Jurnal Manajemen Hutan Tropika, *28*(1), 40–48, April 2022
EISSN: 2089-2063
DOI: 10.7226/jtfm.28.1.40

Scientific Article
ISSN: 2087-0469

The Use of Forest Refugia by Ungulate After the 2015'Fire in Tesso Nilo National Park, Riau-Indonesia

Wiwid Prayoga, Muhammad Ali Imron[*]

Department of Forest Resources Conservation, Faculty of Forestry, Universitas Gadjah Mada, Komplek Agro No. 1
Bulaksumur, Yogyakarta, Indonesia 55280

Received August 24, 2021/Accepted March 16, 2022

Abstract

Wildfires in Indonesia 2015 burnt forests in many protected areas, including remaining forests in Tesso Nilo National Park. We investigated the extent to which 2015's wildfires affect the remaining habitat and the spatial distribution of ungulates in the park by using satellite images to identify burnt and unburnt forest patches. Habitat conditions and the presence of ungulates indicated by the number of ungulate signs were compared between burnt and unburnt areas. The fire devastated trees at different rates depending on tree life form stages. We found that younger trees were more vulnerable to fire. The abundance of ungulates did not differ significantly between burnt and unburnt areas, but all vegetation characteristics were significantly different (Mann-Whitney U-test p-value < 0.05*). We surmise that food such as new shoots or leaves, particularly at the edge of burnt areas, attracted ungulates out of unburnt areas. The remaining forest is relatively small, and the park is under continued pressure from illegal conversion, so any further loss of remaining forest as refugia will likely harm the ungulate population. We recommend that management should prioritize the preservation of remaining pristine habitat and the reduction of fire suppression, especially during the dry season.*

Keywords: *disturbed habitats, fire events, refugee areas, restoration, ungulate conservation*

**Correspondence author, email*: *maimron@ugm.ac.id, tel. +62-274-550541*

Introduction

Wildfires in Indonesia occur annually and are mainly induced by anthropogenic factors, affect biodiversity, health, the domestic and international economy, and create environmental hazards (Yulianti et al., 2012; Hayasaka et al., 2014; 2016; Sze et al., 2019). Wildfires also disrupt ecosystems, communities, population structures, resources, and physical conditions of habitats (Forsyth et al., 2012), and therefore they profoundly affect wildlife. Although ecologists historically assumed that fires were universally harmful, they can play an important role in maintaining heterogeneity of wildlife communities (Bowleset al., 2011; He et al., 2019).

Wildlife, including large mammals such as ungulates, reacts towards fire events by a variety of strategies, including fleeing away during the fire and then recolonizing burned areas after the fire; sheltering below ground in diapause or diapause-like states or life stages; or sheltering in refugia, however, it is dependent on land use conditions of the landscape (van Mantgem et al., 2015). Consequently, remaining forests and unburnt areas are important to be functioning as refugia. As wildfire plays roles in shaping the structure and composition of vegetation which then affect the response of ungulates to post-burnt patches (Allred et al., 2011), it can have multiple interconnected positive and/or negative effects on ungulate populations. For example, fires create canopy gaps in the forest, which allows understory food plants to thrive (Sollmann et al., 2015). However, when gaps are created, the cover is reduced (Li et al., 2014), which exposes ungulates to predation.

A recent event of wildfires occurred in Sumatera 2015 and impacted several protected areas, including Tesso Nilo National Park (TNNP) and its remaining forests. In the park, wildlife habitat has been suppressed by massive expansion of agricultural plantations, mainly palm oil (Poor et al., 2019), accordingly, the remaining forests act as refugia for ungulates such as wild boar (*Sus scrofa*), sambar deer (*Rusa unicolor*), mouse deer (*Tragulus kanchil* and *T. napu*), muntjac deer (*Muntiacus muntjak*), and malayan tapir (*Tapirus indicus*). In addition, those ungulate species also become potential prey of the critically endangered sumatran tiger (*Panthera tigris sumatrae*) (IUCN, 2019).

Disturbance of refugia by fire can influence the distribution of ungulates, and consequently their main predator (Sunarto et al. 2012). Unlike ungulates of the African Savanna and higher altitude in Nepal which are well adapted with recurrent fires (Allred et al., 2011; Lamichhane et al., 2020), sumatran ungulates are less experienced because wildfires are a recent, accidental and anthropogenic phenomenon (Adrianto et al., 2019; Sze et al., 2019). Furthermore, the effect of fires on the distribution of ungulates on Sumatera is less studied and leads to a gap of crucial knowledge on wildlife conservation. Therefore, this study aimed to investigate the extent to which 2015's wildfires affect the remaining habitat of ungulates in the TNNP and describe spatial distribution patterns of ungulates as a proxy for ungulate habitat used post fire disturbance.

Methods

Study area The TNNP is a relatively new national park in Riau Province, established in 2004 with a total size of 38,576 ha and has been expanded in 2008 to its current size of 83,068 ha (Figure 1). The park is threatened by massive deforestation and habitat destruction. Between 1982 and 2005, smallholders and corporations reduced Riau province's forests by 65% annually (Uryu et al., 2007). Illegal oil-palm plantations and other land cover types replaced the natural forest, of which only small and isolated patches remain (Poor et al., 2019). Generally, the forest is cleared by slash-and-burn, causing unregulated and unsuppressed annual wildfires that spread far beyond their intended targets (Adrianto et al., 2019; Sze et al., 2019). Therefore, the remaining forest patches potentially serve as refugia for fauna to escape from fires and other anthropogenic disturbances such as hunting (Berry et al., 2015; van Mantgem et al., 2015)

Land cover classification and burned area detection We delineated 7,525.26 ha of the forested area within the TNNP using Landsat 8 images with manually digitizing at 7-5-3 bands (assessed at December 17, 2015 and February 19, 2016) (https://landsat.usgs.gov/). The same images were used to delineate burnt and unburnt areas manually. The study area covered a relatively natural or secondary forest area, which was disturbed by fire in 2015. We used Landsat 8 images (WRS2 path: 126 rows: 060) assessed at June 2, 2014 to classify land cover in our study area (https://landsat.usgs.gov/). Figure 1 shows the study area, which consists of mainly natural vegetation and a few disturbed areas. Because fire is an incidental event in this study area, we applied impact-reference design (Morrison et al., 2008) by comparing the burnt and unburnt area for further data collection. In each burnt area, we also selected nearby un-burnt area for data collection as described in following section.

Data collection We collected data on habitat condition in burnt and unburnt patches, as well as the presence and absence of ungulates from December 2015 to February 2016. We utilized 26.5 km of 10 m wide transects, distributing the transects between burnt and unburnt patches with the length of each transect varying between 250–1,000 m (Figure 2). Each transect consisted of 50 m sub-segments with a total of 220 sub-segments (which equate to 11 km in total length) in the burnt area, and 310 sub-segments (15.5 km) in the unburnt area. In each sub-segment, we searched for any signs of ungulates and marked the GPS position at each track or feces. We used track identification guideline following

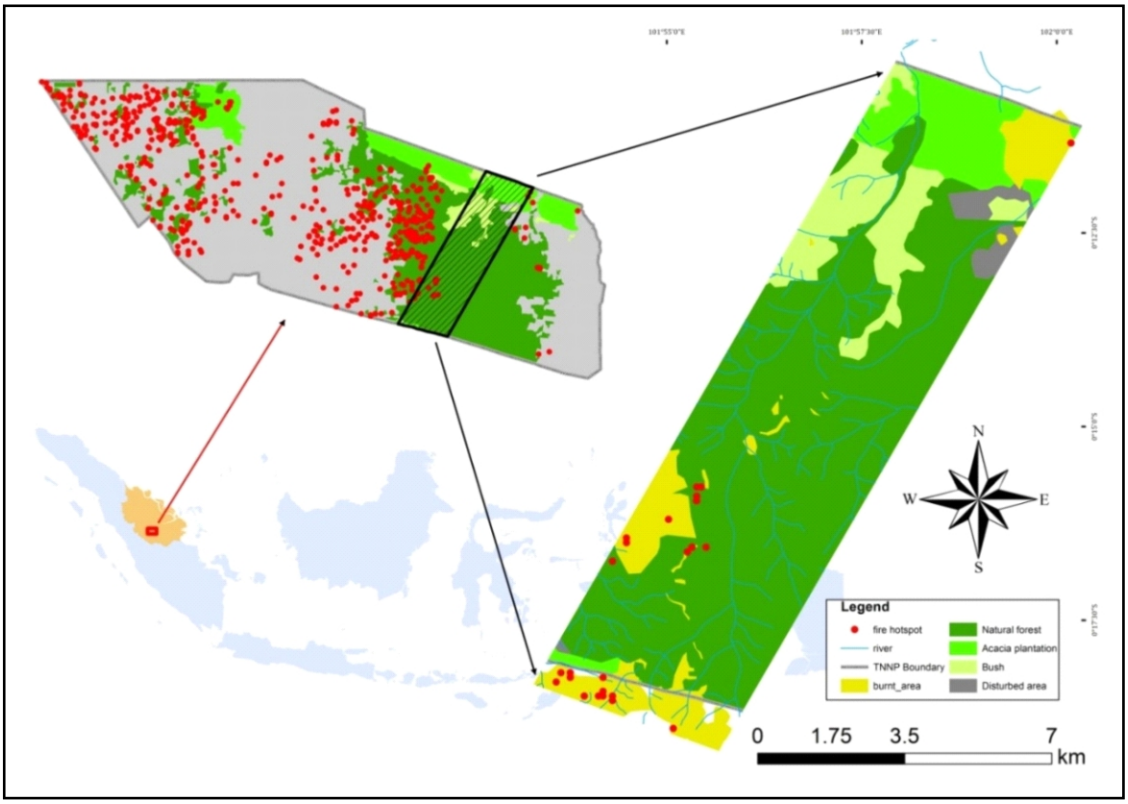

Figure 1 Map of the study area, Tesso Nilo National Park in Riau Province, Sumatera, Indonesia. The selected rectangular area represents a relatively less disturbed forest area in the park. Red dots indicate the location of 1-km active fires detected by MODIS during 2015.

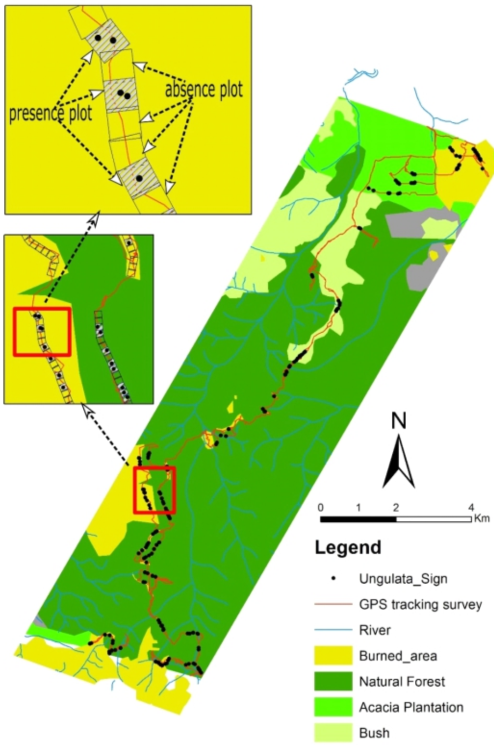

Figure 2 The distribution map of ungulates in the burnt area (yellow), natural forest (dark green), acacia plantation (light green) and bushes (light yellow) and the transect location as well as segments (middle) and plot position in the transect (above) in the study area of the Tesso Nilo National Park.

Francis (2008), and if we did not sure with identification, we collected photos of the tracks/signs and consulted to experts.

We calculated the distance between ungulate detection and the border of the burnt area by applying the proximity-near tool for calculating the distance in ArcGIS 10 (Mitchell, 2005). Then, we classified distances into intervals of 250 m. The burnt patches were small, so we defined a distance of > 500 m as was the largest category of distance from the edge of a burnt patch. However, in unburnt patches, the our larges farthest category of distance was > 2,000 m. We projected the distribution of ungulates using the distance from the edge of burnt area to find possible pattern of distribution from burnt-patches.

Within each segment, we measured vegetation characteristics every 250 m, measured horizontal and vertical cover within a circular plot with a radius of 11.3 m (Beest et al., 2012). We measured the horizontal cover of vegetations at 0–30 cm, 30–100 cm, 100–200 cm, and 200–300 cm as proxy for ungulate cover. We calculated the ratio between the covered area and the total sampling area to determine the percentage of both horizontal and vertical vegetation cover. We also measured the vegetation density using nested plots for grass (1×1 m); seedlings, herbs, and shrubs (2×2 m); saplings (5×5 m); poles (10×10 m); and trees (20×20 m).

We determined the number of dead stems at sapling, pole and tree levels. We did not include seedling level as seedling growth can be promoted by fires. We defined dead trees using the physical appearance especially the cambium condition, if the tree still had fresh cambium or bark, we classified it as a live tree (Marschall et al., 2014). We calculated the ratio between dead vegetation and total vegetation in the plot. In each 1×1 m plot we also measured the organic material composition to identify potential fuel for wildfire on the forest floor by estimated the percent of each component of litter, deadwood, or no materials. We performed the test of group difference using Mann-Whitney U and Kruskal-Wallis tests to investigate the difference in vegetation characteristics in burnt and unburnt patches as it is commonly used for site comparison (Keuling et al., 2008).

Results

Effect of fire on habitat condition Identification from Landsat image analysis showed that the size of the burnt area was varied. Burnt areas covered almost 10% of the study area (604.18 ha). Burnt areas occurred mainly close to the border of the park (Figure 1.). Although the total area of burnt land in natural forest is the largest compared to the other land cover types (355.62 ha), only 6.37% of the natural forest was burnt, compared to 24.95% of acacia plantations (Table 1).

In total, we established 62 plots in unburnt patches and 44 plots in burnt patches. The fire killed trees at different rates depending on the vegetation stage. We found high mortality at sapling, pole, and tree stage, ranging from 23% at the tree stage to 100% at the sapling stage (Figure 3). When comparing vegetation characteristic of the ungulate between burnt and unburnt patches, we found that all variables were significantly different between burnt and unburnt areas (Table 2). The availability of grass, herbs, and seedlings was more abundant in the burnt patches than in unburnt patches (Figure 4).

The response of vegetation density to distance from patch edge varied among vegetation types. Grass and herbs were more dense at the center (> 500 m from the edge of the burnt patch) than at the edge (0–250 m) and have higher densities in burnt patches overall (Table 2, Figure 4). Seedlings were also denser in burnt patches but decreased in density with increasing distance from the edge of burnt patches. Saplings were more dense in unburnt patches and almost absent in burnt patches (Table 2, Figure 4).

Ungulate distribution We identified five ungulates (wild boar *S. scrofa*, sambar deer *R. unicolor*, mouse deers *T. kanchil* and *T. napu*, muntjac deer *M. muntjak*, and malayan tapir *T. indicus* in the TNNP through the identification of feces, tracks and direct sighting in the field. The wild boar was most frequently detected in burnt and unburnt areas, followed by muntjac, tapir, mouse deer and finally, sambar deer (Table 3). We detected ungulates slightly less frequently in unburnt patches. In addition, detections were clumped close to the edge of all patches (< 250 m from the edge). Detections of ungulates in burnt patches declined sharply from the near-edge (< 250 m) to the next furthest (250–500 m) distance classification, whereas a smooth decline in ungulate detections with distance from the edge was found in unburnt patches.

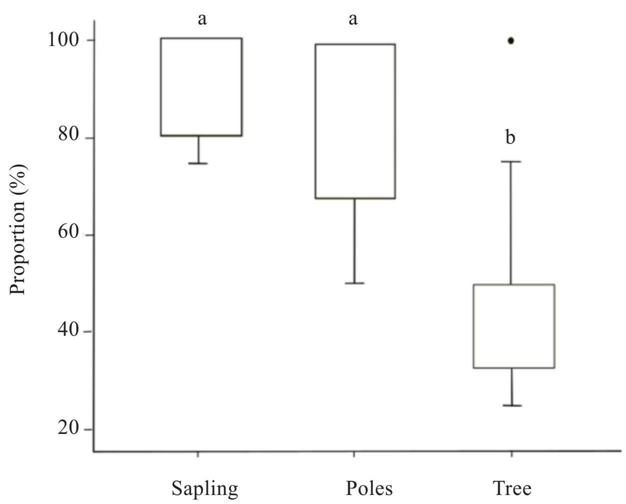

Figure 3 Boxplot of the percentage of death vegetation at sapling, poles, and trees level in burnt patches.

Discussion

Our study provides insight into the ability of Sumatera's tropical forest ungulates to adapt to habitat disturbance caused by fire and elucidates the underlying vegetation shifts. We discuss here the conservation consequences of habitat disturbance for ungulates living in the TNNP and implications for conserving ecosystem as a whole and the Sumatran tiger as flagship species in the area.

Vegetation changes This study reveals that habitat characteristics of ungulates were significantly different at the ground cover (seedling, grass, and herbs), saplings, and poles in burnt and unburnt areas.. As fire is able to remove most ground cover (Leahy et al., 2015; Sollmann et al., 2015), we observed fresh vegetation in burnt patches, which we surmise is post-fire regrowth. The shoots appeared in the burnt patches that might be supported by the high rainfall in the region Page et al., 2013; Field et al., 2015). Contrarily, sapling and pole levels had lower density and higher mortality.

Our results showed that canopy and ground cover has reverse conditions as a response to fire. Among ground cover vegetation, grass has received more benefit after a forest fire

Table 1 Proportion of burnt and unburnt area in each land cover in the study site of the Tesso Nilo National Park

Land cover	Burnt		Unburnt		Total (ha)	Proportion of burnt area (%)
	Area (ha)	Proportion (%)	Area (ha)	Proportion (%)		
Natural forest	355.62	58.86	5,221.84	75.45	5,577.46	6.37
Plantation	237.39	39.29	713.80	10.31	951.19	24.96
Bush	1.48	0.24	813.11	11.75	814.59	0.18
Disturbed area	9.69	1.61	172.32	2.49	182.01	5.32
Total	604.18	100	6,921.08	100	7,525.25	31.56

Table 2 Mann-Whitney U test of habitat variables between burnt and unburnt patches in the study area of the Tesso Nilo National Park

Habitat variable	Test statistic	
	Mann-Whitney U-test	Asymp. Sig.
Vertical cover density (%)		
Canopy closure	2.000; *p*-value < 0.001; N = 44	2.00E-18
Ground cover	539.000; *p*-value < 0.001; N = 44	1.00E-07
Horizontal cover density(%)		
0–30 cm	0.000; *p*-value < 0.001; N = 44	2.00E-18
30–100 cm	27.500; *p*-value < 0.001; N = 44	1.00E-17
100–200 cm	184.000; *p*-value < 0.001; N = 44	4.00E-14
200–300 cm	2.000; *p*-value < 0.001; N = 44	2.00E-18
Surface component (%)		
Leaf litter	9.500; *p*-value < 0.001; N = 44	4.00E-18
Decayed wood	240.500; *p*-value < 0.001; N = 44	6.00E-13
Soil	217.000; *p*-value < 0.001; N = 44	2.00E-13
Vegetation density (ind ha^{-1})		
Grass	673.500; *p*-value < 0.001; N = 44	3.00E-06
Seedling	741.000; *p*-value < 0.001; N = 44	4.00E-05
Herb	627.500; *p*-value < 0.001; N = 44	8.00E-07
Sapling	158.000; *p*-value < 0.001; N = 44	2.00E-15
Pole	469.000; *p*-value < 0.001; N = 44	2.00E-09
Tree	299.500; *p*-value < 0.001; N = 44	3.00E-12

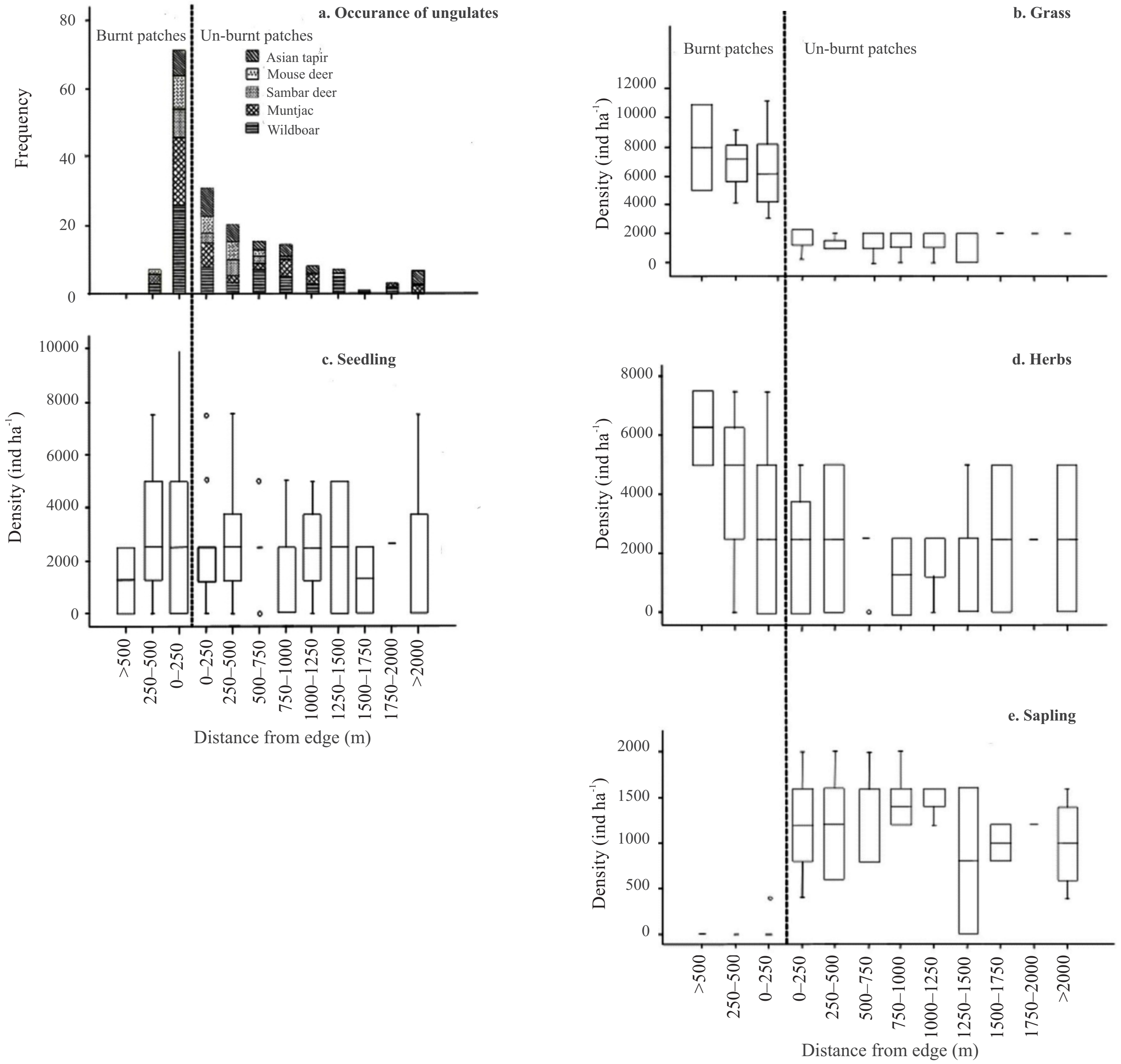

Figure 4 The proportion of the presence of each ungulate species (a) and habitat conditions: grasses (b), herb (c), seedling (d), and sapling (e) from the border of burnt and unburnt area in the study site of the Tesso Nilo National Park.

(Leahy et al., 2015; Sollmann et al., 2015) particularly imperata grass (*Imperata cylindrica*), an invasive species that has been commonly found in burnt patches (Yassir et al., 2010). Whereas at seedling level, *Acacia mangium* seeds survived from fire and fast-spreading covering the forest floor and invasive as suggested by Krisnawati et al. (2011). The high temperature of wildfire has caused some part of vegetation has died and caused gaps which then promoted undergrowth to sprout (Leahy et al., 2015; Sollmann et al., 2015). The density of seedlings and grass in burnt patches were higher than in unburnt patches which improved the quantity of food. Whereas reverse pattern in sapling showed that fire-based disturbance of ungulate's habitat reducing covers. Canopy gaps and cover loss provided an opportunity for ground cover and seedling to grow (Leahy et al., 2015; Sollmann et al., 2015) and accelerated by the shifting from dry to wet season in particular during the wet season (Crotteau et al., 2013). This natural event would likely happen in the TNNP as burnt patches might facilitate the new growth of seedlings, herbs, and grasses as shown inFigure 5.

Fires largely attacked the surface of forest areas in the TNNP impacting lower vegetation such as sapling and poles than the taller trees. The diameter size of trees might change their resilience to fire and larger diameter size might be less susceptible to fire (Marschall et al., 2014). However, since we did not differentiate the severity of fire in each plot, we could not relate to vegetation mortality. Other factors such as tree species, the thickness of bark, water content as well as species-specific traits might also receive different impact from the fire (Leahy et al., 2015; Sollmann et al., 2015).

Figure 5 Vegetation conditions in a burnt patch, which show heavy loss of vertical cover in burnt patches promoting growth of grass, herbs, and seedlings.

These can be further investigated for better insight into knowledge on the effect of fire for resources of ungulates.

The response of ungulates on disrupted habitat after fire

Our study highlights that ungulates in the TNNP occupied both burnt and unburnt patches and show a tendency to stay close to the border of both where resources are abundant and they are able to find cover in intact forest. In an intact forest with closed canopies, lack of sunlight limits the availability of food at ground level. Therefore, ungulates seek open spaces such as forest gaps from falling trees or forest edges (Peterson & Dodson, 2016) . Burnt patches mimic forest gaps where open canopies facilitate the growth of forest floor vegetation. After being driven out by fire, the ungulates in the TNNP returned to burnt patches (Allred et al., 2011; van Mantgem et al., 2015), attracted by nutritious grasses, seedlings, and herbs (Allred et al., 2011). These findings also agree with other studies that newly sprouted vegetation attracts ungulates (Allred et al., 2011; Peterson & Dodson, 2016; Smith et al., 2016; Zecherle et al., 2020) and results in a fire adaptation mechanism (Pausas & Parr, 2018). Other studies highlighted that the high-quality resources after fire improves population growth of sambar, muntjac, and mouse deer, including wild boar (Massei & Genov, 2004). Ungulates in other regions of Indonesia such as in eastern Indonesia have shown a positive response after fires when shoots are emerged and providing nutritious food for timor deer (Kayat et al., 2017).

Our findings show that ungulates has different spatial pattern after fires (Figure 4a), but they distributed in a different pattern. This study provides a basic description and up to date information of the response of ungulates to fire disturbance in Sumatera, but this topic needs to be reinforced by further studies, especially on the relationship between wildfires and the movement of ungulates by implementing more reliable techniques such as using GPS collar or radio-tracking during fire occurrence (Nurvianto et al., 2015; Mossbrucker et al., 2016). Such further studies are expected to draw clearer pictures of the interaction of ungulates, habitat use and wildfires. For example, using near real-time data of ungulate movement using GPS collar will give insight on the behavior of both prey and predator to avoid various disturbances such as fire, roads, humans, etc (Poor et al., 2019). In addition, modeling studies can be used in predicting movement of ungulates, for instance, using agent-based models for understanding mechanisms of adaptation during and after fire (Widyastuti et al., 2021).

Conclusion

The TNNP was hit by fire in 2015 and affected habitat for ungulate in particular vegetation community. Fire has caused mortality and destruction of trees at different rates depending on tree life form stages. While the abundance of ungulates did not differ significantly between burnt and unburnt areas, all vegetation characteristics as habitat for ungulate were significantly different. Food availability such as new shoots or leaves, particularly at the edge of burnt areas, attracted ungulates out of unburnt areas. Because the remaining forest is relatively small, and the park is under continued pressure from illegal conversion, any further loss will likely harm the ungulate population. We recommend that management should prioritize the preservation of remaining pristine habitat and the reduction of fire suppression, especially during the dry season.

Acknowledgment

This study was part of thesis from Wiwid Prayoga in the Faculty of Forestry Universitas Gadjah Mada. We thanks to Balai Taman Nasional Tesso Nilo for providing the permit, Erin Poor for supporting this research, Febri Angriawan, WWF Riau team and Yogi for supporting the field work, and anonymous reviewers for providing feedback. We thank to LPDP-DIPI for providing funds through TOFEWSI: Toward Fire Early Warning System for Indonesia with contract number NE/P014801/1. We tank to Katherine Lauck for providing proofread for this manuscript.

References

Allred, B. W., Fuhlendorf, S. D., & Engle, D. M. (2011). Ungulate preference for burned patches reveals strength of fire – grazing interaction. *Ecology and Evolution*, 1–13. https://doi.org/10.1002/ece3.12

Ariyanto, T., Dinata, Y., Dwiyanto, Turyanto, E., Sugito, W., Kirklin, S., & Amin, R. (2021). Status of sumatran tiger in the Berbak-Sembilang landscape (2020). *Journal of Threatened Taxa*, *13*(6), 18419–18426. https://doi.org/10.11609/jott.6271.13.6.18419-18426

Beest, F. M. V., Moorter, B. V., & Milner, J. M. (2012). Temperature-mediated habitat use and selection by a heat-sensitive northern ungulate temperature-mediated habitat use and selection by a heat-sensitive northern ungulate. *Animal Behaviour*, *84*(3), 723–735. https://doi.org/10.1016/j.anbehav.2012.06.032

Berry, L. E., Driscoll, D. A., Stein, J. A., Blanchard, W., Banks, S. C., Bradstock, R. A., & Lindenmayer, D. B. (2015). Identifying the location of fire refuges in wet forest ecosystems. *Ecological Applications*, *25*(8), 150409201937008. https://doi.org/10.1890/14-1699.1

Bowles, M., Apfelbaum, S., Haney, A., Lehnhardt, S., & Post, T. (2011). Canopy cover and groundlayer vegetation dynamics in a fire managed eastern sand savanna. *Forest Ecology and Management*, *262*(11), 1972–1982. https://doi.org/10.1016/j.foreco.2011.08.036

Crotteau, J. S., Varner, J. M., & Ritchie, M. W. (2013). Post-fire regeneration across a fire severity gradient in the southern Cascades. *Forest Ecology and Management*, *287*, 103–112. https://doi.org/10.1016/j.foreco.2012.09.022

Field, R. D., Spessa, A. C., Aziz, N. A., Camia, A., Cantin, A., Carr, R., …, & Wang, X. (2015). Development of a Global Fire Weather Database. *Natural Hazards and Earth System Sciences*, *15*(6), 1407–1423. https://doi.org/10.5194/nhess-15-1407-2015

Forsyth, D. M., Gormley, A. M., Woodford, L., & Fitzgerald, T. (2012). Effects of large-scale high-severity fire on occupancy and abundances of an invasive large mammal in south-eastern Australia. *Wildlife Research*, *39*(7), 555–564. https://doi.org/10.1071/WR12033

Hayasaka, H., Noguchi, I., Putra, E. I., Yulianti, N., & Vadrevu, K. (2014). Peat-fire-related air pollution in Central Kalimantan, Indonesia. *Environmental Pollution*, *195*(0), 257–266. https://doi.org/10.1016/j.envpol.2014.06.031

Hayasaka, H., Takahashi, H., Limin, S. H., Yulianti, N., & Usup, A. (2016). Peat fire occurence. In *Tropical peatland ecosystems* (pp. 377–395). https://doi.org/10.1007/978-4-431-55681-7

He, T., Lamont, B. B., & Pausas, J. G. (2019). Fire as a key driver of earth's biodiversity. *Biological Review*, 19832010. https://doi.org/10.1111/brv.12544

Imron, M. A., Herzog, S., & Berger, U. (2011). The influence of agroforestry and other land-use types on the persistence of a sumatran tiger (*Panthera tigris sumatrae*) population: An individual-based model approach. *Environmental Management*, *48*(2), 276–288. https://doi.org/10.1007/s00267-010-9577-0

Kayat, Pudyatmoko, S., Maksum, M., & Imron, M. A. (2017). Potensi konflik penggembalaan kuda pada habitat rusa jawa (*Rusa timorensis* Blainville 1822) di kawasan Tanjung Torong Padang, Nusa Tenggara Timur. *Jurnal Ilmu Kehutanan*, *1*(2), 22–29. https://doi.org/10.1111/gcb.13051

Keuling, O., Stier, N., & Roth, M. (2008). How does hunting influence activity and spatial usage in wild boar *Sus scrofa* L.? *European Journal of Wildlife Research*, *54*(4), 729–737. https://doi.org/10.1007/s10344-008-0204-9

Krisnawati, H., Kaliio, M., & Kanninen, M. (2011). *Acacia mangium Willd.*: *Ecology, silviculture and productivity*. Bogor: CIFOR.

Leahy, L., Legge, S. M., Tuft, K., McGregor, H. W., Barmuta, L. A., Jones, M. E., & Johnson, C. N. (2015). Amplified predation after fire suppresses rodent populations in Australia's tropical savannas. *Wildlife Research*, *42*(8), 705–716. https://doi.org/10.1071/WR15011

Li, X., Buzzard, P., & Jiang, X. (2014). Habitat associations of four ungulates in mountain forests of southwest China, based on camera trapping and dung counts data. *Population Ecology*, *56*(1), 251–256. https://doi.org/10.1007/s10144-013-0405-2

Linkie, M., & Ridout, M. S. (2011). Assessing tiger-prey interactions in sumatran rainforests. *Journal of Zoology*, *284*(3), 224–229. https://doi.org/10.1111/j.1469-7998.2011.00801.x

Luskin, M. S., Albert, W. R., & Tobler, M. W. (2017). Sumatran tiger survival threatened by deforestation despite increasing densities in parks. *Nature Communications*, *8*(1). https://doi.org/10.1038/s41467-017-01656-4

Marschall, J. M., Guyette, R. P., Stambaugh, M. C., & Stevenson, A. P. (2014). Fire damage effects on red oak timber product value. *Forest Ecology and Management*, *320*, 182–189. https://doi.org/10.1016/j.foreco.2014.03.006

Massei, G., & Genov, P. V. (2004). The environmental impact of wild boar. *Galemys*, *135*(especial), 135–145. Retrieved from https://bepls.com/nov_2015/7BEPLS (1).pdf

Meijaard, E., Sheil, D., & Nasi, R. (2006). *Hutan pasca pemanenan*. Bogor: CIFOR.

Mitchell, A. (2005). *The ESRI guide to GIS analysis* (Volume 2). ESRI Press.

Morrison, M. L., Block, W. M., Strickland, M. D., Collier, B. A., & Peterson, M. J. (2008). *Wildlife study design*. New York: Springer. https://doi.org/10.1007/978-0-387-75528-1

Mossbrucker, A. M., Fleming, C. H., Imron, M. A., Pudyatmoko, S., & Sumardi. (2016). AKDE$_c$ home range size and habitat selection of sumatran elephants. *Wildlife Research*, *43*(7). https://doi.org/10.1071/WR16069

Nurvianto, S., Imron, M. A., & Herzog, S. (2015). Activity patterns and behaviour of denning dholes (*Cuonalpinus*) in a dry deciduous forest of East Java, Indonesia. *Bulletin of Environment, Pharmacology, and Life Science*, *4*(November), 45–54. Retrieved from https://bepls.com/nov_2015/7BEPLS (1).pdf

Page, S., Rieley, J., Hoscilo, A., Spessa, A., & Weber, U. (2013). Current fire regimes, impacts and the likely changes–IV: Tropical Southeast Asia. In J. G. Goldberg (Ed.), *Vegetation fires and global change–challenges for concerted international action a white paper directed to the united nations and international organizations* (pp. 89–99). Kessel. http://oro.open.ac.uk/39954/1/Global-Vegetation-Fire-White-Paper-2013-Chapter-07-Page-89-99.pdf

Pausas, J. G., & Parr, C. L. (2018). Towards an understanding of the evolutionary role of fire in animals. *Evolutionary Ecology*, *32*(2), 1–13. https://doi.org/ 10.1007/s10682-018-9927-6

Peterson, D. W., & Dodson, E. K. (2016). Post-fire logging produces minimal persistent impacts on understory vegetation in northeastern Oregon, USA. *Forest Ecology and Management*, *370*, 56–64. https://doi.org/10.1016/j.foreco.2016.04.004

Poor, E. E., Frimpong, E., Imron, M. A., & Kelly, M. J. (2019). Protected area effectiveness in a sea of palm oil: A Sumatran case study. *Biological Conservation*, *234*, 123–130. https://doi.org/10.1016/j.biocon.2019.03.018

Poor, E., Imron, M. A., & Kelly, M. J. (2019). The road to deforestation: Edge effects in an endemic ecosystem in Sumatra, Indonesia. *PLoS ONE*, *14*(7), e0217540. https://doi.org/10.1371/journal.pone.0217540

Risdianto, D., Martyr, D. J., Nugraha, R. T., Harihar, A., Wibisono, H. T., Haidir, I. A., …, & Linkie, M. (2016). Examining the shifting patterns of poaching from a long-term law enforcement intervention in Sumatra. *Biological Conservation*, *204*, 306–312. https://doi.org/10.1016/j.biocon.2016.10.029

Slik, J. W. F., Bernard, C. S., Beek, M. Van, Van Beek, M., Breman, F. C., & Eichhorn, K. A. O. (2008). Tree diversity, composition, forest structure and aboveground biomass dynamics after single and repeated fire in a bornean rain forest. *Oecologia*, *158*(3), 579–588. https://doi.org/10.1007/s00442-008-1163-2

Smith, D. S., Fettig, S. M., & Bowker, M. A. (2016). Elevated rocky mountain elk numbers prevent positive effects of fire on quaking aspen (*Populus tremuloides*) recruitment. *Forest Ecology and Management*, *362*, 46–54. https://doi.org/10.1016/j.foreco.2015.11.020

Sollmann, R., White, A. M., Gardner, B., & Manley, P. N. (2015). Investigating the effects of forest structure on the small mammal community in frequent-fire coniferous forests using capture-recapture models for stratified populations. *Mammalian Biology*, *80*(4), 247–254. https://doi.org/10.1016/j.mambio.2015.03.002

Steinmetz, R., Chutipong, W., Seuaturien, N., Chirngsaard, E., & Khaengkhetkarn, M. (2010). Population recovery patterns of Southeast Asian ungulates after poaching. *Biological Conservation*, *143*(1), 42–51. https://doi.org/10.1016/j.biocon.2009.08.023

Sze, J. S., Jefferson, & Lee, J. S. H. (2019). Evaluating the social and environmental factors behind the 2015 extreme fire event in Sumatra, Indonesia. *Environmental Research Letters*, *14*(1), 015001. https://doi.org/10.1088/1748-9326/aaee1d

van Mantgem, E. F., Keeley, J. E., & Witter, M. (2015). Faunal responses to fire in chaparral and sage scrub in California, USA. *Fire Ecology*, *11*(3), 128–148. https://doi.org/10.4996/fireecology.1103128

Widodo, F. A., Hanny, S., Utomo, E. H. S., Zulfahmi, Kusdianto, Septayuda, E., …, & Sunarto. (2017). Tigers and their prey in Bukit Rimbang Baling: Abundance baseline for effective wildlife reserve management. *Jurnal Ilmu Kehutanan*, *11*(118–129). Retrieved from https://jurnal.ugm.ac.id/jikfkt/article/view/28275/17239

Widyastuti, K., Imron, M. A., Pradopo, S. T., Suryatmojo, H., Sopha, B. M., Spessa, A., & Berger, U. (2021). PeatFire: An agent-based model to simulate fire ignition and spreading in a tropical peatland ecosystem. *International Journal of Wildland Fire*, *30*(2), 71–89. https://doi.org/10.1071/WF19213

Yassir, I., Kamp, J. van Der, & Buurman, P. (2010). Secondary succession after fire in Imperata grasslands of East Kalimantan, Indonesia. *Agriculture, Ecosystems & Environment*, *137*(1–2), 172–182. https://doi.org/

10.1016/j.agee.2010.02.001

Yulianti, N., Hayasaka, H., & Usup, A. (2012). Recent forest and peat fire trends in Indonesia. The latest decade by MODIS hotspot data. *Global Environmental Research*, *16*(1), 105–116.

Zecherle, L. J., Bar-david, S., Nichols, H. J., Templeton, A. R., Hipperson, H., Horsburgh, J., & Brown, R. P. (2020). Landscape resistance affects individual habitat selection but not genetic relatedness in a reintroduced desert ungulate. *Biological Conservation*, *252*, 108845. https://doi.org/10.1016/j.biocon.2020.108845