in and around Endau-Rompin National Park, Peninsular Malaysia

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ABSTRACT Protected areas (PAs) are crucial for conserving biodiversity, but there is controversy over whether they effectively reduce deforestation because, for many of the PAs evaluated to date, pressure from deforestation occurred both inside the PAs and in the areas surrounding them. This is especially the case for tropical PAs in Southeast Asia. We examined the impact of a protected area on tropical rainforests inside and surrounding Endau-Rompin National Park (ERNP) in southern Peninsular Malaysia by mapping and analyzing forest cover changes and fragmentation between 1992 and 2016. The results showed that the forests inside ERNP were well protected, but greater forest loss and fragmentation were found surrounding ERNP, especially beyond the 10-km buffer zone of the park, due to large-scale agricultural land conversion, particularly for oil palm plantation. The deforestation rate in areas surrounding PAs in the region increased from 250 ha yr⁻¹ during 1992–2007 to 1,700 ha yr⁻¹ during 2012–2016. In the buffer zone (BZ), the deforestation rate was extremely high during 2007–2012 at 1,800 ha yr⁻¹, but decreased to 440 ha yr⁻¹ during 2012–2016. This suggests that in this region, PAs might be ineffective against deforestation in their surrounding areas. Continual deforestation and fragmentation are expected to occur surrounding ERNP, threatening its protected boundary. These activities may reduce the effectiveness of ERNP for wildlife habitat conservation.

Key words: biodiversity conservation, buffer zone, land use and land cover change, protected area, remote sensing, tropical rainforest

INTRODUCTION

Tropical forests are a major conservation target as they are the biologically richest ecosystems on Earth and support approximately two-thirds of the world's biodiversity (Laurance et al. 2012). In tropical forest reserves, most tropical biodiversity is effectively protected within the administrative boundary, but the outer forest is not (Bruner et al. 2001). However, the presence of protected areas (PAs) has significantly reduced forest loss rates in the buffer zone (BZ) closest to the PA boundary (Spracklen et al. 2015).

The establishment of PAs is one of the greatest and most successful efforts in protecting and conserving global biodiversity (Jenkins and Joppa 2009, Wang et al. 2013). In the last few decades, biodiversity has declined globally, attributed in large part to forest fragmentation and land use and land cover (LULC) change (Woodley et al. 2015). To conserve biodiversity and the associated ecosystem services, various types of protected areas (PAs) such as strict nature reserves and wilderness areas, national parks, natural monuments or features, habitat or species managements areas, protected land or seascapes, and protected area with sustainable use of natural resources have been formed

(Dennis 1999, Bruner et al. 2001, Arturo Sánchez-Azofeifa et al. 2002, Leisher et al. 2013, Mascia et al. 2014, Hashim et al. 2017). The global extent of these PAs reached 14.7 % in 2018, from 3.48 % in 1985, with more than 202,000 terrestrial PAs (Jenkins and Joppa 2009, Jones et al. 2018). The efficiency of biodiversity conservation in a PA is affected by both deforestation and forest fragmentation within the PA and its surrounding landscape (Kintz et al. 2006, Townsend et al. 2009). There is evidence that some PAs have undergone degradation, and this degradation is increasing, especially in the tropics (Chapman and Lambert 2000, Gaston et al. 2008, Spracklen et al. 2015). The forest surrounding the PAs is important for conserving the species richness within and outside individual PAs (DeFries et al. 2005) including large terrestrial mammals, such as the Asian elephant and Malayan tiger (Shevade et al. 2017, Bahar et al. 2018). Endau-Rompin National Park (ERNP) in Peninsular Malaysia is an important habitat for such endangered large mammals, its current size is inadequate to sustain long term viable populations for the species (Robert Aiken and Leigh 1984, Clements et al. 2010, Garland et al. 2017). Aiken (1994), reported that the species richness was actually higher outside ERNP than within its boundary. The

estimated Asian elephant population was 48 within the ERNP and 87 in the contiguous Permanent Forest Reserve (Saaban 2011). Thus, the condition of the area surrounding a PA is important for biodiversity conservation (Hansen et al. 2011) and has important implications for the management and conservation of the PA (Kintz et al. 2006). Unfortunately, intensive losses of forest area are frequently observed, especially in areas surrounding protected lands (Hansen et al. 2004, Willis 2015). Extensive forest loss in areas surrounding PAs in Southeast Asia could be an indication that human impacts have leaked from the designated PAs to nearby, unrestricted areas (Oliveira et al. 2007, Ewers and Rodrigues 2008).

Leakage is categorized as the negative impact of terrestrial PAs on their unprotected surrounding areas such that the deforestation that is restricted in the PAs is displaced to unprotected areas (Ewers and Rodrigues 2008, Renwick et al. 2015, Kamlun et al. 2016, Fuller et al. 2019). Consequently, leakage might accelerate deforestation in areas surrounding PAs, isolating the PA's habitat, which may offset any benefits achieved inside PAs (Ewers and Rodrigues 2008, Fuller et al. 2019).

Forest fragmentation can be defined as the process of a formerly large continuous forest changing into smaller isolated forest patches that are increasingly affected by edge effects and are vulnerable to other land use conversions (Farhadur 2016, Boro Choudhury and Sharma 2017, Rajani Kumari. R and Smitha Asok V 2017). Forest fragmentation is associated with deforestation (Batar et al. 2017). Forest loss refers only to the conversion of forested land to other land uses, but forest fragmentation occurs when a large region of forest is broken down, or fragmented, into a collection of smaller patches of forest habitat (Collingham and Huntley 2000, Fahrig 2010, Batar et al. 2017, Ramachandran et al. 2018). Forest loss is most detrimental in an already-fragmented landscape (Betts et al. 2017).

Remote sensing is possibly the most frequently used technique in mapping LULC change, as well as the biophysical and ecological attributes of the terrestrial surface (Lambin et al. 2001, Geist and Lambin 2002, Nagendra 2008, Bailey et al. 2016). The availability of remotely sensed data in various temporal and spatial scales provides more effective means of monitoring and measuring the rates and consequences of LULC change over large spatial and temporal extents, thus offering a better understanding of previous, current, and future LULC changes and magnitudes (Houghton 1994, Kintz et al. 2006, Bailey et al. 2016).

The present study focuses on the efficacy of PAs on forests inside and outside a national park, in terms of the

quality and quantity of forests. Endau-Rompin National Park (ERNP) is one of the last remaining extensive tracts of largely undisturbed rainforest in southern Peninsular Malaysia. Although ERNP is almost intact, the forest cover surrounding the park has significantly declined due to intensive agricultural activities (Clements et al. 2010). Although these agricultural activities occurred outside the ERNP administrative boundary, they still pose threats to the ecological processes and biodiversity in ERNP. Such intensive agricultural activities might lead to overexploitation of natural resources in and around the region, but data and documentation on forest quantity and quality surrounding the Endau-Rompin National Park (ERNP) are very limited. In this study, we investigated deforestation and fragmentation inside the PA and in its unprotected adjacent surroundings to assess the effectiveness of the PA in reducing deforestation. First, we compared land use and land cover within ERNP with that in areas surrounding the park to quantitatively examine the nature of deforestation among the different areas. Subsequently, we compared the forest fragmentation in these areas to provide a qualitative examination of the deforestation. Finally, the trends in forest area change and fragmentation inside ERNP and in its surrounding areas were assessed to examine the effectiveness of the PA in limiting deforestation.

MATERIALS AND METHODS

Study site

ERNP is the second largest national park in Peninsular Malaysia. It is located partly in the Malaysian states of Pahang and Johor. The northern part (378 km²) of the park is designated as Endau-Rompin State Park and managed by the Pahang state government, while the southern part (489 km²) is under the management of the Johor National Parks Corporation (JNPC). There are two entry points to ERNP on the Johor side, the Peta and Selai entrances. For ERNP, Pahang, there is one entrance in Rompin. ERNP is a lowland rainforest with an average annual temperature of 27 °C, rainfall of 3,400 mm, and humidity of 85 %.

In 1972, the suggestion to establish Endau-Rompin as a national park to conserve the Sumatran rhinoceros was rejected by the state government at that time due to the controversy over the management rights of the forested land between the federal and the state government (Robert Aiken and Leigh 1984). Until 1987, both the Pahang and Johor state governments agreed that the two state parks were joined as Endau-Rompin. In 1993, Endau-Rompin,

Johor, was declared to be a state-controlled national park (Shahriza et al. 2012, Birdlife International 2018).

ERNP provides habitats for 149 species of mammals, including critically endangered mammals such as the Sumatran rhinoceros (*Dicerorhinus sumatrensis*), and endangered species such as the Malayan tiger (*Panthera tigris*), Malayan tapir (*Tapirus indicus*), and Asian elephant (*Elephas maximus*) (Aihara et al. 2016). In the park, 253 bird species, including seven species of hornbills, can be found (Birdlife International 2018). The rich aquatic habitats support more than 76 species of freshwater fish recorded from the rivers, including scarce species such as the prized Malayan mahseer (*Tor tambroides*) and green arowana (*Scleropages formosus*) (Shahriza et al. 2012). There are 71 species of palms and 150 species of ferns documented to date (Kiew et al. 1987). However, the population of Sumatran rhinoceros has decreased to merely a trace since 1996, due to degradation, land use conversion, habitat disturbance, logging, forest encroachment, and poaching pressure (Flynn and Abdullah 1983, Kadir 2006, Clements et al. 2010, Aihara et al. 2016). According to Clements et al. (2010), the last Sumatran rhinoceros in ERNP was poached in 2003. ERNP was chosen for this study due to its importance for conservation, its high degree of protection status (category II: national park) as defined

by the International Union for the Conservation of Nature (IUCN), its relatively large size (>25,600 ha) and forest cover (>60%) within the administrative boundaries in the early 1980s, and its reserved boundaries as delineated in the World Database of Protected Areas (WDPA) developed by United Nations Environment Programs (Bruner et al. 2001, DeFries et al. 2005). Most importantly, the uniqueness of ERNP is that it is located in the “Riouw Pocket,” a meeting point of the western Borneo, Sumatran, and Malayan flora (Birdlife International 2018). It is characterized by a high degree of endemism, and a significant number of plant species that are locally endemic or restricted to the southern region of the peninsula are found in ERNP (Birdlife International 2018). There are at least eight species endemics to the park, including *Calamus endauensis* (Arecaceae) and *Loxocarpus tunkui* (Gesneriaceae). Ten species found within the park have a limited distribution in Johor State, and eight occur only in the southern and eastern parts of the peninsula (Birdlife International 2018).

The 3,721.45 km² study area is located in southeastern Peninsular Malaysia, extending from 2°16'23.5" N to 2°50'18.5" N and 102°58'10.9" E to 103°30'23.2" E (Fig. 1). The area includes the ERNP, a 10 km buffer zone extending from the ERNP's administrative boundary, and an extended area beyond the buffer zone. Due to the remote location of

Fig. 1. Maps of (a) Peninsular Malaysia: location of Endau-Rompin National Park (ERNP) and (b) study site: Endau-Rompin National Park, 10 km buffer zone (BZ), and outskirts region.

ERNP, the park is surrounded by a large area of forest connecting the park and agricultural land. Indigenous communities with a total population of less than 1,000 are located close to the entrance of ERNP (Garland *et al.* 2017). Facilities such as road networks are not well developed within the 10 km buffer zone area.

Data collection

The data acquisition and assessment processes are

outlined in Fig. 2. To obtain the spatial and temporal LULC information of ERNP and its surrounding BZ, Landsat images with cloud cover < 30 % from 1992, 2007, 2012, and 2016 were selected. A Landsat dataset with 30-m spatial resolution was sufficient for monitoring both the patterns in forest loss and fragmentation and the factors influencing the forest patterns at a local scale (Cassidy *et al.* 2010). The Landsat images were radiometrically and atmospherically corrected and then georeferenced to WGS 84/UTM zone 48.

Fig. 2. Methodological flowchart: data acquisitions, land use land cover dynamic assessment, and forest fragmentation assessment.

Land cover classification

According to the Food and Agricultural Organization (FAO), land cover is the observed biophysical cover on the Earth's surface, while land use is the arrangements, activities, and inputs that people undertake in a certain land cover type to produce, change, or maintain it. The pre-processed Landsat images were divided into subsets and the areas with cloud cover in all images were masked to avoid bias. The images were then used to determine the current extent and changes of LULC in this study. The LULC distribution in ERNP and its surroundings (10-km buffer from the administrative boundary) was mapped. According to Mas (2005), a 10-km buffer distance is large enough to obtain reliable statistics on LULC change based on multi-temporal Landsat imagery. The classification scheme adopted is described in Table S-1. Using a supervised classification method, the maximum likelihood algorithm was employed to classify the Landsat images (1992, 2007, 2012, and 2016) with different training sites selected based on the land use maps published by the Department of Agriculture Malaysia in 2008, 2013, and 2015.

Accuracy assessment

Confusion matrix and kappa statistics were used to evaluate the produced classification output using a land use map as reference information. The accuracy for each class was indicated through the statistics in the error matrix, while the accuracy of the map produced was determined by the overall accuracy and kappa statistic. A kappa statistic ≥ 0.80 was indicative of strong agreement (Jensen 2005, McHugh 2012). For accuracy assessment purposes, totals of 260, 190, and 260 ground truth points were selected based on Department of Agriculture land use maps from 2008, 2013, and 2015 for the years 2007, 2012, and 2016, respectively. However, the accuracy assessment for 1992 was unable to be conducted due to a lack of ground data for the study area in that year.

Forest fragmentation assessment

The fragmentation of forest land was evaluated spatially using the Landscape Fragmentation Tool (LFT) v2.0, which plays a significant role in assessing temporal forest fragmentation within the core and buffer zones in a study area (Vogt et al. 2007). The LFT classifies spatial patterns at the pixel level on binary land cover maps (Vogt

et al. 2007). Therefore, the map is reclassified into forest (comprising forest land) and non-forest (comprising agriculture and cleared land) classes, while the waterbody class is excluded at this stage. The LFT maps fragmentation as four main categories (patch, edge, perforated, and core forests), as described in Table S-2 and Fig. S-1. The areas occupied by different classes represent the quality (fragmented class) and quantity (total forest area) assessment of the forest in the studied area (Farhadur 2016). The edge width indicates the distance over which non-forest land covers can degrade forest land covers (Farhadur 2016, Batar et al. 2017). The specific edge width of 100 m was selected, as it is often used as a general edge width (Vogt et al. 2007, Batar, Watanabe and Kumar 2017, Boro, Choudhury and Sharma 2017).

As the edge width was set to 100 m, forest patches did not contain any forest pixels that were more than 100 m from non-forest, and these forest patches were entirely encompassed by the edge-effect. Core forests consist of any forest pixels that are more than 100 m from non-forest, and peripheral forest consists of forest pixels that are within 100 m of non-forest and the tract contains core forest (the peripheral forest is then further classed into edge and perforated forest) (CLEAR 2018).

In this study, patches of forests were categorized into four main classes: patch, perforated, edge, and core forests. Core areas were further divided into small (<250 ha), medium (250–500 ha), and large (>500 ha) forests to indicate the viability of the core patches with respect to the patch size (Fig. 3).

Annual change rate

The annual change rates of forest area and forest fragmentation were calculated by comparing the area under each class type in the same region at two different times. The annual rate of change for forest area and the annual fragment creation rate were calculated using an equation derived from the compound interest law, as follows (Puyravaud 2003):

$$r = \left[\left(\frac{1}{t_2 - t_1} \right) \times \ln \left(\frac{A_2}{A_1} \right) \right] \times 100 \quad (1)$$

where r (unit: percentage per year) is the annual change rate, and A_2 and A_1 are forest land area at times t_2 and t_1 , respectively.

Fig. 3. Maps of forest fragmentation patterns obtained at 100 m edge width using LFT v2 for year (a) 1992, (b) 2007, (c) 2012, and (d) 2016.

RESULTS

Forest area change

LULC classification maps of ERNP and its BZ and outskirts were produced with forest areas for the years 1992, 2007, 2012, and 2016 (Fig. 4). The overall accuracies and kappa statistics for the classification outputs for 2007, 2012, and 2016 were 92.72 % and 0.82, 91.71 % and 0.80, and 94.97 % and 0.80, respectively (Table S3).

Generally, forest land remained the dominant land cover over the studied period, followed by agricultural land, built-up or cleared land, and waterbody. ERNP and its 10-km BZ were both dominated by forested land, except for the outskirts region, which was dominated by agricultural land within the studied period (Fig. 4).

From 1992 to 2016, large declines in forest area were found in both the BZ and the outskirts region. In ERNP, forest land covered more than 90 % of the total area and there was no significant change in forest area (Fig. 5). The forest area within ERNP increased slightly from 97.7 % in 1992 to 98.0 % in 2016. In the 10-km BZ of ERNP, the changes in forest areas did not follow a consistent pattern. Initially,

there was an increase in forest land area from 67.9 % to 71.0 % in the first period (1992–2007), but forest land area began to decrease in 2012 and reached 64.1 % by 2016. By contrast, agricultural land in the BZ decreased initially during the first period by approximately 8 % and then increased to 27.3 % by 2012, followed by another decrease, reaching 24.3 % in 2016. A similar change in agricultural land was also observed in the outskirts region, but the forest land area in the outskirts region showed a continuous decrease from 37.6 % in 1992 to 22.7 % in 2016. Greater forest loss occurred in the outskirts region than in the BZ. The lost forest land and agricultural land were replaced by increased cleared land area in the BZ and outskirts region (Fig. 5). Significant conversion was observed, especially in the BZ and outskirts region. Large clearances of forest land and agricultural land occurred, and majority of the cleared land was replaced by agricultural land. Only a minority of the cleared land experienced forest regrowth.

Forest fragmentation

In 1992, the outskirts region was dominated by a large

Fig. 4. LULC maps obtained by supervised classification using Landsat images for year 1992, 2007, 2012, and 2016.

core forest area (24.2 %), followed by edge forest (6.5 %), perforated forest (3.3 %), patch forest (1.3 %), small core forest (1.3 %), and medium core forest (0.9 %) (Fig. 6) (Table 1). Our results revealed a significant decrease in large core forest area from 24.2 % in 1992 to 13.9 % in 2016. The areas of edge, perforated, and small core forest increased in the first period but then decreased progressively in the second and third periods. The medium core forest area decreased by 0.8 % in the first period and then increased slightly by 0.3 % and 0.2 % in the second and the third periods, respectively. The patch forest area increased slightly in the first period, then decreased in the second period, and increased again in the third period.

Similarly, the BZ was also dominated by a large core forest area (48.3 %), followed by edge forest (9.5 %), perforated forest (6.8 %), small core forest (1.8 %), patch

forest (1.1 %), and medium core forest (0.38). Different from the outskirt region, a continuous decrease in small core forest occurred from 1992 (1.8 %) to 2016 (1.0 %). The areas of patch and medium core forest gradually decreased in the first and second periods but increased in the third period. The large core area in the BZ increased by 4.4 % and 0.4 % in the first and second periods, respectively, but a slight decrease of 0.3 % was observed in the third period.

ERNP was dominated by a large core area (93.2 %) with little forest fragmentation from 1992 to 2016. The remaining forest area was occupied by patch, perforated, edge, and core forests. According to the annual change rate (Table 1), the large core forest in ERNP showed no fragmentation occurrences between 1992 and 2016.

Fig. 5. LULC trajectory diagram for Endau-Rompin National Park, the buffer zone, and the outskirts region from 1992 to 2016.

DISCUSSION

Trends in forest area change

Our results revealed significant changes in forest area in the surroundings of ERNP between 1992 to 2016 due to increases in agricultural land in the BZ and outskirts region. This result was consistent with a previous study's report that conversion to agricultural land is the main cause of forest cover loss in Peninsular Malaysia (McMorrow and Talip 2001).

Comparing the annual forest loss between ERNP, the

BZ, and the outskirts region, forest loss in the outskirts area was greater than in the BZ and ERNP. The outskirts region lost 40 % of its forest area from 1992 to 2016, which was ten times greater than that of the BZ. Miettinen, Shi, and Liew (2011) reported that annual forest loss in insular Southeast Asia after the 2000 s was lower than in the 1990 s. However, annual forest loss in the outskirts region of ERNP increased in the 2000 s compared to the 1990 s (Fig. 7). Particularly, large-scale (100–1,000 ha) and very large-scale (>1,000 ha) forest loss occurred in both the BZ and outskirts region. These results suggest that the forest inside the protected area in ERNP was effectively conserved, but

Fig. 6. Trends in forest fragmentation of ERNP, BZ, and outskirts region between 1992 and 2016. Fragmentation is measured by distribution of forest fragmentation classes area in percentage.

the surroundings were more susceptible to deforestation. There are other possible causes to consider further, such as accessibility, topographic condition, economic opportunities, or agricultural land suitability (Lambin et al. 2001, Cassidy et al. 2010, Bonilla-Moheno et al. 2012).

The increase in forest loss in the surroundings of ERNP was the result of large commodity crop expansion, specifically the oil palm, which is consistent with the findings of previous studies (Wicke et al. 2011, Linder and Palkovitz 2016, Austin et al. 2017). Large scale land conversion to oil palm plantation in the surroundings of ERNP is of great concern, as it may pose a threat to conservation in ERNP (Linder and Palkovitz 2016). According to the Malaysia Palm Oil Board (MPOB 2018), both Pahang and Johor states had the largest areas of oil palm plantation, and the area is increasing annually. Based on current trends and the results of this study, more extensive deforestation is expected to occur in the coming years, shifting from the outskirts, and approaching the ERNP boundary.

Trends in forest area fragmentation inside and outside ERNP

From 1992 to 2016, the forests in the outskirts region

experienced the highest fragmentation (Fig. 6). The fragmentation increased as the large core forest was continually disconnected from the core forest and diminished at a significant rate (Fig. 6). In the BZ, forest fragmentation increased during the second and third period. Although very large-scale forest loss ($>1,000 \text{ ha yr}^{-1}$) occurred during the second period, the large core forest area was well protected. The patch forest, small core forest, and medium core forest areas were decreased during the second period, showing that smaller or fragmented forest areas are more prone to deforestation compared to large core forest areas. During the third period, there was an increase in patch forest and medium core forest areas and a decrease in large core forest area, suggesting continued disconnection of forests from the large core forest. The smaller forest areas disconnected from the large core forest have high potential to be deforested in the future. Meanwhile, forest fragmentation conditions within ERNP were well maintained, as there was no sign of increased forest fragmentation during the studied period.

Agricultural land expansion was the major driver of forest fragmentation in the present study. The extensive conversion of forest cover to agricultural land, especially oil palm plantation, increased forest fragmentation, which is likely to cause decreases in the population sizes of many animal species within the PA, including the Asian elephant

Table 1. Forest fragmentation area and percentage, and the annual rate of change of each class in (i) Endau-Rompin National Park; (ii) the buffer zone; and (iii) the outskirts region.

(i) ERNP

Class	1992		2007		2012		2016		Change %			Annual Change Rate %		
	Ha	%	Ha	%	Ha	%	Ha	%	T ^a	T ^b	T ^c	T ^a	T ^b	T ^c
Patch	26.42	0.03	17.03	0.02	15.39	0.02	65.33	0.08	-0.01	0.00	0.06	-2.93	-2.02	36.14
Edge	1069.40	1.23	1067.06	1.23	1264.15	1.46	1131.54	1.30	0.00	0.23	-0.15	-0.01	3.39	-2.77
Perforated	2664.55	3.07	3925.68	4.52	1826.97	2.10	1365.03	1.57	1.45	-2.42	-0.53	2.58	-15.30	-7.29
Core (<250 ha)	82.12	0.09	14.99	0.02	6.94	0.01	31.41	0.04	-0.08	-0.01	0.03	-11.34	-15.40	37.75
Core (250–500 ha)	0.00	0.00	0.50	0.00	0.00	0.00	66.83	0.08	0.00	0.00	0.08	—	—	—
Core (>500 ha)	80945.12	93.24	79667.07	91.77	81515.62	93.90	82434.25	94.96	-1.47	2.13	1.06	-0.11	0.46	0.28
Total	84787.62	97.67	84692.32	97.56	84629.07	97.49	85094.38	98.02	-0.11	-0.07	0.54	-0.01	-0.01	0.14

(ii) BZ

Class	1992		2007		2012		2016		Change %			Annual Change Rate %		
	Ha	%	Ha	%	Ha	%	Ha	%	T ^a	T ^b	T ^c	T ^a	T ^b	T ^c
Patch	1775.46	1.13	1663.57	1.06	487.92	0.31	1011.95	0.64	-0.07	-0.75	0.33	-0.43	-24.53	18.24
Edge	14928.90	9.47	9962.11	6.32	10534.28	6.68	10091.46	6.40	-3.15	0.36	-0.28	-2.70	1.12	-1.07
Perforated	10711.36	6.80	13169.33	8.36	5971.75	3.79	4721.74	3.00	1.56	-4.57	-0.79	1.38	-15.82	-5.87
Core (<250 ha)	2803.27	1.78	1907.33	1.21	1808.81	1.15	1603.71	1.02	-0.57	-0.06	-0.13	-2.57	-1.06	-3.01
Core (250–500 ha)	597.17	0.38	458.95	0.29	253.98	0.16	326.38	0.21	-0.09	-0.13	0.05	-1.76	-11.83	6.27
Core (>500 ha)	76193.61	48.34	83122.35	52.74	83686.23	53.10	83224.27	52.80	4.40	0.36	-0.29	0.58	0.14	-0.14
Total	107009.77	67.90	110283.64	69.97	102742.97	65.19	100979.51	64.07	2.08	-4.78	-1.12	0.20	-1.42	-0.43

(iii) Outskirt

Class	1992		2007		2012		2016		Change %			Annual Change Rate %		
	Ha	%	Ha	%	Ha	%	Ha	%	T ^a	T ^b	T ^c	T ^a	T ^b	T ^c
Patch	1534.77	1.33	2131.62	1.85	860.19	0.75	1155.72	1.01	0.52	-1.11	0.26	2.19	-18.15	7.38
Edge	7415.65	6.45	7457.01	6.49	6414.32	5.58	5293.62	4.60	0.04	-0.91	-0.97	0.04	-3.01	-4.80
Perforated	3836.59	3.34	4663.84	4.06	2143.85	1.86	1637.66	1.42	0.72	-2.19	-0.44	1.30	-15.54	-6.73
Core (<250 ha)	1513.45	1.32	2234.61	1.94	1952.91	1.70	1380.96	1.20	0.63	-0.25	-0.50	2.60	-2.69	-8.66
Core (250–500 ha)	1001.59	0.87	116.01	0.10	460.08	0.40	645.03	0.56	-0.77	0.30	0.16	-14.37	27.55	8.45
Core (>500 ha)	27868.15	24.24	22628.04	19.68	21356.10	18.57	15939.16	13.86	-4.56	-1.11	-4.71	-1.39	-1.16	-7.31
Total	43170.20	37.55	39231.13	34.12	33187.44	28.86	26052.14	22.66	-3.43	-5.26	-6.21	-0.64	-3.35	-6.05

T^a (1992–2007), T^b (2007–2012), T^c (2012–2016)

Fig. 7. Deforestation rate in ERNP, BZ, and outskirt region. Deforestation rates were calculated over average time periods of 15 (1992–2007), 5 (2007–2012) and 4 (2012–2016) years in the unit of ha/yr.

and Malayan tiger (Powell 1997, Shevade et al. 2017, Bahar et al. 2018), posing a significant threat to local biodiversity (Linder and Palkovitz 2016). Large-scale forest loss and fragmentation in the BZ and outskirt region due to land conversion to oil palm plantation resulted in habitat shrinking (Munsi et al. 2010) and may lead to species shifts and biodiversity loss in the national park (Chai et al. 2009).

CONCLUSION

Forest area loss and forest fragmentation in and surrounding ERNP from 1992 to 2016 were assessed in this study. ERNP was well protected in terms of deforestation and forest fragmentation, although the areas surrounding ERNP experienced significant decreases in forest cover and increases in forest fragmentation due to agricultural land expansion, especially oil palm plantation. The rate of deforestation and the level of forest fragmentation were more severe in the surroundings of ERNP than in other forested areas in Peninsular Malaysia, suggesting that protected areas are ineffective as protection against deforestation in their surrounding areas. In this area, an important habitat for the Asian elephant and Malayan tiger, the secondary forest surrounding ERNP is an important source of the elephant's food supply (Powell 1997). Large-scale land conversion and fragmentation of secondary forests surrounding ERNP will affect the home range sizes and movement patterns of the local Asian elephants (Bahar,

Kasim and Hambali 2018).

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Supplementary Information

Fig. S-1. Illustration of four types of forest fragmentation class on an artificial map (Vogt et al. 2007).

Table S-1. Classification scheme (Ramachandran et al. 2018).

Land Use Land Cover Class (Level I)	Description
Forest	Includes all land classified as forest both under any legal enactment or administration.
Agriculture	Includes all cultivable land and land under plantation (both forest plantation and commercial plantation).
Waterbody	Includes all waterbodies
Built-up/Cleared Land	Includes land occupied by buildings, roads, and railways or underwater or land being cleared

Table S-2. Forest fragmentation classes and criteria (Boro, Choudhury and Sharma, 2017).

No	Class	Descriptions
1	Patch	Forest pixels that comprise a small forested area surrounded by non-forested land cover
2	Edge	Forest pixels that define the boundary between core forest and large non-forested land cover features
3	Perforated	Forest pixels that define the boundary between core forest and relatively small clearing (perforations) within the forested landscape
4	Core	Forest pixels that are relatively far from the forest non-forest boundary. Essentially these are forested areas surrounded by more forested area
	Small core	Smaller than 250 acres
	Medium core	Between 250 and 500 acres
	Large core	Larger than 500 acres

Table S-3. Accuracy assessment.

(i) 2007

	producer accuracy [%]	user accuracy [%]	Kappa hat
Agriculture land	88	88.48	0.77
Built-up/Cleared land	—	—	—
Forest land	94.2	99.49	0.98
Water	88.89	100	1
Overall accuracy [%] = 92.72			
Kappa hat classification = 0.83			

(ii) 2012

	producer accuracy [%]	user accuracy [%]	Kappa hat
Agriculture land	86.96	86.95	0.85
Built-up/Cleared land	—	—	—
Forest land	93.29	99.28	0.97
Water	85.71	100	1
Overall accuracy [%] = 91.71			
Kappa hat classification = 0.80			

(iii) 2016

	producer accuracy [%]	user accuracy [%]	Kappa hat
Agriculture land	66.67	90.91	0.9
Built-up/Cleared land	—	—	—
Forest land	97.4	98.04	0.86
Water	100	100	1
Overall accuracy [%] = 94.97			
Kappa hat classification = 0.80			