Forest Ecology and Management 427 (2018) 124–134

Contents lists available at ScienceDirect

Forest Ecology and Management

journal homepage: www.elsevier.com/locate/foreco

Effects of logging on wildlife communities in certified tropical rainforests in East Kalimantan, Indonesia

Agus Sudibyo Jati[a,b,*], Hiromitsu Samejima[c], Shogoro Fujiki[a], Yuyun Kurniawan[d], Ryota Aoyagi[a], Kanehiro Kitayama[a]

[a] Graduate School of Agriculture, Kyoto University, Kitashirakawa Oiwake-cho, Sakyo-ku, Kyoto 606-8502, Japan
[b] Faculty of Forestry, Universitas Gadjah Mada, Jl. Agro No. 1, Bulaksumur, Kabupaten Sleman, Daerah Istimewa Yogyakarta 55281, Indonesia
[c] Institute for Global Environmental Strategies, 2108-11 Kamiyamaguchi, Hayama, Miura District, Kanagawa Prefecture 240-0115, Japan
[d] WWF Indonesia, Graha Simatupang, Tower 2 Unit C, 7th-11th Floor, Jl. Letjen T.B. Simatupang, Jakarta 12540, Indonesia

ARTICLE INFO

Keywords:
Camera-traps
Terrestrial wildlife
Forest intactness
Tree species composition
Biodiversity conservation

ABSTRACT

Logging has been operated throughout Bornean tropical forests that harbor diverse fauna species. We examined the responses of ground-dwelling wildlife to logging intensity in two certified forest management units under the same climate and vegetation in East Kalimantan, Indonesia, in 2012–2016. A total of ten and 13 circular plots with a diameter of 1 km were established respectively, and 10–18 camera points were distributed randomly in each plot to record wildlife for two years. A total of 41 species of medium to large-sized terrestrial wildlife were recorded in the two management units. Canonical Correspondence Analysis (CCA) was performed to examine the response of wildlife community to the logging influences with the following three variables: forest intactness (as a surrogate of tree-species composition), above-ground carbon (as a surrogate of forest structure), and distance to the nearest village to represent hunting pressure. Forest intactness was the strongest factor affecting the assemblage of wildlife species in both management units, suggesting that changes in tree-species composition due to logging were a decisive factor influencing animal assemblage. Poisson Generalized Linear Model (GLM) was employed to examine the response of each species to forest intactness in each management unit. We found that eight out of the 32 tested wildlife species responded to forest intactness consistently between the two units, while the other species responded more individualistically. We suggest that consistent responses of several species, as well as individualistic responses of the other species to the changes of tree-species composition, determine the animal assemblage in logged-over Bornean rainforests. Sustainably managed logged-over forests in Borneo are a habitat for diverse wildlife community and therefore also require attention for biodiversity conservation.

1. Introduction

Borneo has attracted significant investments in logging, which contributed to the deforestation and forest degradation during the last four decades (Fuller et al., 2004; Gaveau et al., 2014). Suitability of logged forests as wildlife habitat has long been a controversy (Johns, 1985). The effectiveness of logged forests for wildlife conservation, even under the sustainable mechanism such as reduced impact logging, has been doubted (Bowles et al., 1998). However, solely relying on primary forests for conserving the wildlife is not realistic. Protected areas in Kalimantan, which are aimed to protect primary forests, also suffer from deforestation (Fuller et al., 2004). Logged-over forests, on the other hand, may support populations of many rain-forest species. It may be unlikely that an area of logged forests will maintain all species formerly present, but a large area of logged-over forests may support species that would vanish from isolated primary forests (Johns, 1985). Therefore, it has been suggested that logged forests are better than non-forest vegetation to provide adequate habitat for Bornean wildlife (Meijaard and Sheil, 2007).

In Borneo, 67.5% of the areas where timber has been harvested since the 1970s remain as forest (Gaveau et al., 2014). Although they are still standing as forest, the structure of the remaining vegetation has been altered by intensive logging activities (Johns, 1997). Many studies have emphasized the prolonged logging effects on forest structure and tree species compositions. For example, eight years after the last logging event, the total basal area of selectively-logged forest was still only half of the unlogged one (Cannon et al., 1994). Selective logging opened the canopy of previously unlogged forests, which affects the microclimate of understory environment (Frumhoff, 1995). Tree species richness has been reported to be equal in primary vs. logged forests, but

* Corresponding author at: Perumahan Salatiga Permai No. 104, Kel. Blotongan, Kec. Sidorejo, Kodya Salatiga, Jawa Tengah 50715, Indonesia.
E-mail address: agussudibyojati@yahoo.co.id (A.S. Jati).

https://doi.org/10.1016/j.foreco.2018.05.054
Received 21 March 2018; Received in revised form 22 May 2018; Accepted 23 May 2018
Available online 05 June 2018
0378-1127/

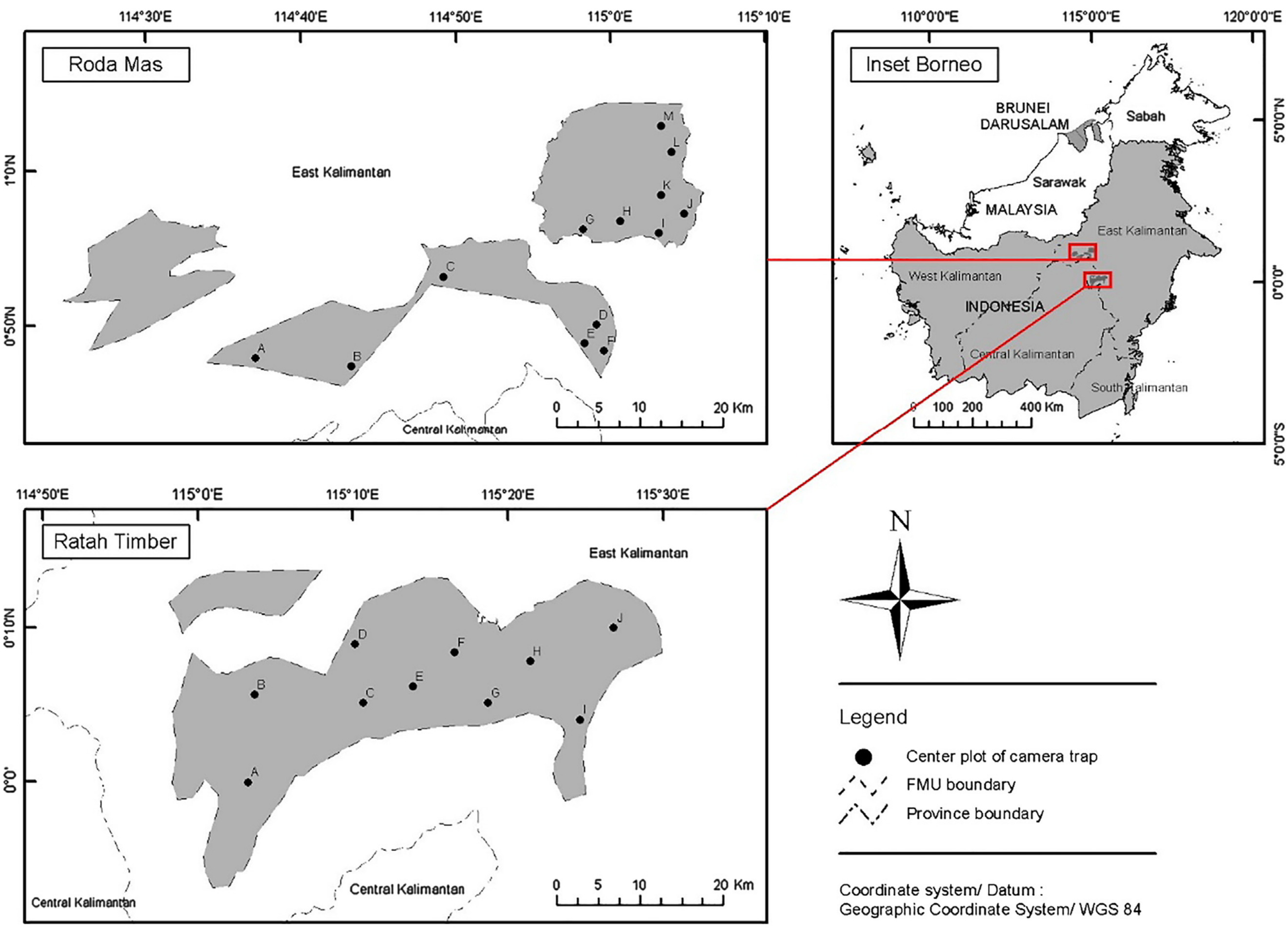

Fig. 1. Location of FMU Ratah Timber and Roda Mas with camera trap plot placements.

the species composition is shifted from shade-tolerant to pioneer species (Aoyagi et al., 2017; Imai et al., 2014, 2012). Canopy opening also stimulates the production of new leaves underneath the canopy as the result of increased light availability (Johns, 1988). Also, the road networks developed during logging increase the accessibility of human to the remote sites; adding potential disturbance to the forest (Kleinschroth and Healey, 2017).

Structural changes of logged forests alter wildlife habitat as well. Animals would respond to structural changes at different scales. Some species can be identified as specialists which highly depend on a particular condition; some others are generalists who can tolerate various conditions (Morrison et al., 2006; Pfeifer et al., 2017). Forest specialists showed a strong decline as the result of fragmentation, while less sensitive species may acclimate to modified habitats (Brodie et al., 2015; Pfeifer et al., 2017; Samejima et al., 2013). Floristic changes after logging may also affect animal assemblages. Different tree composition along a post-logging chronosequence provided different food resources and attracted different animals (DeWalt et al., 2003). Small frugivorous may find secondary forests more suitable than old-grown forests because more understory fruits are available in secondary forests (DeWalt et al., 2003). On the other hand, animals which depend on the specialist plants to undisturbed forests, such as palms, may show an opposite pattern (DeWalt et al., 2003). Because forests became more accessible to human, the abundance of wildlife, especially game species, may also be affected by increasing hunting pressure (Auzel and Wilkie, 1999; Bennett and Robinson, 1999). It has been reported that the abundance of game species decreased with decreasing distance to a nearby village (Muchaal and Ngandjui, 1999).

The impact of logging on wildlife community has been widely studied in Borneo. Most of those studies compared the abundance of particular wildlife between logged and unlogged forests (e.g., Brodie et al., 2015; Colón, 2002; Costantini et al., 2016; Granados et al., 2016; Heydon and Bulloh, 1996) or between conventional and reduced impact logging (Samejima et al., 2013). The heterogeneous physical condition of the landscape, such as topographic factor, makes forest structure highly variable among forest compartments even if they were logged in the same period (Cannon et al., 1994). The tree-species composition of logged-over forests may also greatly vary depending on the logging intensity, recovery time and topography (Bischoff et al., 2005). Direct human influences such as hunting pressure may also spatially, greatly vary within a given landscape. These together suggest that logging exerts complex influences on animal assemblages and a simple landscape-level comparison will not elucidate decisive factors of animal assemblages.

We here report the results of an intensive camera-trap study over a period of four years in natural production forests of Borneo. Our study aimed to explain how logging affected fauna species and communities in Bornean production forests with continuous variables of logging intensity but not just management histories (e.g. time elapsed after logging). We raised two factors, forest structure and tree-species composition, as influenced by logging and tested how these factors affected fauna species and communities. We used above-ground biomass as a surrogate of forest structure, and forest intactness index of Fujiki et al. (2016) as a surrogate of tree species composition. We also considered distance to the nearest village as a possible factor affecting wildlife distribution as it may represent hunting access. We tested the influences of these three factors (as continuous variables) on wildlife communities each in two forest management units (FMUs), East Kalimantan, Indonesia. We hypothesized that different wildlife communities are assembled in response to different forest conditions as the result of logging intensity. Both FMUs are certified by Forest Stewardship Council (FSC) and have similar logging history and management. Therefore, we considered these two FMUs as replicates and expected that wildlife communities assembled similarly in response to the same logging intensity between the two management units.

2. Methods

2.1. Study sites

We conducted this study in two FMUs of logging concession in East Kalimantan, Indonesia: Ratah Timber and Roda Mas (Fig. 1). Ratah Timber (0°7′S–0°13′N, 114°58′–115°30′E) occupies 93,425 ha of concession areas and Roda Mas (0°46′–1°05′N, 114°25′–115°06′E) 109,950 ha, both within the Mahakam River watershed. Conventional logging had been conducted in Ratah Timber since 1971, then reduce impact logging (RIL) is being implemented after 2011. Roda Mas performed conventional logging since 1973 and is implementing RIL since 2008. From 1971 to 2014, Ratah Timber had harvested 94,141 ha areas with an average volume of 29.03 m^3/ha (PT Ratah Timber, 2014). Roda Mas had logged 41,037 ha areas with an average volume of 37.3 m^3/ha from 1973 to 2006 (PT Roda Mas Timber Kalimantan, 2008). Both FMUs have been certified under the FSC-Forest Management scheme. Those FMUs are also certified under national certification, which is mandatory in Indonesia.

Forests of Ratah Timber and Roda Mas are both categorized as lowland mixed dipterocarp tropical forests. From the data of Aoyagi et al. (2017), tree community in Ratah Timber was dominated by the genera *Macaranga, Shorea, Syzygium, Litsea, Dipterocarpus, Elateriospermum, Hopea, Xanthophyllum*, and *Canarium*. While the genera *Shorea, Neolamarkia, Syzygium, Neonauclea, Macaranga, Dipterocarpus, Litsea*, and *Diospyros* dominate tree community in Roda Mas. Genus *Macaranga* is an indicator of the disturbed forest while the rest of them are intact forest indicators (Aoyagi et al., 2017).

2.2. Wildlife inventory

Wildlife inventory was conducted using camera traps. The usage of camera trap for studying wildlife has become increasingly popular for its advantages (Ancrenaz et al., 2012). The greatest benefit of using camera trap, in comparison with other sampling methods, is that it can record data without capturing animals or without the presence of researchers (O'Connell and Nichols, 2011). This ability to work independently with little or almost no nuisance from the presence of researchers makes it able to record shy and elusive species (O'Connell and Nichols, 2011; Trolliet et al., 2014). However, camera trap is ideally suited to sample medium to large sized ground-dwelling animals. Small animals, arboreal, or aquatic species might be difficult to be identified or well sampled (Ancrenaz et al., 2012). If the camera was installed in a remote area with less frequent visits by researchers, the risk of camera malfunctions also becomes greater (O'Connell and Nichols, 2011).

We used "Bushnell" camera traps which were equipped with infrared night photography and can take video or still picture. In Ratah Timber, ten circular plots, each with a 500-m radius, were established systematically in the entire area and 10 to 18 camera points were placed randomly in each plot. The mean (± standard deviations) distance between any nearest two plots was 8.13 (± 1.90) km and between any nearest camera points was 161.9 (± 74.10) m. For Roda Mas, 13 plots were similarly established, and ten random camera points were selected in each plot (Fig. 1). The mean (± standard deviations) distance between any nearest plots was 5.6 (± 4.3) km and between any nearest camera points was 160.3 (± 79.0) m. In total, there were 157 camera points deployed in Ratah Timber and 130 points in Roda Mas. Cameras were set facing downward or facing slope surface to optimize camera sensor and avoid too many false-triggered videos. Cameras were first deployed in Ratah Timber from June 2012 until July 2014. Cameras were set to take 10-seconds video when they were triggered. In Roda Mas cameras were installed in the forest from November 2014 to May 2016. For better animal identification, cameras in Roda Mas were set to take 30-seconds video for each recording event.

In this paper, we excluded plot E in Ratah Timber and plot K in Roda Mas (see Fig. 1 for the location) because those areas were logged during camera trapping periods. We also could not collect data from several camera units because they were malfunctioned or even missing. The trapping day of each camera was calculated from the first day it was installed until the day it was taken back. If the camera malfunctioned, batteries run out, or SD-card was full during the installation period, the trapping day was calculated only until the day it took the last video. In the end, we managed data from 129 camera units distributed in nine plots in Ratah Timber, with a total 19,502 trapping days, and 95 camera units distributed in 12 plots in Roda Mas with a total 27,426 trapping days.

All animals recorded by camera traps were identified and counted if possible. Because camera trap best suits to sample medium to large-sized terrestrial animals, only species fallen in that category were used for later analysis. Animals smaller than moonrat (*Echinosorex gymnura*) and crested partridge (*Rollulus rouloul*) were considered small animals. Small-sized and highly arboreal or aquatic species were excluded from the analysis even if they were recorded. Images that could not be identified to species were also excluded, except for mousedeer species (*Tragulus* spp.). Two *Tragulus* species existing in Borneo are difficult to distinguish from each other, and we treated them as one taxon in this paper. Following the assumption that the animals recorded within 30 min were the same individuals, only one video that contained the highest number of individuals was included if there were videos of similar species that were taken continuously within 30 min. We referred to IUCN Redlist and Criteria (https://www.iucnredlist.org) for taxonomic and conservation status of each species. If IUCN has not evaluated the species, we referred to Catalogue of Life (https://www.catalogueoflife.org) for its nomenclature.

To evaluate the relative abundance of each fauna species for each plot, we calculated mean trapping rate (MTR) per 1000 days by the following calculation:

$$MTR\ \ sp.\ \ i = \frac{\Sigma\ \ trapping\ \ rate\ \ sp.\ \ i}{\Sigma\ \ trapping\ \ day}\ \ x\ \ 1000$$

Trapping rate is the total number of individuals recorded by camera traps and trapping day is the total number of days when cameras worked properly. We combined data from all cameras in each circular plot to calculate MTR for statistical analysis. To compare relative abundance in FMU level, MTR was calculated from combined data of all cameras in each FMU.

2.3. Logging influences

Forest intactness (see below for the definition) and above-ground biomass have been studied to evaluate forest degradation in logged forest (Fujiki et al., 2016; Imai et al., 2014, 2012). Therefore, we used forest intactness as a surrogate of tree species composition and above-ground biomass as a surrogate of forest structure to represent forest degradation. Forest intactness was evaluated based on the ratio of pioneer and climax tree species of the tree community composition (Fujiki et al., 2016; Imai et al., 2014) which can also represent the forest succession after logging. We obtained information of forest intactness of Ratah Timber and Roda Mas (Fig. 2) from Fujiki et al. (2016). Fujiki et al. (2016) have created intactness maps of several FMUs in Borneo, including Ratah Timber and Roda Mas, by extrapolating non-metric multidimensional scaling (nMDS) axis-1 values of tree community on the Landsat satellite image (with 30 × 30 m pixel resolution) of the areas. One nMDS axis-1value was housed in each 30 × 30 m pixel as forest intactness index. A high nMDS axis-1 value on these maps (dark green color) means that tree genus composition in that pixel is similar with those in intact forests and a small value (red color) indicates the occurrence of more pioneer species, i.e., tree genus composition is similar with those of disturbed forests. For Ratah Timber, the intactness

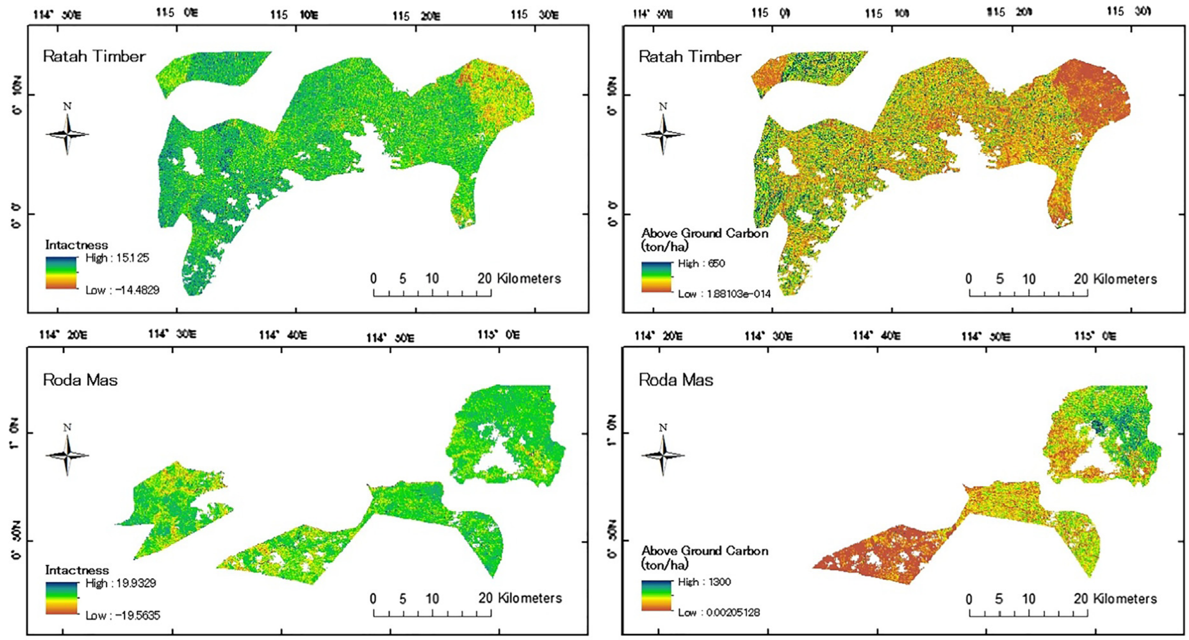

Fig. 2. Maps of forest intactness and above-ground carbon for FMU Ratah Timber (above) and Roda Mas (below) obtained from Fujiki et al. (2016) and Fujiki et al. (unpublished).

map was created from a 2010 satellite image, while Roda Mas from a 2015 satellite image. Detail method of the estimation of forest intactness is explained in Fujiki et al. (2016). We extracted the mean value of forest intactness over all pixels within a radius of 1 km from the plot center to represent the degree of forest intactness in each plot. The mean forest intactness (standardized nMDS axis-1 values) for the entire FMU was 0.220 for Ratah Timber and 0.289 for Roda Mas, and it was not significantly different between the FMUs.

Above-ground biomass (AGB) in this research was indexed with above-ground carbon (AGC) values based on the AGC maps of Fujiki et al. (unpublished; Fig. 2). Fujiki et al. (unpublished) made an AGC map of Ratah Timber by extrapolating AGC values of tree inventory plots (inventoried in 2012) on the 2010 satellite image after correcting for the biomass growth between 2010 and 2012. An AGC map of Roda Mas was produced by extrapolating AGC values of tree inventory plots (inventoried in 2015) on the 2015 satellite image. In their work, above-ground biomass (AGB) value was estimated from an allometric equation and was subsequently converted to above-ground carbon (Carvalho et al., 1995; Chave et al., 2005). Therefore, AGC can be used to represent AGB. An AGC value was housed in each pixel as forest intactness. Mean AGC value of each plot was extracted with the same method as forest intactness. In addition, the distance of each plot to the nearest village was also added to see if the distance limits hunting pressure from local villagers.

2.4. Response of wildlife community to logging influences

2.4.1. Community analysis

Canonical Correspondence Analysis (CCA) was employed to evaluate the response of wildlife to logging influences at a community level. For this analysis, we created an environmental matrix with nine plots and three variables (mean AGC per plot, mean forest intactness per plot, and the distance from a plot to the nearest village) for Ratah Timber and similarly with 12 plots and the three variables for Roda Mas. Based on the data recorded by camera traps, we created a wildlife community matrix consisting of nine plots and 36 species for Ratah Timber and 12 plots and 38 species for Roda Mas; both with MTR as species abundances. CCA was applied to these environmental matrix and wildlife community matrix. CCA was conducted for each FMU with a backward-stepwise procedure to select variables that can best represent relationships between wildlife community and the three variables. P-value and Akaike Information Criterion (AIC) of each variable at each step were compared to evaluate which variable can best explain this relationship.

Spearman's rank correlation was also run to test significant relationships between CCA axis-1 values of fauna species between the two FMUs. This test was conducted only for the shared fauna species between the two FMUs to examine if fauna species assembled similarly along the major gradient (i.e.,CCA axis-1). These analyses were run with the statistical software "R" version 3.3.1 under "vegan" package (Oksanen et al., 2018).

2.4.2. Relationships between species with logging influences

Generalized Linear Model (GLM) analysis with a Poisson error distribution was employed to evaluate the response of species abundance to the logging influences. We used the trapping rate of each species per plot as the dependent variable and the selected strongest or significant variable of logging influences in the CCA stepwise analysis as the independent variable. All species with trapping rate more than 30 individuals were analyzed. A total of 17 species were analyzed for Ratah Timber and 20 species for Roda Mas, of which 15 species were shared between the two FMUs. Because the total numbers of trapping day were different among plots, total trapping day per plot was included as a co-variate (offset term) in the model, so there is no need to control the response variable by dividing them by total trapping day. The response of each species towards logging influences was evaluated from its p-value and direction (positive or negative) of the coefficient. If the value is significant, the variable of logging intensity is significantly correlated with the abundance of a given species. Poisson GLM was conducted on fauna species in Ratah Timber and Roda Mas separately, and the results were compared between the two FMUs to investigate if the directions of the responses of fauna species to increasing logging influences were the same. This analysis was run with statistical software "R" version 3.3.1 under "stats" package (R Core Team, 2017).

3. Results

3.1. Species records

In total there were 75 fauna species identified from camera trap video in the two FMUs (Appendix A). Based on IUCN Redlist & Criteria, 18 species were listed as threatened by extinction (vulnerable, endangered, and critically endangered); 53 species not threatened by extinction (least concern and near threatened); one species does not have enough data to be categorized (data deficient); and three species have not been evaluated. Among them, 12 species are endemic to Borneo Island. Small-sized animals such as rats, small squirrels, and small birds were frequently recorded; yet because of clarity of the images, only relatively few individuals were successfully identified. Some highly arboreal and aquatic species, such as white-fronted surili (*Presbytis frontata*) and small-clawed otter (*Aonyx cinereus*), were also recorded. Because only a few proportions of small animals were identified and highly arboreal and aquatic species infrequently appeared in the terrestrial environment, such species were not well sampled, and their relative abundances could not be adequately estimated.

In Ratah Timber, there were 37 species of medium to large-sized terrestrial animals, consisting of six birds, 28 mammals, and three reptiles. In Roda Mas, there were 39 species in the same category, consisting of six birds, 31 mammals, and two reptiles (Table 1). Roda Mas and Ratah Timber shared 33 species. Five species were recorded only in Roda Mas and three species only in Ratah Timber. Mousedeers (*Tragulus* spp.) was the most abundant species in Ratah Timber (MTR = 47.97) followed by pig-tailed macaque (*Macaca nemestrina*; MTR = 45.76), while Malay weasel (*Mustela nudipes*) and water monitor (*Varanus salvator*) were the fewest (MTR = 0.05). In Roda Mas, bearded pig (*Sus barbatus*) was the most abundant species (MTR = 71.25) followed by pig-tailed macaque (MTR = 62.20). Otter civet was the rarest species recorded in Roda Mas (MTR = 0.04). For a quick comparison of species abundance, see Appendix B.

3.2. Response of wildlife community to logging influences

Canonical Component Analysis (CCA) of wildlife community and logging influences was run at three steps for both study sites (Table 2). The variables selected were consistent between the FMUs. In both FMUs, above ground carbon (AGC) was removed at step 2 and distance to nearest village was removed at step 3. Only forest intactness was selected for the last CCA model for both Ratah Timber and Roda Mas. The AIC value of forest intactness at the last step was the lowest compared to the other variables in both FMUs. Final CCA ordination diagrams are demonstrated with forest intactness in Fig. 3. Forest intactness of step 3 significantly

Table 1
Number of medium to large-sized terrestrial fauna species per family recorded with camera traps in two FMUs.

Class	Family	Number of Species	
		Ratah Timber	Roda Mas
Birds	*Phasianidae* (pheasants and relatives)	5	5
	Cuculidae (cuckoos)	1	1
Mammals	*Cercopithecidae* (monkeys)	2	2
	Cervidae (deer and relatives)	3	3
	Erinaceidae (moonrat)	1	0
	Felidae (cats)	4	5
	Herpestidae (mongooses)	2	2
	Hystricidae (porcupines)	2	3
	Manidae (pangolin)	1	1
	Mustelidae (weasels and relatives)	2	2
	Prionodontidae (linsang)	1	1
	Sciuridae (squirrels)	1	1
	Suidae (pig)	1	1
	Tragulidae (mousedeers)*	2	2
	Ursidae (bear)	1	1
	Viverridae (civets and relatives)	5	7
Reptiles	*Varanidae* (monitor lizards)	3	2
	Total species	37	39

* Two mousedeer species were identified in both FMUs but they were considered as a single species in the analyses.

Table 2
Summary of backward-stepwise CCA on FMU Ratah Timber and Roda Mas wildlife community.

Ratah Timber			Roda Mas		
Variable	AIC	p-value	Variable	AIC	p-value
Step 1			Step 1		
AGC	48.798	0.77⁻	AGC	66.178	0.425⁻
Intactness	48.968	0.665	Intactness	66.5	0.22
Near village	49.343	0.47	Near village	67.027	0.055
Step 2			Step 2		
Near village	48.114	0.54⁻	Near village	65.811	0.225⁻
Intactness	49.006	0.15	Intactness	66.136	0.105
Step 3			Step 3		
Intactness	48.056	0.07	Intactness	65.783	0.045

Note:
AGC = above ground carbon; Intactness = forest intactness; Near village = distance to nearest village.
−: Removed at the next step.

correlated with wildlife communities in Roda Mas (p = 0.045), while it marginally significantly correlated with wildlife communities in Ratah Timber (p = 0.07). Spearman's rank correlation of the values of CCA axis-1 for the shared fauna species between the two FMUs was low ($r_s = 0.203$, $p > 0.05$, data not shown).

Following the selection of variables in CCA, Poisson Generalized Linear Model (GLM) was run for each species with forest intactness as predictor variable. Forest intactness significantly explained the abundance of ten fauna species in Ratah Timber and 15 species in Roda Mas. For Ratah Timber, eight species showed a significant positive response and two had a significant negative response; seven did not have a significant response. For Roda Mas, seven species had a significant positive response and eight had a significant negative response; five were not significant.

Table 3 shows the comparison of the responses of fauna species to forest intactness between the two FMUs. Statistical summaries of Poisson GLM for each fauna species in Ratah Timber and Roda Mas are presented in Appendix C. Eight species showed a consistent response to the forest intactness between the two FMUs: five species (*Carpococcyx radiceus, Muntiacus atherodes, Rollulus rouloul, Sus barbatus* and *Viverra tangalunga*) had a significant positive response (i.e., increasing in more intact forests); one species (*Macaca nemestrina*) had a significant negative response (i.e., increasing in more degraded forests); and two species (*Hystrix brachyura* and *Trichys fasciculata*) did not respond significantly. Seven species showed different statistical results: one species (*Lophura ignita*) did not show a significant response in Ratah Timber whereas it had a significant positive response in Roda Mas; three species (*Hemigalus derbyanus, Muntiacus muntjak* and *Rusa unicolor*) had no significant response in Ratah Timber whereas they had a significant negative response in Roda Mas; and three species (*Argusianus argus, Helarctos malayanus,* and *Tragulus* spp.) responded positively in Ratah Timber whereas they did not have a significant response in Roda Mas. Two species in Ratah Timber, one with a negative response (*Echinosorex gymnura*) and the other without (*Paradoxurus hermaphroditus*), were not tested for Roda Mas. Five species which were tested for Roda Mas but not in Ratah Timber had responses as follows: *Lophura bulweri* had a positive response while *Herpestes brachyurus, Hystrix crassispinis, Martes flavigula* and *Paguma larvata* had a negative response.

4. Discussion

Forest intactness as the surrogate of tree species composition was suggested to most strongly relate to wildlife community in our logged-over forests. AGC as the surrogate of forest structure was not selected as an important factor in our analysis. However, previous studies

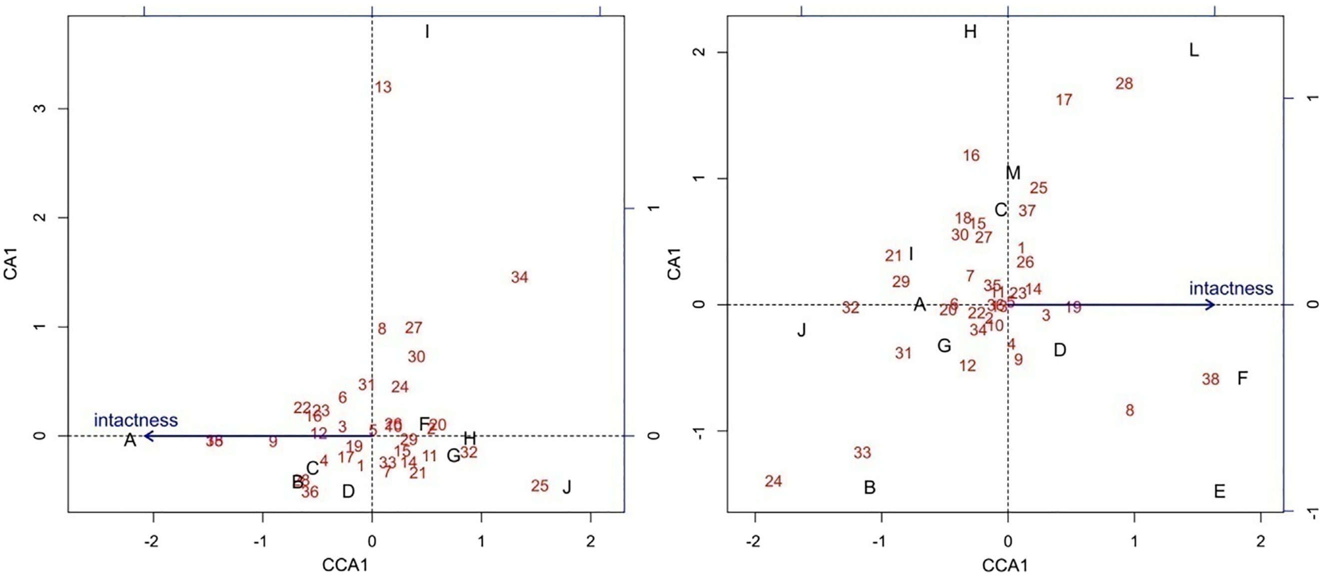

Fig. 3. CCA ordination diagrams of wildlife community in FMU Ratah Timber (left) and Roda Mas (right). The numbers represent fauna species (refer to Appendix B for species number), capital letters represent camera plots, and the arrow represents forest intactness. Arrowhead indicates the direction of increase in forest intactness and arrow length represents its relative influence.

Table 3
Response of each species to the degree of forest intactness derived from the Poisson GLM analysis.

Species	Response	
	Roda Mas	Ratah Timber
Carpococcyx radiceus	+	+
Muntiacus atherodes	+	+
Rollulus rouloul	+	+
Sus barbatus	+	+
Viverra tangalunga	+	+
Macaca nemestrina	−	−
Hystrix brachyura	=	=
Trichys fasciculata	=	=
Lophura ignita	+	=
Hemigalus derbyanus	−	=
Muntiacus muntjak	−	=
Rusa unicolor	−	=
Argusianus argus	=	+
Helarctos malayanus	=	+
Tragulus spp.	=	+
Lophura bulweri	+	x
Herpestes brachyurus	−	x
Hystrix crassispinis	−	x
Martes flavigula	−	x
Paguma larvata	−	x
Echinosorex gymnura	x	−
Paradoxurus hermaphroditus	x	=

+: significant, positive response.
−: significant, negative response.
=: no significant response.
x: not tested (trap-rates less than 30 individuals).

highlighted forest structure as a factor affecting the abundance of some particular animals in the logged forest. Areas with a high tree cover have been reported important for arboreal or semi-arboreal species like birds and primates (Harvey et al., 2004; Meijaard and Sheil, 2008; Peh et al., 2005). It is likely that AGC was less important in our research because we analyzed terrestrial species which do not depend on canopy coverage to move across the landscape. The selection of forest intactness (tree species composition) suggests that the availability of food resources is more important to determine animal communities in our sites. Different tree composition provides different resources for animals, which affects their distribution across our forests.

The low rank-correlation value of CCA axis-1 between animal communities of the two FMUs indicates that wildlife community assembled differently between the two forest management units in response to forest intactness against our hypothesis, although forest intactness explained the largest variation of animal communities in both FMUs. In the species-level analysis, eight species had consistent responses to forest intactness between the two FMUs, while seven species demonstrated different responses (Table 3). Food availability seems to be the main determinant of these responses. Leaf litter production varied according to the tree composition and accumulated better in a more intact forest (Scherer-Lorenzen et al., 2007; Sundarapandian and Swamy, 1999) which then attracts more invertebrates (Burghouts et al., 1992; Negrete-Yankelevich et al., 2007). Invertebrates are food sources for *Carpococcyx radiceus, Rollulus rouloul, Sus barbatus* and *Viverra tangalunga,* which explains why those species consistently respond positively towards forest intactness. The *Sus barbatus* has also been reported to follow dipterocarp distributions to prey on its seeds (Curran and Webb, 2000). In contrast, *Macaca nemestrina* consistently showed a negative response towards forest intactness because it eats understory fruits which are more abundant in degraded forest (Costa and Magnusson, 2003; DeWalt et al., 2003; Frumhoff, 1995; Meijaard and Sheil, 2008). However, other species may be able to adapt to the changes caused by logging and can persist both in intact and degraded forests (Peh et al., 2005).

Seven species showed different responses in our two FMUs. Inconsistent responses are also seen when comparing our results with the results of previous studies (Table 4). These inconsistent individualistic responses towards logging intensity might be caused by unmeasured local circumstances such as hunting pressure (Hart, 1999; Reyna-Hurtado and Tanner, 2007; Taper et al., 1995), interspecific interactions (Berger and Gese, 2007; Taper et al., 1995), niche availability (Taper et al., 1995) or landscape configurations (Mazerolle and Villard, 1999). However, it is also possible that different responses showed in Table 4 reflect the inconsistent research designs among previous studies. The detection of inconsistent individualistic responses of animal species between the two FMUs does not necessarily mean that the species do not respond to logging (Taper et al., 1995). It is generally accepted that forest degradation threatens the forest fauna. Forest intactness which was derived from tree species composition has been reported as a robust indicator to evaluate logging disturbances (Fujiki et al., 2016; Imai et al., 2014). Therefore, we suggest that consistent responses of several species, as well as individualistic responses of the other species to the changes of tree-species composition, determine the

Table 4
Comparison of fauna species responses tested in this study ([1] in Ratah Timber; [2] in Roda Mas) with other similar studies (in the parentheses) towards logging.

Group	Species	Response		
		Positive	Negative	Not significant
Terrestrial birds	*Argusianus argus*	This study[1](Peh et al., 2005; Samejima et al., 2013; Wilson and Johns, 1982)		This study[2](Samejima et al., 2012)
	Carpococcyx radiceus	This study[1,2]		
	Lophura bulweri	This study[2]		
	Lophura ignita	This study[2]		This study[1](Samejima et al., 2012)
	Rollulus rouloul	This study[1,2](Peh et al., 2005)		
Porcupines	*Hystrix brachyura*			This study[1,2](Brodie et al., 2015; Samejima et al., 2012)
	Hystrix crassispinis	(Samejima et al., 2012)	This study[2]	
	Trichys fasciculata			This study[1,2](Samejima et al., 2012)
Ungulates	*Muntiacus atherodes*	This study[1,2](Brodie et al., 2015; Granados et al., 2016; Samejima et al., 2013)		(Samejima et al., 2012)
	Muntiacus muntjak	(Brodie et al., 2015)	This study[2]	This study[1]
	Rusa unicolor		This study[2](Brodie et al., 2015)	This study1 (Samejima et al., 2012)
	Sus barbatus	This study[1,2](Wilson and Johns, 1982)		(Brodie et al., 2015; Granados et al., 2016; Samejima et al., 2012)
	Tragulus spp.	This study[1]		this study[2](Brodie et al., 2015; Granados et al., 2016; Samejima et al., 2012)
Civets	*Hemigalus derbyanus*	(Heydon and Bulloh, 1996; Samejima et al., 2013)	This study[2]	this study[1](Brodie et al., 2015; Samejima et al., 2012)
	Paguma larvata	(Brodie et al., 2015)	This study[2]	
	Paradoxurus hermaphroditus	(Heydon and Bulloh, 1996; Samejima et al., 2013)	(Brodie et al., 2015)	This study[1](Samejima et al., 2012)
	Viverra tangalunga	this study[1,2](Brodie et al., 2015; Colón, 2002; Heydon and Bulloh, 1996; Samejima et al., 2013)	(Syakirah et al., 2000)	(Granados et al., 2016; Samejima et al., 2012)
Primate	*Macaca nemestrina*		This study[1,2]	(Brodie et al., 2015; Granados et al., 2016; Johns, 1986; Samejima et al., 2012)
Bear	*Helarctos malayanus*	this study[1](Samejima et al., 2013; Wilson and Johns, 1982)	(Samejima et al., 2012)	This study[2](Brodie et al., 2015)
Weasel	*Martes flavigula*	(Brodie et al., 2015)	This study[2]	
Mongoose	*Herpestes brachyurus*	(Samejima et al., 2013)	This study[2]	(Brodie et al., 2015; Samejima et al., 2012)
Moonrat	*Echinosorex gymnura*		This study[1]	(Samejima et al., 2012)

animal assemblage in the logged-over rainforests of the two FMUs.

In our study, the distance to the nearest village was not selected on the final model of CCA, indicating that this variable was less important than forest intactness for wildlife community in our forests. Hunting was suspected to affect wildlife population more than logging itself in many logged-over forests (Frumhoff, 1995; Kleinschroth and Healey, 2017; Meijaard and Sheil, 2008; Poulsen et al., 2011; Wilson and Johns, 1982). Hunting could also be an important factor affecting animal communities in our study sites. Probably, the distance to the village was less significant because hunting pressure was not related with distance to the nearest village. The creation of road network may facilitate hunters to explore and transport the game out of the forest (Kleinschroth and Healey, 2017; Wilkie et al., 2000; Wilson and Johns, 1982), which may allow them to hunt far away from the village. However, a specific investigation on this issue is necessary to understand how hunting affected wildlife in our study sites.

Total species recorded in this study (Appendix A) shows that FSC-certified logged-over forests in East Kalimantan are still able to support a high diversity of wildlife. The number of endemic species and endangered species identified in this study emphasizes the high conservation value possessed by tropical forests which have been harvested for timber for decades. Furthermore, some elusive species were also recorded, including Hose's palm civet (*Diplogale hosei*) in Roda Mas, which is the second record of this species in Indonesia (first record by Samejima and Semiadi (2012)). Compared to previous camera-trap studies in Borneo's protected areas (Azlan and Lading, 2006; Bernard et al., 2013), the number of mammalian species is richer in our study, although this gap is likely due to the result of different sampling design and trapping efforts.

It has been shown in this study, as well as in the previous other studies, that logged-over forests are habitats for wildlife communities. Because forests managed for timber cover a vast area of tropical forests in Kalimantan, concern on biodiversity conservation would be more reasonable if not only given to designated protected forests, but also to the concession forests which are harvested continuously. Since the main purpose of such forests is timber production, the impacts of logging activities on native wildlife are rarely assessed by forest managers. As long as concession areas are still standing as natural forests (though degraded), they still provide better habitats for wildlife than nonforest (Meijaard and Sheil, 2007) and could serve as corridors among the scattered protected forests. Therefore, we urge the practice of sustainable forest management to prevent the shrinkage of forest cover, especially in production forests. Understanding how logging intensity affects wildlife community will give sufficient information to explore the importance of wildlife conservation in logged-over forests and enhance the development of sustainable forest management.

Acknowledgments

This study was supported by the Global Environment Research Fund D-1006 and 1-1403 of the Ministry of the Environment, Japan, and by the UNU GGS program fund to K.K, and was conducted in close collaboration with WWF-Indonesia and WWF-Japan. We thank Mr. Wahyul Ikhsani, Mr. Tri Sudjatmiko, Mr. Wayan Sujana and other managerial and field staffs of PT. Ratah Timber and PT. Roda Mas for allowing us to conduct and assisted this research in their concession areas. We thank Mr. Sarjuni (ALeRT), Mr. Arif Data Kusuma (GGGI) and Mr. Mutai Hashimoto (WWF-Japan) for their kind assistance and collaboration during our fieldwork. We thank Mrs. Hidemi Tatsuno (Kyoto University) for her help in data input.

Appendix A

See Table A.1.

Table A.1
Checklist of fauna species identified from camera-traps deployed in FMU Ratah Timber and Roda Mas.

No.	Scientific name		English name	IUCN Status	Borneo endemism	Identified in	
						Ratah Timber	Roda Mas
1	Bird	*Accipiter trivirgatus*	Crested goshawk	LC		●	
2		*Alophoixus phaeocephalus*	Yellow-bellied bulbul	LC			●
3		*Arachnothera longirostra*	Little spiderhunter	LC			●
4		*Argusianus argus*	Great argus	NT		●	●
5		*Carpococcyx radiceus*	Bornean ground cuckoo	NT	●	●	●
6		*Chalcophaps indica*	Emerald dove	LC		●	●
7		*Copsychus malabaricus*	White-rumped shama	NE			●
8		*Erythropitta granatina*	Garnet pitta	NT		●	
9		*Geokichla interpres*	Chestnut-capped Thrush	NT			●
10		*Hydrornis baudii*	Blue-headed Pitta	Vu	●	●	●
11		*Hydrornis caeruleus*	Giant pitta	NT			●
12		*Hypothymis azurea*	Black-naped blue flycatcher	LC			●
13		*Larvivora cyane*	Siberian Blue Robin	LC			●
14		*Lophura bulweri*	Bulwer's pheasant	Vu	●		●
15		*Lophura erythrophthalma*	Crestless fireback	Vu		●	●
16		*Lophura ignita*	Crested fireback	NT		●	●
17		*Macronous ptilosus*	Fluffy-backed tit-babbler	LC			●
18		*Malacocincla sepiaria*	Horsfield's Babbler	LC			●
19		*Pellorneum capistratum*	Black-capped babbler	LC			●
20		*Pitta moluccensis*	Blue-winged Pitta	LC			●
21		*Polyplectron schleiermacheri*	Bornean peacock-pheasant	En	●	●	
22		*Rhyticeros undulatus*	Wreathed Hornbill	LC			●
23		*Rollulus rouloul*	Crested partridge	NT		●	●
24		*Sasia abnormis*	Rufous piculet	LC			●
25		*Spilornis cheela*	Crested Serpent Eagle	LC			●
26		*Trichastoma bicolor*	Ferruginous babbler	LC			●
27	Mammal	*Aonyx cinereus*	Oriental small-clawed otter	Vu		●	
28		*Arctictis binturong*	Binturong	Vu		●	●
29		*Callosciurus* cf. *notatus*	Plantain Squirrel	LC			●
30		*Callosciurus* cf. *prevostii*	Prevost's squirrel	LC			●
31		*Catopuma badia*	Borneo bay cat	En	●	●	●
32		*Cynogale bennettii*	Otter civet	En			●
33		*Diplogale hosei*	Hose's palm civet	Vu	●		●
34		*Echinosorex gymnura*	Moonrat	LC		●	
35		*Helarctos malayanus*	Sun bear	Vu		●	●
36		*Hemigalus derbyanus*	Banded palm civet	NT		●	●
37		*Herpestes brachyurus*	Short-tailed mongoose	NT		●	●
38		*Herpestes semitorquatus*	Collared mongoose	NT		●	●
39		*Hystrix brachyura*	Malayan porcupine	LC		●	●
40		*Hystrix crassispinis*	Thick-spined porcupine	LC	●		●
41		*Lariscus insignis*	Three-striped ground squirrel	LC			●
42		*Macaca fascicularis*	Long-tailed macaque	LC		●	●
43		*Macaca nemestrina*	Pig-tailed macaque	LC		●	●
44		*Manis javanica*	Sunda pangolin	CE		●	●
45		*Martes flavigula*	Yellow-throated Marten	LC		●	●
46		*Muntiacus atherodes*	Bornean yellow muntjac	NT	●	●	●
47		*Muntiacus muntjak*	Red muntjac	LC		●	●
48		*Mustela nudipes*	Malay weasel	LC		●	●
49		*Nannosciurus melanotis*	Black-eared squirrel	LC			●
50		*Neofelis diardi*	Clouded leopard	Vu		●	●
51		*Paguma larvata*	Masked palm civet	LC		●	●
52		*Paradoxurus hermaphroditus*	Common palm civet	LC		●	●
53		*Pardofelis marmorata*	Marbled cat	NT		●	●
54		*Presbytis frontata*	White-fronted surili	Vu	●	●	●
55		*Presbytis rubicunda*	Red leaf monkey	LC			●
56		*Prionailurus bengalensis*	Leopard cat	LC		●	●
57		*Prionailurus planiceps*	Flat-headed cat	En			●
58		*Prionodon linsang*	Banded linsang	LC		●	●
59		*Ptilocercus lowii*	Pen-tailed treeshrew	LC		●	
60		*Rheithrosciurus macrotis*	Tufted Ground Squirrel	Vu	●	●	●
61		*Rusa unicolor*	Sambar	Vu		●	●
62		*Sundasciurus hippurus*	Horse-tailed squirrel	NT		●	●
63		*Sundasciurus lowii*	Low's squirrel	LC			●
64		*Sus barbatus*	Bornean bearded pig	Vu		●	●
65		*Tarsius bancanus*	Horsfield’s Tarsier	Vu			●
66		*Tragulus kanchil*	Lesser mouse-deer	LC		●	●
67		*Tragulus napu*	Greater mouse-deer	LC		●	●

(*continued on next page*)

Table A.1 (*continued*)

No.	Scientific name		English name	IUCN Status	Borneo endemism	Identified in	
						Ratah Timber	Roda Mas
68		*Trichys fasciculata*	Long-tailed porcupine	LC		●	●
69		*Tupaia dorsalis*	Striped treeshrew	DD	●		●
70		*Tupaia* cf. *picta*	Painted treeshrew	LC	●		●
71		*Tupaia tana*	Large treeshrew	LC			●
72		*Viverra tangalunga*	Malayan civet	LC		●	●
73	Reptile	*Varanus dumerilii*	Dumeril's monitor	NE		●	
74		*Varanus rudicollis*	Black Roughneck Monitor	NE		●	●
75		*Varanus salvator*	Water monitor	LC		●	●

IUCN category abbreviation: NE, Not evaluated; NT, Near threatened; CE, Critically endangered; DD, Data deficient; Vu, Vulnerable; LC, Least concern; En, Endangered.

Appendix B

See Fig. B.1.

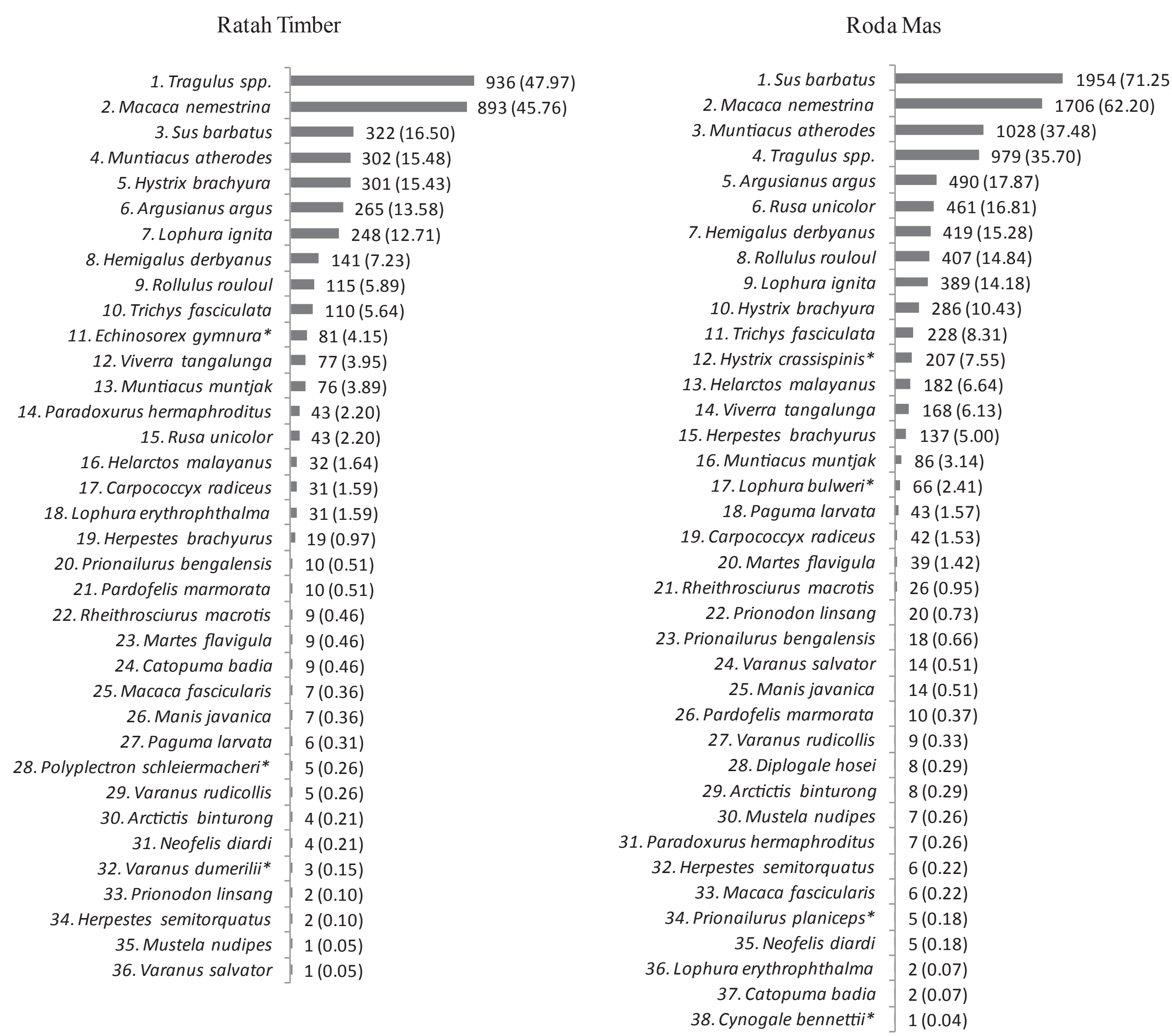

Fig. B.1. Trap-rates of medium to large-sized terrestrial fauna species recorded in Ratah Timber (left) and Roda Mas (right). Numbers indicate total numbers of individual per species per FMU and numbers in the parentheses are MTR per species. Species names followed by one asterisk symbol (*) were only identified in one FMU.

Appendix C

See Tables C.1 and C.2.

Table C.1
Summary of Poisson GLM between each fauna species and forest intactness of Ratah Timber. Intercept values are not shown.

Species	Coefficient	Std. error	p-value
Argusianus argus	0.951	0.164	< 0.001
Carpococcyx radiceus	1.019	0.485	0.036
Echinosorex gymnura	-0.925	0.223	< 0.001
Helarctos malayanus	1.998	0.542	< 0.001
Hemigalus derbyanus	-0.012	0.192	0.950
Hystrix brachyura	0.266	0.138	0.054
Lophura ignita	-0.268	0.139	0.054
Macaca nemestrina	-0.920	0.067	< 0.001
Muntiacus atherodes	1.540	0.167	< 0.001
Muntiacus muntjak	0.031	0.264	0.906
Paradoxurus hermaphroditus	-0.573	0.319	0.073
Rollulus rouloul	3.951	0.363	< 0.001
Rusa unicolor	-0.522	0.321	0.104
Sus barbatus	0.952	0.149	< 0.001
Tragulus spp.	0.401	0.080	< 0.001
Trichys fasciculata	-0.172	0.212	0.417
Viverra tangalunga	1.798	0.341	< 0.001

Table C.2
Summary of Poisson GLM between each fauna species and forest intactness of Roda Mas. Intercept values are not shown.

Species	Coefficient	Std. Error	p-value
Argusianus argus	0.115	0.143	0.422
Carpococcyx radiceus	1.671	0.557	0.003
Helarctos malayanus	-0.118	0.232	0.611
Hemigalus derbyanus	-0.937	0.149	< 0.001
Herpestes brachyurus	-0.847	0.261	0.001
Hystrix brachyura	-0.161	0.184	0.381
Hystrix crassispinis	-0.835	0.213	< 0.001
Lophura bulweri	1.073	0.418	0.010
Lophura ignita	0.481	0.164	0.003
Macaca nemestrina	-0.317	0.075	< 0.001
Martes flavigula	-1.247	0.490	0.011
Muntiacus atherodes	1.003	0.105	< 0.001
Muntiacus muntjak	-1.074	0.329	0.001
Paguma larvata	-1.085	0.466	0.020
Rollulus rouloul	4.026	0.240	< 0.001
Rusa unicolor	-1.088	0.142	< 0.001
Sus barbatus	0.264	0.072	< 0.001
Tragulus spp.	0.017	0.101	0.867
Trichys fasciculata	-0.219	0.206	0.288
Viverra tangalunga	0.492	0.250	0.049

References

Ancrenaz, M., Hearn, A.J., Ross, J., Sollman, R., Wilting, A., 2012. Handbook for Wildlife Monitoring using Camera-traps. BBEC Publication. JC Printer, Kinabalu. https://doi.org/10.1134/1.558554.

Aoyagi, R., Imai, N., Fujiki, S., Sugau, J.B., Pereira, J.T., Kitayama, K., 2017. The mixing ratio of tree functional groups as a new index for biodiversity monitoring in Bornean production forests. For. Ecol. Manage. 403, 27–43. http://dx.doi.org/10.1016/j.foreco.2017.07.026.

Auzel, P., Wilkie, D., 1999. Wildlife use in northern Congo: hunting in a commercial logging concession. In: Robinson, J.G. (Ed.), Hunting for Sustainability in Tropical Forests. Columbia University Press, New York, pp. 413–426.

Azlan, M., Lading, E., 2006. Camera trapping and conservation in Lambir Hills National Park, Sarawak. Raffles Bull. Zool. 54, 469–475.

Bennett, E.L., Robinson, J.G., 1999. Hunting for sustainability: the start of a synthesis. In: Robinson, J.G., Bennett, E.L. (Eds.), Hunting for Sustainability in Tropical Forests. Columbia University Press, New York, pp. 499–519.

Berger, K.M., Gese, E.M., 2007. Does interference competition with wolves limit the distribution and abundance of coyotes? J. Anim. Ecol. 76, 1075–1085. http://dx.doi.org/10.1111/j.1365-2656.2007.01287.x.

Bernard, H., Ahmad, A.H., Brodie, J., Giordano, A.J., Lakim, M.B., Amat, R., Hue, S.K.P., Khee, L.S., Tuuga, A., Malim, P.T., Lim-Hasegawa, D., Wai, Y.S., Sinun, W., 2013. Camera trapping survey of mammals in and around Imbak Canyon Conservation Area in Sabah, Malaysian Borneo. Raffles Bull. Zool. 61, 409–418.

Bischoff, W., Newbery, D.M., Lingenfelder, M., Schnaeckel, R., Petol, G.H., Madani, L., Ridsdale, C.E., 2005. Secondary succession and dipterocarp recruitment in Bornean rain forest after logging. For. Ecol. Manage. 218, 174–192. http://dx.doi.org/10.1016/j.foreco.2005.07.009.

Bowles, I.A., Rice, R.E., Mittermeier, R.A., Fonseca, G.A.B. da, 1998. Logging and tropical forest conservation. Science (80-) 280, 1899–1900. https://doi.org/10.1126/science.280.5371.1899.

Brodie, J.F., Giordano, A.J., Ambu, L., 2015. Differential responses of large mammals to logging and edge effects. Mamm. Biol. 80, 7–13. http://dx.doi.org/10.1016/j.mambio.2014.06.001.

Burghouts, T., Ernsting, G., Korthals, G., Vries, T.D., 1992. Litterfall, Leaf Litter Decomposition and Litter Invertebrates in Primary and Selectively Logged Dipterocarp Forest in Sabah, Malaysia. Philos. Trans. R. Soc. B Biol. Sci. 335, 407–416. http://dx.doi.org/10.1098/rstb.1992.0032.

Cannon, C., Kartawinata, K., Leighton, M., Peart David, R., 1994. The structure of lowland rainforest after selective logging in West Kalimantan, Indonesia. For. Ecol. Manage. 67, 49–68. http://dx.doi.org/10.1016/0378-1127(94)90007-8.

Carvalho Jr, J.A., Santos, J.M., Santos, J.C., Leitão, M.M., Higuchi, N., 1995. A tropical rainforest clearing experiment by biomass burning in the Manaus region. Atmos. Environ. 29, 2301–2309. http://dx.doi.org/10.1016/1352-2310(95)00094-F.

Chave, J., Andalo, C., Brown, S., Cairns, M.A., Chambers, J.Q., Eamus, D., Fölster, H., Fromard, F., Higuchi, N., Kira, T., Lescure, J.P., Nelson, B.W., Ogawa, H., Puig, H., Riéra, B., Yamakura, T., 2005. Tree allometry and improved estimation of carbon stocks and balance in tropical forests. Oecologia 145, 87–99. http://dx.doi.org/10.1007/s00442-005-0100-x.

Colón, C.P., 2002. Ranging behaviour and activity of the Malay civet (Viverra tangalunga) in a logged and an unlogged forest in Danum Valley, East Malaysia. J. Zool. L. 257, 473–485. http://dx.doi.org/10.1017/S0952836902001073.

Costa, F.R.C., Magnusson, W.E., 2003. Effects of selective logging on the diversity and abundance of flowering and fruiting understory plants in a Central Amazonian forest. Biotropica 35, 103–114. http://dx.doi.org/10.1111/j.1744-7429.2003.tb00267.x.

Costantini, D., Edwards, D.P., Simons, M.J.P., 2016. Life after logging in tropical forests of Borneo: a meta-analysis. Biol. Conserv. 196, 182–188. http://dx.doi.org/10.1016/j.biocon.2016.02.020.

Curran, L.M., Webb, C.O., 2000. Experimental tests of the spatiotemporal scale of seed predation in mast-fruiting dipterocarpaceae. Ecol. Monogr. 70, 129–148. http://dx.doi.org/10.1890/0012-9615(2000) 070[0129:ETOTSS]2.0.CO;2.

DeWalt, S.J., Maliakal, S.K., Denslow, J.S., 2003. Changes in vegetation structure and composition along a tropical forest chronosequence: Implications for wildlife. For. Ecol. Manage. 182, 139–151. http://dx.doi.org/10.1016/S0378-1127(03)00029-X.

Frumhoff, P.C., 1995. Conserving wildlife in tropical forests managed for timber. Bioscience 45, 456–464. http://dx.doi.org/10.2307/1312789.

Fujiki, S., Aoyagi, R., Tanaka, A., Imai, N., Kusma, A., Kurniawan, Y., Lee, Y., Sugau, J., Pereira, J., Samejima, H., Kitayama, K., 2016. Large-scale mapping of tree-community composition as a surrogate of forest degradation in Bornean tropical rain forests. Land 5, 45. http://dx.doi.org/10.3390/land5040045.

Fuller, D.O., Jessup, T.C., Salim, A., 2004. Loss of forest cover in Kalimantan, Indonesia, since the 1997–1998 El Nino. Conserv. Biol. 18, 249–254. http://dx.doi.org/10.1111/j.1523-1739.2004.00018.x.

Gaveau, D.L.A., Sloan, S., Molidena, E., Yaen, H., Sheil, D., Abram, N.K., Ancrenaz, M., Nasi, R., Quinones, M., Wielaard, N., Meijaard, E., 2014. Four decades of forest persistence, clearance and logging on Borneo. PLoS One 9, 1–11. http://dx.doi.org/10.1371/journal.pone.0101654.

Granados, A., Crowther, K., Brodie, J.F., Bernard, H., 2016. Persistence of mammals in a selectively logged forest in Malaysian Borneo. Mamm. Biol. 81, 268–273. http://dx.doi.org/10.1016/j.mambio.2016.02.011.

Hart, J.A., 1999. Impact and sustainability of indigenous hunting in the Ituri forest, congo-zaire: a comparison of unhunted and hunted duiker populations. In: Robinson, J.G., Bennett, E.L. (Ed.), Hunting for Sustainability in Tropical Forests. Columbia University Press, New York, pp. 106–153.

Harvey, C.A., Medina, A., Sánchez, D.M., Vilchez, S., Hernández, B., Saenz, J.C., Maes, J.M., Casanoves, F., Sinclair, F.L., 2004. Patterns of animal diversity in different forms. Ecol. Appl. 16, 1986–1999. http://dx.doi.org/10.1890/1051-0761(2006)016[1986:POADID]2.0.CO;2.

Heydon, M.J., Bulloh, P., 1996. The impact of selective logging on sympatric civet species in Borneo. Oryx 30, 31. http://dx.doi.org/10.1017/S0030605300021360.

Imai, N., Seino, T., Aiba, S., Takyu, M., Titin, J., Kitayama, K., 2012. Effects of selective logging on tree species diversity and composition of Bornean tropical rain forests at different spatial scales. Plant Ecol. 213, 1413–1424. http://dx.doi.org/10.1007/s11258-012-0100-y.

Imai, N., Tanaka, A., Samejima, H., Sugau, J.B., Pereira, J.T., Titin, J., Kurniawan, Y., Kitayama, K., 2014. Tree community composition as an indicator in biodiversity monitoring of REDD+. For. Ecol. Manage. 313, 169–179. http://dx.doi.org/10.1016/j.foreco.2013.10.041.

Johns, A.D., 1988. Effects of "Selective" timber extraction on rain forest structure and composition and some consequences for Frugivores and Folivores. Biotropica 20, 31. http://dx.doi.org/10.2307/2388423.

Johns, A.D., 1986. Effects of selective logging on the behavioral ecology of West Malaysian primates. Ecology 67, 684–694. http://dx.doi.org/10.2307/1937692.

Johns, A.D., 1985. Selective logging and wildlife conservation in tropical rain-forest: problems and recommendations. Biol. Conserv. 31, 355–375. http://dx.doi.org/10.1016/0006-3207(85)90091-6.

Johns, A.G., 1997. Timber Production and Biodiversity Conservation in Tropical Rain Forests. Cambridge University Press, Cambridge.

Kleinschroth, F., Healey, J.R., 2017. Impacts of logging roads on tropical forests. Biotropica 49, 620–635. http://dx.doi.org/10.1111/btp.12462.

Mazerolle, M.J., Villard, M.-A., 1999. Patch characteristics and landscape context as predictors of species presence and abundance: a review. Écoscience 6, 117–124. http://dx.doi.org/10.1080/11956860.1999.11682524.

Meijaard, E., Sheil, D., 2008. The persistence and conservation of Borneo's mammals in lowland rain forests managed for timber: Observations, overviews and opportunities. Ecol. Res. 23, 21–34. http://dx.doi.org/10.1007/s11284-007-0342-7.

Meijaard, E., Sheil, D., 2007. A logged forest in Borneo is better than none at all. Nature 446http://dx.doi.org/10.1038/446974a. 974–974.

Morrison, M.L., Marcot, B.G., Mannan, R.W., 2006. Wildlife-Habitat Relationship, 3rd ed. Island Press, Washington.

Muchaal, P.K., Ngandjui, G., 1999. Impact of village hunting on wildlife populations in the Western Dja reserve. Cameroon. Conserv. Biol. 13, 385–396. http://dx.doi.org/10.1046/j.1523-1739.1999.013002385.x.

Negrete-Yankelevich, S., Fragoso, C., Newton, A.C., Heal, O.W., 2007. Successional changes in soil, litter and macroinvertebrate parameters following selective logging in a Mexican Cloud Forest. Appl. Soil Ecol. 35, 340–355. http://dx.doi.org/10.1016/j.apsoil.2006.07.006.

O'Connell, A., Nichols, J., 2011. Camera Traps in Animal Ecology, Ecology. Springer Japan, Tokyo. https://doi.org/10.1007/978-4-431-99495-4.

Oksanen, J., Blanchet, F.G., Friendly, M., Kindt, R., Legendre, P., McGlinn, D., R, P., Minchin, O'Hara, R.B., Simpson, G.L., Solymos, P., Stevens, M.H.H., Szoecs, E., Wagner, H., 2018. vegan: Community Ecology Package.

Peh, K.S.H., De Jong, J., Sodhi, N.S., Lim, S.L.H., Yap, C.A.M., 2005. Lowland rainforest avifauna and human disturbance: Persistence of primary forest birds in selectively logged forests and mixed-rural habitats of southern Peninsular Malaysia. Biol. Conserv. 123, 489–505. http://dx.doi.org/10.1016/j.biocon.2005.01.010.

Pfeifer, M., Lefebvre, V., Peres, C.A., Banks-Leite, C., Wearn, O.R., Marsh, C.J., Butchart, S.H.M., Arroyo-Rodríguez, V., Barlow, J., Cerezo, A., Cisneros, L., D'Cruze, N., Faria, D., Hadley, A., Harris, S.M., Klingbeil, B.T., Kormann, U., Lens, L., Medina-Rangel, G.F., Morante-Filho, J.C., Olivier, P., Peters, S.L., Pidgeon, A., Ribeiro, D.B., Scherber, C., Schneider-Maunoury, L., Struebig, M., Urbina-Cardona, N., Watling, J.I., Willig, M.R., Wood, E.M., Ewers, R.M., 2017. Creation of forest edges has a global impact on forest vertebrates. Nature 551, 187–191. http://dx.doi.org/10.1038/nature24457.

Poulsen, J.R., Clark, C.J., Bolker, B.M., 2011. Decoupling the effects of logging and hunting on an Afrotropical animal community. Ecol. Appl. 21, 1819–1836. http://dx.doi.org/10.1890/10-1083.1.

PT Ratah Timber, 2014. Rencana Kerja Tahunan Usaha Pemanfaatan Hasil Hutan Kayu Dalam Hutan Alam Pada Hutan Produksi Periode Tahun 2015.

PT Roda Mas Timber Kalimantan, 2008. Rencana Kerja Usaha Pemanfaatan Hasil Hutan Kayu Dalam Hutan Alam Pada Hutan Produksi Periode Tahun 2007 s/d 2016.

R Core Team, 2017. R: A Language and Environment for Statistical Computing.

Reyna-Hurtado, R., Tanner, G.W., 2007. Ungulate relative abundance in hunted and non-hunted sites in Calakmul Forest (Southern Mexico). Biodivers. Conserv. 16, 743–756. http://dx.doi.org/10.1007/s10531-005-6198-7.

Samejima, H., Lagan, P., Kitayama, K., 2013. Impacts of two different forest management practices on the abundance of mammals. In: Kitayama, K. (Ed.), Co-Benefits of Sustainable Forestry: Ecological Studies of a Certified Bornean Rain Forest. Springer Japan, Tokyo, pp. 89–111.

Samejima, H., Ong, R., Lagan, P., Kitayama, K., 2012. Camera-trapping rates of mammals and birds in a Bornean tropical rainforest under sustainable forest management. For. Ecol. Manage. 270, 248–256. http://dx.doi.org/10.1016/j.foreco.2012.01.013.

Samejima, H., Semiadi, G., 2012. First record of Hose's Civet Diplogale hosei from Indonesia, and records of other carnivores in the Schwaner Mountains, Central Kalimantan, Indonesia. Small Carniv. Conserv. 46, 1–7.

Scherer-Lorenzen, M., Bonilla, J.L., Potvin, C., 2007. Tree species richness affects litter production and decomposition rates in a tropical biodiversity experiment. Oikos 116, 2108–2124. http://dx.doi.org/10.1111/j.2007.0030-1299.16065.x.

Sundarapandian, S.M., Swamy, P.S., 1999. Litter production and leaf-litter decomposition of selected trees species in tropical forests at Kodayar in the Western Ghats. India. For. Ecol. Manage. 123, 321–344.

Syakirah, S., Zubaid, A., Prentice, C., Lopez, A., Azmin, M.R., Mohd Yusof, A., 2000. A small-mammal survey at Tasek Bera, Pahang, Malaysia's first Ramsar site. Malayan Nat. J. 54, 31–41.

Taper, M.L., Bohning-Gaese, K., Brown, J.H., 1995. Individualistic responses of bird species to environmental change. Oecologia 101, 478–486. http://dx.doi.org/10.1007/BF00329427.

Trolliet, F., Huynen, M.C., Vermeulen, C., Hambuckers, A., 2014. Use of camera traps for wildlife studies. A review. Biotechnol. Agron. Soc. Environ. 18 466–454.

Wilkie, D., Shaw, E., Rotberg, F., Morelli, G., Auzel, P., 2000. Roads, development, and conservation in the Congo Basin. Conserv. Biol. 14, 1614–1622. http://dx.doi.org/10.1046/j.1523-1739.2000.99102.x.

Wilson, W.L., Johns, A.D., 1982. Diversity and abundance of selected animal species in undisturbed forest, selectively logged forest and plantations in East Kalimantan. Indonesia. Biol. Conserv. 24, 205–218. http://dx.doi.org/10.1016/0006-3207(82)90058-1.