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EVALUATING EARLY POST-FIRE TROPICAL LOWER MONTANE FOREST RECOVERY IN INDONESIA

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Little is known about post-fire tropical montane forest succession in Indonesia. It is an important step to understand whether forests are recovering and if there is a need for reforestation or other rehabilitation efforts. This study assessed the structure and composition of post-fire forest regrowth in Raden Soerjo Grand Forest Park, a conservation forest management unit in East Java, Indonesia. Three high-severity burn units were sampled to assess forest response across a range of post-fire recovery times and one or two fire entries. Mean stems and basal area per unit by size class (tree, pole, sapling and seedling) and tree size class distribution by species, and mean percent cover of understory by life form were calculated. It was found that units experiencing short-term absence of fire and two fire entries were dominated by non-tree understory vegetation, including high grass cover, indicating that understory cover may hamper the establishment of tree regeneration. It was also found that higher species diversity and richness followed a single fire entry. The results provided a better understanding of early post-fire forest recovery. Managers should actively monitor burned areas and plan for active restoration, especially if non-forest vegetation dominates the area for a long period (> 8 years), or after a second fire entry.

Keywords: Regeneration, high-severity fire, fire frequency, understory diversity, non-native species

INTRODUCTION

Indonesia has experienced many large-scale high-severity forest fires in recent years, i.e. 1997, 1998, 2009, 2012 and 2015. In addition to increased loss in forest cover during these fire years, Indonesia has also become one of the highest emitters of greenhouse gasses (GHG) in the world (Fuller et al. 2004, Margono et al. 2014, Hooijer et al. 2010, Pearson et al. 2017, Tepley et al. 2017). High-severity fires are associated with El-Nino Southern Oscillation (ENSO) events which initiate long dry seasons and droughts between May and November (Tacconi et al. 2007, Wooster et al. 2012).

High-severity fire enlarges forest gaps, increasing solar radiation and causing subsequent negative ecological impacts in tropical forests (Tacconi et al. 2007, Jones et al. 2016). Recurring forest fire delays recovery and can alter short- and long-term forest structure and composition (Smiet 1992, Cochrane & Schulze 1999, Mostacedo et al. 2001, ITTO 2002, Sutomo 2009). In Indonesia, little is known about forest recovery following high-severity fires. Rahmasari (2011) discovered that high severity fire in Raden Soerjo Grand Forest Park (GFP), East Java, reduced species richness, stand density and basal area across all size classes. However, this study only examined a single fire entry (2009), one-year post-fire. Greater understanding of post-fire trajectories is necessary to improve reforestation and rehabilitation efforts, and recover ecosystem services including carbon storage.

To further understand the effect of forest fire in the tropical montane forest, a new research project was established in Raden Soerjo GFP (Figure 1). Raden Soerjo GFP has experienced numerous forest fires, including the recent fires of 2015 (Raden Soerjo GFP Agency 2015). Post-fire burn areas were examined to quantify the effects of high-severity fire frequency on post-fire recovery. The research objective was specifically to assess high-severity fire impacts on post-fire forest recovery following different fire frequency (first-entry and second-entry fires), and time since fire (two-, five- and eight-years post-fire).

MATERIALS AND METHODS

Study site

Three study sites were located in Raden Soerjo GFP, East Java, Indonesia, all of which burned at least once since 2009 in high-severity fires (Figure 1). Raden Soerjo GFP is a Conservation Management Unit under the Agency of Forestry, East Java Province, that contains both Mount Arjuno–Welirang and Mount Bromo Tengger Semeru. It is within the Bromo Tengger Semeru-Arjuno Biosphere Reserve, designated as a World Biosphere Reserve in 2015 by UNESCO. The purpose of a conservation forest is to maintain the biodiversity and all other ecosystem services (Undang-Undang Republik Indonesia, 1990). Geographically, Raden Soerjo GFP is located at 7° 40′ latitude and 112° 30′ longitude, and is 27,868 hectares in size. Raden Soerjo GFP ranges in elevation from 1,000 to 3,340 m and is characterised by pronounced cycles of wet and dry seasons. The wet season is typically November–April and is followed by the dry season, May–October. Mean annual precipitation is 2,500–4,500 mm and mean annual temperature varies between 5–15 °C (Raden Soerjo GFP Agency 2015).

Tropical montane forest vegetation dominates the Raden Soerjo GFP ecosystem. Typical native tree species composition includes *Casuarina junghuniana* (cemara gunung), *Litocarpus sundaicus* (pasang), *Engelhardia spicata* (kukrup), *Pygeum parviflorum* (nyampuh), *Trema orientalis* (anggrung), *Pinus merkusii* (pinus), *Malotus* sp. (tutup), *Ficus* sp. (dampul), *Acmena acuminatissima* (kelis) and *Macropanax dispermum* (endos endogan). The understory is composed of herbaceous plants, grasses and shrubs, such as *Panicum repens* (kolonjono), *Chromolaena odorata* (grebes), *Eupatorium riparium* (teh-tehan) and *Vaccinium varingiaefolium* (manis rejo). Numerous non-native, invasive overstory and understory species are also found in these forest ecosystems.

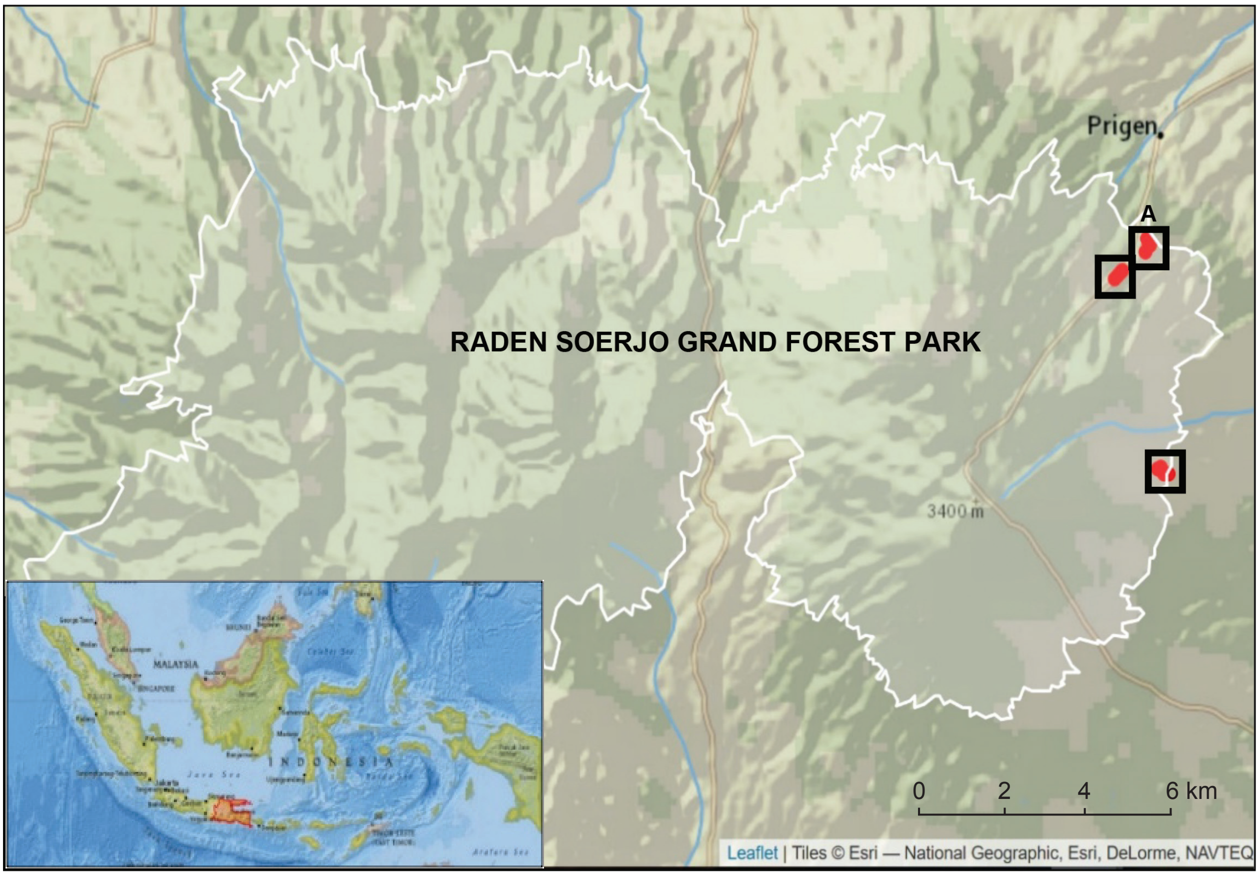

Figure 1 Study site located in the Province of East Java, shown in red on the inset map (right), Raden Soerjo Grand Forest Park (Taman Hutan Raya) and location of three high severity burn units sampled in the study (left); A = TSF8/BI, B = TSF2/BII, C = TSF5/BII

Field site selection

The general location sites from Rahmasari (2011) were used to locate areas first burned in 2009 at high severity (Table 1). To validate the high-severity burn units reported in Rahmasari (2011), 30-meter spatial resolution, cloud-free Landsat images acquired between 2008–2017, were used. These encompassed six Landsat 7 ETM + and 8 OLI/TIRS collection 1 level 1 images. Acquired images were processed using ENVI 5.4 software, including radiometric calibration, atmospheric correction using Fast Line of Sight Atmospheric Analysis of Spectral Hypercubes (FLAASH) and cloud masking. To assess burn severity, areas that burned in 2009, 2012 and 2015 were examined using normalised burn ratio (NBR) and difference of NBR (dNBR), an index of NBR between pre- and post-fire spectral signals (Roy et al. 2006). Near-infrared (NIR) and shortwave infrared bands (SWIR) were used to calculate both fire severity indices, using Formula 1 for NBR and Formula 2 for dNBR (United States Geological Survey 2004).

$$\mathrm{NBR} = \frac{(\mathrm{NIR} - \mathrm{SWIR})}{(\mathrm{NIR} + \mathrm{SWIR})} \qquad (1)$$

$$\text{dNBR values} = \text{NBR pre-fire} - \text{NBR post-fire} \qquad (2)$$

where NIR was band 4 in Landsat 7 ETM+ and band 5 in Landsat 8 OLI/TIRS, and SWIR was band 7 for both Landsat 7 ETM+ and Landsat 8 OLI/TIRS. The threshold of high-severity fire was defined as values > 0.25 and as at least 50% of the investigated units having burned with predicted high ecological damages, including a high level of tree mortality and dark soil >1 cm in depth (DeBano et al. 1998, Keeley 2009). Areas within the 2009 high-severity areas that also burned in 2012 or 2015 were identified to assess forest recovery along a fire chronosequence and across areas burned once or twice.

Climate variables during the fires

To better characterise the weather during the years of fire occurrence, climate data for the period 1999–2017 were obtained from the Indonesia Meteorological, Climatological and Geophysical Agency, including mean temperature, precipitation and relative humidity. Wind velocity and wind direction were also obtained for fire years 2009, 2012 and 2015.

Field data collection

Thirteen plots were randomly selected and established in July 2017 across three time since fire and two burn entry variables (Table 1). Each plot consisted of a nested circular plot with radius 20 m (0.1 ha, overstory plots) and smaller, sub-plot of radius 5 m (0.008 ha, sapling plots) established using the same center point. Three square sub-plots of 1 m × 1 m (0.0003 ha, understory plots) were established 10 m from the center point at 0 °, 120 ° and 240 °.

All standing (live and dead) trees ≥ 20 cm diameter at breast height (DBH, 1.4 m height above ground) and poles (10.0–19.9 cm DBH) in each overstory plot were measured, and DBH and species of each stem were recorded. Saplings (< 10 cm DBH and ≥ 1.4 m tall) were tallied by species in the sapling plots, and seedlings (< 1.4 m tall) by species in each understory plot. Finally, the coverage (%) of epiphytes, woody lianas, non-woody lianas, pandanus, palms and understory in each understory plot were recorded. For understory cover, coverage by nine life form categories were recorded, i.e. herbaceous (HE), woody shrubs (WS), grasses (GR), ferns/fern allies (FE), rock (RO), mosses/lichens (MO), bare ground (BG), wood (WO) and leaves/organic matter (OM). Each life form coverage percentage was recorded to a maximum of 100% due to extensive ground cover layering in these forests, yielding a possible total percent cover of 900%.

Data analysis

The analysis summarised burn unit variability of abiotic factors, namely climate, elevation, slope and aspect (Table 1), but did not test for significant differences between categories or burn units, given low sample sizes and influence of topography on burn patterns. All statistical analyses were completed in R 3.4 (R Core Team 2017).

To address the research objective, data from the burn units were analysed, based on the fire chronosequence (time since fire) and fire frequency (burn entry). In the fire chronosequence, three burn units were investigated including (1) TSF2, a burn unit which experienced high-severity fires first in 2009 and second in 2015 (last fire occurrence 2 years ago), (2) TSF5, a burn unit which experienced high-severity fires first in 2009 and second in 2012 (last fire occurrence 5 years ago) and (3) TSF8, a burn unit which experienced high-severity fire only in 2009 (last fire occurrence 8 years ago). To address the effect of burn entry, burn units were categorised as (1) BI, burn units that experienced one high-severity fire in 2009 and (2) BII, burn units that experienced two high-severity fires (Table 1).

Using both, the time since fire and the burn entry variables, post-fire regrowth stand structure were enumerated based on the mean tree density (stems ha^{-1}) by size class (trees, poles, saplings, and seedlings), basal area of live trees and poles (m^2 ha^{-1}), mean percent understory cover by life form category and all life forms combined. Due to low sample size and concerns around statistical test assumptions, observed trends were interpreted using variable means with corresponding standard errors. To define the overstory stand species composition and size class distribution, diameter class distributions were developed for all trees, by species, using four size classes: 10–19.9 cm, 20–29.9 cm, 30–39.9 cm and 40+ cm.

Importance value (IV), Shannon-Wiener diversity index (H'), evenness (E) and species richness or Margalef index (d) were calculated based on the abundance of stems by species recorded in each time since fire unit (Curtis & McIntosh 1950, Shannon & Weaver 1963, Pielou 1966, Margalef 1968).

Importance values were computed as the sum of relative density, relative frequency and relative dominance (Smith 1980), and calculated separately for trees, poles, saplings and seedlings. Maximum value of IV can be up to 300% for tree and pole size classes and 200% for sapling and seedling size classes (Curtis & McIntosh 1950, Smith 1980). Dominant species was defined as the most abundant species (highest IV by size class and species) recorded in each time since fire unit (Smith 1980).

The Shannon-Wiener species diversity index (H') was calculated to distinguish species variability in times since fire units using Formula 3 (Shannon & Weaver 1963).

$$H' = -\sum_{i-1}^{s}(p_i)(\log_2 p_i) \qquad (3)$$

where s is the number of species, p_i is the proportion of individuals of the total sample referring to the i^{th} species, and $\log_2$ is the natural logarithm base 2. The exponents of all total H' values of each species in the site were computed to explore species diversity based on each time since fire unit. An evenness index (E, relative abundance of each species) was calculated according to Pielou (1966) using the following formula:

$$E = \frac{H'}{\log S} \qquad \text{(Formula 4)}$$

where H' is the Shannon-Wienner species diversity index and S is the total number of species. Evenness can be unlimited or undefined if only one species is present.

Finally, the species richness (d) or Margalef index were calculated, as the total number of species recorded in the unit (Margalef 1968) using the following:

$$d = \frac{S - 1}{\log N} \qquad \text{(Formula 5)}$$

where S is the number of species and N is total number of all individuals in the burn unit.

RESULTS

The fire years, 2009, 2012 and 2015 had below-average daily precipitation and relative humidity (Figure 2). Additionally, the high-severity fires occurred during the dry season, with precipitation less than 100 mm $month^{-1}$ (between June and October), and wet season precipitation during the same years totaled over 400 mm $month^{-1}$. Wind direction during the the fires was easterly, but no pattern emerged related to wind velocity.

Post-forest fire regrowth, time since fire and burn entry

Trees ha^{-1} varied between 2.5 and 35 across burn units while total basal area varied between 0.34 and 3.03 m^2ha^{-1} (Table 2). It was observed that of 17 different tree species, thirteen were native and four were non-native.

Increasing trends were observed in mean tree, sapling and seedling densities with increasing time since fire, from TSF2 (lowest) to TSF8 (highest) (Table 2). First, pole density increased,

Table 1 High-severity burn units sampled in Raden Soerjo Grand Forest Park, East Java, Java, Indonesia, showing management block location, years burned, time since last fire, burn unit identifiers used in analysis, number of plots installed, and mean plot elevation, slope and aspect

Location	Burning occurrence		Time since fire (years)	Burn unit identifier		# Plots	Elevation (m)	Slope (%)	Aspect (°, direction)	
	First entry	Second entry		Time since fire	Times burned					
Block of Dali Pentongan (Village of Ledug District of Prigen, Region of Pasuruan)	2009	-	8	TSF8	BI	4	1148	26	49	ENE
Block of Gumandar (Village of Jatiarjo District of Prigen Region of Pasuruan)	2009	2012	5	TSF5	BII	5	1527	14	25	NEE
Block of Sembung Rubuh (Village of Pecalukan Distrcit of Prigen, Region of Pasuruan)	2009	2015	2	TSF2		4	1465	16.5	30	NEE

TSF2 = two-years post-fire, TSF5 = five-years post-fire, TSF8 = eight-years post-fire, BI = single burn entry, BII = two burn entries, NE = north east, ENE = east-northeast, NNE = north-northeast

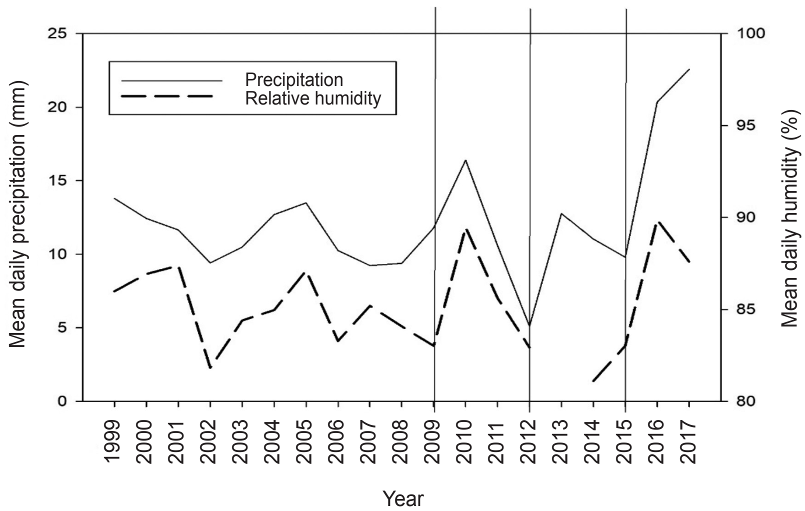

Figure 2 Mean daily precipitation and relative humidity for Raden Soerjo Grand Forest Park, East Java, Java, Indonesia from 1999–2017; note the scale differences between the variables; vertical lines indicate fire years (2009, 2012, 2015)

and then decreased from TSF2 to TSF5, and TSF8 (Table 2). Observations with respect to burn entries varied. Single-entry burns had higher values in all structural variables when compared to second-entry burn units, except for poles where second-entry burn units were highest (Table 2).

Of the four diameter classes, TSF2 and TSF5 were represented by just two, while all four diameter classes were represented in TSF8 (Figure 3a). The diameter class distribution of TSF5 contained two classes, with lower stem densities in the larger size class (Figure 3a). In TSF8, poles and trees were distributed throughout

Table 2 Total and mean stand density per hectare by size class, basal area of live trees and poles, and percent cover of understory across the fire chronosequence

	TSF2		TSF5		TSF8		BI		BII	
	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE
Overstory (stems ha^{-1})										
Trees	2.50	(2.50)	4.00	(4.00)	35.00	(17.08)	35.00	(17.08)	3.33	(2.36)
Poles	7.50	(4.79)	54.00	(31.08)	12.50	(12.50)	12.50	(12.50)	33.33	(18.41)
Saplings	312.50	(148.78)	2175.00	(1017.50)	4000.00	(1210.85)	4000.00	(1210.85)	1347.22	(631.14)
Seedlings	1666.65	(1666.65)	3333.30	(3333.30)	21666.45	(12056.84)	21666.45	(12056.84)	2592.57	(1906.58)
Large stems (trees and poles)	10.00	(7.07)	58.00	(32.47)	47.50	(12.50)	47.50	(12.50)	36.67	(19.29)
Regeneration (saplings and seedlings)	1979.15	(1566.09)	5508.30	(4130.72)	25666.45	(12935.20)	25666.45	(12935.20)	94459	(2352.20)
Basal area (m^2 ha^{-1})										
Trees and poles	0.34	(0.29)	1.07	(0.56)	3.03	(1.50)	3.03	(1.50)	0.74	(0.34)
Understory cover (%)										
Herbaceous	0.00	(0.00)	0.00	(0.00)	34.17	(3.08)	34.17	(3.08)	0.00	(0.00)
Woody shrubs	18.83	(10.03)	21.00	(6.86)	20.00	(7.82)	20.00	(7.82)	20.04	(5.48)
Grasses	95.83	(3.15)	83.00	(5.33)	12.17	(6.02)	12.17	(6.02)	88.70	(3.83)
Ferns	17.08	(8.12)	19.67	(4.03)	6.25	(5.71)	6.25	(5.71)	18.52	(3.96)
Rocks	53.75	(20.75)	0.00	(0.00)	0.42	(0.42)	0.42	(0.42)	23.89	(12.69)
Mosses/lichens	68.75	(22.95)	0.00	(0.00)	0.00	(0.00)	0.00	(0.00)	30.56	(15.29)
Bare ground	0.00	(0.00)	0.00	(0.00)	4.58	(2.08)	4.58	(2.08)	0.00	(0.00)
Downed wood	0.50	(0.1)	13.00	(9.4)	29.17	(5.51)	29.17	(5.51)	7.44	(5.42)
Organic matter	93.75	(1.85)	100.00	(0.00)	81.67	(5.00)	81.67	(5.00)	97.22	(1.33)
All understory	348.5	(33.21)	236.67	(6.81)	188.42	(10.59)	188.42	(10.59)	286.37	(24.14)

TSF2 = two-years post-fire, TSF5 = five-years post-fire, TSF8 = eight-years post-fire, BI = single burn entry, BII = two burn entries four diameter classes, with a peak in the 20–29.9 cm class (Figure 3a). More diameter classes were represented after a single burn (four) than after a second burn entry (three) (Figure 4a). The single-entry burn contained classes 10–19.9 cm through 40+ cm, with 20–29.9 cm having the highest density (Figure 4a). Conversely, a steep reversed-J shaped distribution was found, with three diameter classes in the second-entry burn units (Figure 4a). Total basal area of live trees and poles increased with increasing time since fire (Table 2, Figure 3b) and decreased by 24% with increasing fire frequency (Figure 4b).

Species composition

Four tree species were observed in TSF2, consisting of two native species and two non-native species (Table 5). Five unique species were found in TSF5 (Table 3), including four native species and one non-native species (Table 3). *Acacia decurrens* dominated TSF5 among all size classes (Table 5). More species diversity was found in TSF8 (also single-entry burn unit) than in TSF2 or TSF5 (Figures 3a and 5a), primarily composed of native species (10 vs. 2 non-native). The second-entry burned units, BII, contained

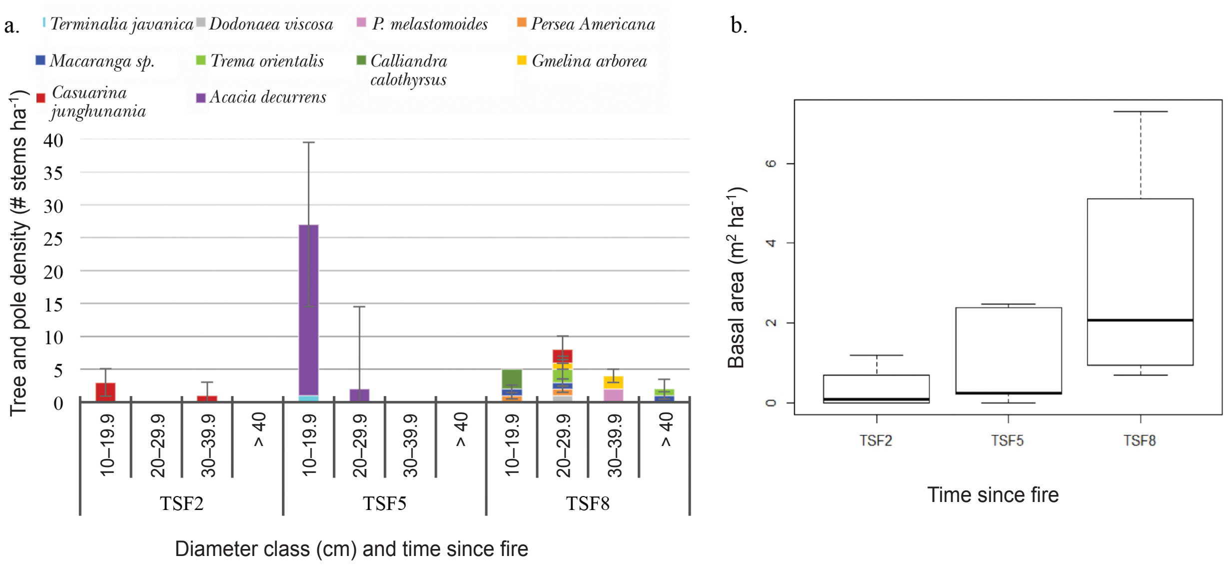

Figure 3 Time since fire (a) diameter class distribution by species and size class and (b) basal area, bars and lines represent category means and standard errors, TSF2 = two-years post-fire, TSF5 = five-years post-fire, TSF8 = eight-years post-fire

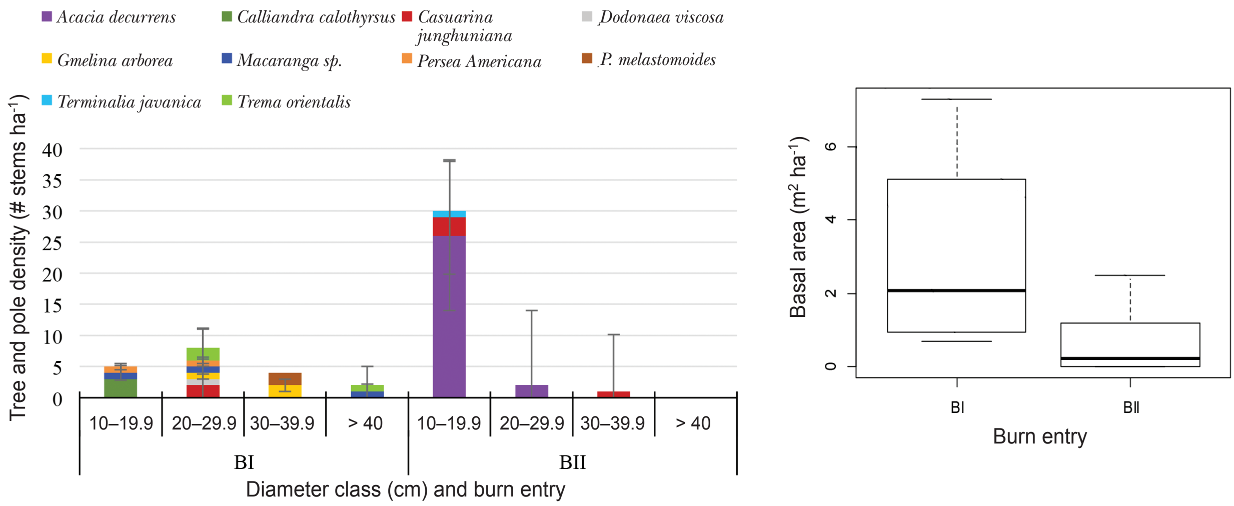

Figure 4 Burn entry (a) diameter class distribution by species and size class and (b) basal area, bars and lines represent category means and standard errors, BI = single burn entry, BII = two burn entries

Table 3 Importance values by species, native status, and tree size class for time since fire and burn entry

Species	Local name	Native/ non-native	Importance value (%)								
			Time since fire					Burn entry			
				Tree	Poles	Sapling	Seedling		Tree	Poles	Sapling	Seedling
Casuarina junghuniana	Casuarina	N	TSF2	300	300	53.57	-	BII	108.89	191.11	9.27	
Calliandra calothyrsus	Calliandra	NN		-	-	78.57	-				16.41	
Cinnamomum cintok	Sintok	N		-	-	67.86	-				10.33	
Eucalyptus sp.	Eucalyptus	NN		-	-	-	200					78.57
Acacia decurrens	Acacia	NN	TSF5	300	236.21	131.61	200		191.11	167.76	111.25	121.43
Terminalia javanica	Ketepeng	N		-	63.79	11.15	-			103.02	8.21	
Moringa oleifera	Keloran	N		-	-	12.30	-				9.27	
Dodonaea viscosa	Kesek	N		-	-	33.79	-				27.05	
Gmelina arborea	Gmelina	N (planted)		-	-	11.15	-				8.21	
Gmelina arborea	Gmelina	N (planted)	TSF8	58.22	-	-	-	BI	58.22	-	-	-
Macaranga sp.	Tutup	N		40.71	88.62	119.53	-		40.71	88.62	119.53	-
Trema orientalis	Anggrung	N		55.46	-	11.89	-		55.46	-	11.89	-
Ficus spp.	Banyon beling	N		44.98	-	-	-		44.98	-	-	-
Persea Americana	Avocado	NN		31.42	79.60	-	-		31.42	79.60	-	-
Casuarina junghunania	Casuarina	N		45.14	-	-	-		45.14	-	-	-
Calliandra calothyrsus	Calliandra	NN		-	131.78	103.65	148.71		-	131.78	103.65	148.71
Dodonaea viscosa	Kesek	N		24.07	-	-	15.95		24.07	-	-	15.95
Ficus septica	Awar	N		-	-	30.82	-		-	-	30.82	-
Swietenia macrophylla	Mahogany	N		-	-	-	19.40		-	-	-	19.40
Mangifera indica	Mango	N (planted)		-	-	-	15.95		-	-	-	15.95
Debregeasia longifolia	Mencokan	N		-	-	11.89	-		-	-	11.89	-

TSF2 = two years since fire; TSF5 = five years since fire, TSF8 = eight years since fire, BI = first – burnt, BII = second – burnt nine species including six native species and three non-native species (Table 3).

Species diversity, species richness and evenness indices differed among time since fire units (Figure 5). An increase in species occurrence was found with additional time since fire, coupled with high variation between burn units (Figure 5). Low tree diversity indices were found in TSF2 and TSF5 units, with primarily one species recorded in most size classes (Figure 5). The highest species diversity, richness and evenness indices were assessed in the tree-sized class of TSF8 (Figure 5). Fewer species were found occurring in BII than in BI. No clear trends were observed between BI and BII in regeneration size classes in terms of species diversity, richness or evenness (Figure 6). Large stem species diversity, species richness and evenness was higher in BI than in BII (Figure 6).

In the understory, variable trends were largely observed in cover with time since fire, with many variables exhibiting no cover in several categories (Table 2). The only clear trends observed were a decrease in mean percent cover of grasses and an increase in downed wood with increasing time since fire. Mean percent cover of herbaceous, bare ground and downed wood were found to be higher in single burn entries while grasses, ferns, rocks, mosses and lichens, organic matter and the sum of all understory cover categories were found to decrease with repreated burn entry (Table 2).

DISCUSSION

Relatively, slow forest recovery was found following fire, and native forest recovery was hindered by a second burn and non-native species. Rahmasari (2011) found higher basal

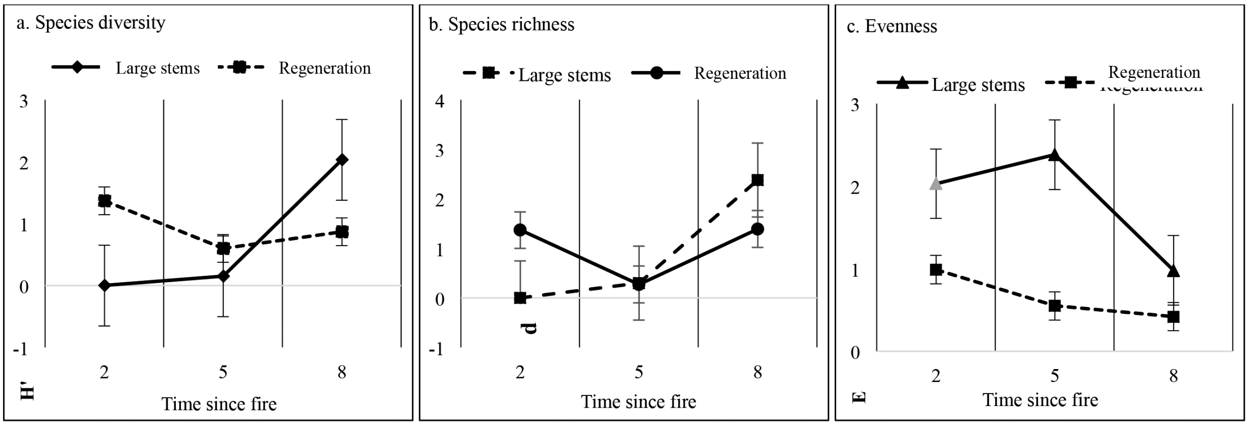

Figure 5 Species diversity (H'), species richness (d), and evenness (E) indices by time since fire and stem size class; points represent means and lines represent +/- one standard error of the mean

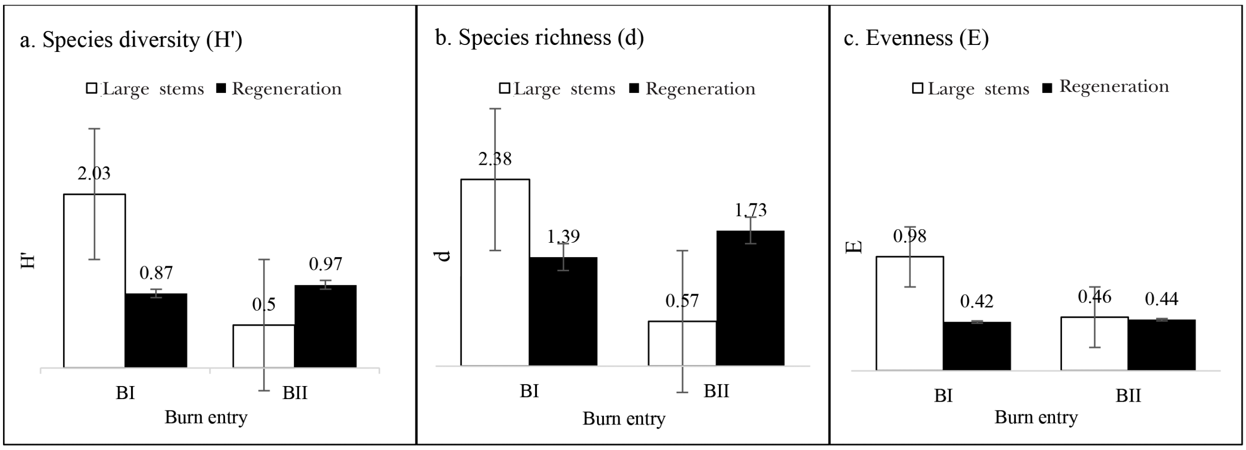

Figure 6 (a) Species diversity (H'), (b) richness (d), and (c) evenness (E) by burn entry and stem size class; bars and lines represent means +/- one standard error, by category

area, lower stem densities and uneven-sized, reverse-j diameter distribution in similar, unburned forest. Increasing time since fire increased stem density and basal area in the study, as expected. Fire severity may have been enhanced by the dry seasons during which they occurred (Cochrane 2003). Most forest fires in the area are caused by abandoned campfires or intentional burning by poachers looking to flush game species, which peak during the dry season (Raden Soerjo GFP Agency 2015).

Stem density across all size classes increased with prolonged time since fire, and more large stems were present following the single-entry burn, than units with a second-entry. Longer post-fire intervals may better promote the recovery of residual and large trees, stressed during the fire, which would also assist with increases in understory tree recruitment and establishment, facilitating initial forest recovery (Donato et al. 2009, Verma et al. 2017). Recruitment and establishment of the understory may be a result of natural regeneration or enrichment planting during post-fire restoration efforts. Lower seedling and sapling densities found in the earlier post-fire years may be a result of competition given the abundant grass coverage (> 90%) (Baudena et al. 2015). Some aggressive native (e.g. *Imperata* and *Sacharum*) and non-native (e.g. *Euphotarium* and *L. camara*) grass species were observed in the burn units, however, the observations were limited as understory cover by species was not recorded.

The second-entry burn increased overall mortality but enhanced regeneration, a finding supported by previous research in other systems (Cochrane & Schulze 1999, Greene et al. 1999). The occurrence of fewer residual trees following the second-entry burn may result in a type of conversion away from forest, if tree recruitment is delayed or unsuccessful given the harsh conditions present on the sites (dry, north-facing slopes), with potential for less seed dispersion by herbivores or mammals, depending on species silvics and gap size, among other factors (Donato et al. 2009, Arroyo-Rodríguez et al. 2017).

In this study, the diameter distribution shifted towards complete dominance by small size classes (seedlings, saplings and poles), while in unburned forest it is expected to find all size classes present, including stems over 100 cm in diameter, higher overall basal area and less overall density (Rahmasari 2011). Some trees were present in the overstory eight-years post-fire, creating a more uneven-sized distribution with comparatively higher total basal areas. These conditions closely resemble second-growth tropical forests of the region than even-sized, small statured structures (Chazdon 2014). The second-entry burn units were dominated by pole-sized *Acacia decurrens,* a fire-tolerant species and had overall less basal area than the first-entry burn.

Two years following fire, a *Casuarina–Calliandra–Eucalyptus* association was found. The native species *Casuarina junghuniana* is a long-lived, shade-intolerant pioneer tree species of Javan tropical montane that can survive fire (Smiet 1992). However, given that we found no *C. junghuniana* seedlings and declining abundance with increasing stem size classes, competition from other species, namely *Eucalyptus sp.* and *Calliandra calothyrus,* may be hindering post-fire recovery (Zouhar et al. 2008). Both of these species are shade-intolerant pioneer trees that can establish quickly following disturbances, such as high-severity fire, and may also be present in unburned forest (National Research Council 1983, Rahmasari 2011, Saharjo & Gago 2011).

Five-years post-fire and following the second-entry burn, it was observed that the non-native species, *Acacia decurrens,* dominated many size classes. Native species present in similar or lesser abundances may be suppressed by the domination of *Acacia decurrens,* an allelopathic tree species that can grow rapidly after a fire or volcanic eruption, creating dense canopies and light limiting conditions in the understory (Padmanaba et al. 2017, Sunardi et al. 2017). Besides its good seed germination, *Acacia decurrens* can also sprout immediately following fire (Hapsari et al. 2014, Afrianto et al. 2017). These characteristics allow continued growth and reproduction of *Acacia decurrens* and it is unlikely that other species will successfully establish and dominate. Furthermore, additional burns are likely to continue perpetuating the dominance of *Acacia decurrens* in these areas, particularly given that most native species do not thrive in the post-fire environment (Sunardi et al. 2017). For example, large stems of *C. junghuniana* may initially survive the fire but have declining seed production after burns (Nieuwstadt 2002).

It was expected to find decreasing species diversity and richness as time since fire increased, and when competition reduced or eliminated some species. Observations showed that five years post-fire did not support higher species diversity and richness over those in two-year post-fire, but the eight-year post-fire unit indicated a trend of increasing species diversity and richness, along with decreasing evenness, over the two- and five-year post-fire units. In the different burn entries, single-entry burns were expected to support higher diversity and richness than second-entry burns. However, single-entry burns exhibited less diversity and richness of tree regeneration than second-entry burns. Given that the single-entry unit was also eight-years post-fire, these patterns may be driven by a lack of overlapping species in the two- and five-year post-fire units, or the presence of the non-native species, *Acacia decurrens*, which dominated the five-year post-fire unit.

Burn entry exhibited a clear influence on most understory life form cover values, particularly grass and organic matter. Second-entry burns exhibited increased understory cover, particularly grasses, likely due to their resistance to high-severity fires and their ability to live in highly disturbed areas (Dennis et al. 2001). The grasses, shrubs and herbaceous species reported in this study were generally fire-resistant and shade-intolerant. Such species can inhibit successful establishment and growth of seedlings and saplings (D'Antonio & Vitousek 1992, Otsamo 2000, Royo & Carson 2006). *Imperata*, *Euphoratrium* and *Sacharum* were found to be the three most important understory genus, suggesting that their dense cover may inhibit tree regeneration through competition for light and soil resources (Royo & Carson 2006). Coupled with the low residual large tree density and multiple entry, high-severity burned areas are more susceptible to conversions to non-forest, which may occur as quickly as 10–15 years post-fire (D'Antonio & Vitousek 1992, Chazdon 2003, Oliveras et al. 2014).

The study results and conclusions were restricted to the East Javan dry tropical forests. In addition, there was limited information regarding the cause, influence and origin of individual fires, as well as total fire-affected area and landscape-scale severity, all of which are important when examining forest structure and composition. Furthermore, the paucity of the sampling units along with a lack of representative unburned units (i.e., untreated controls) resulted in high variability among units and reduced the ability to make inferences about post-fire response. Lastly, to preform statistical tests, assumptions of homogeneity of variances and normality were necessary to make comparisons using confidence intervals, limiting to inferences about general characteristics of each unit and trends between units. Standard errors were provided to facilitate future work and to document observed variability in sample means.

The findings represented early post-fire forest succession, and further research is needed to determine the impacts of these fires on long-term forests structure and composition (i.e., recovery). At least 10 years is recommended to gain a better understanding of successional development (Donato et al. 2009). In addition, the fire-chronosequence study design may have inherent limitations as each study site may have had differing initial conditions, and thus capture slightly different recovery pathways and different species occurrences and assemblages. However, the chronosequence design still provided useful insights into species occurrence, differentiation and two- and eight-years post-fire effects.

In this study, lower total tree density and basal area were found, compared to the forest inventory reported by the Indonesian Ministry of Forestry, and in comparable to unburned forest (Rahmasari 2011, Kementerian 2014). The Ministry of Forestry suggests a Javan dry secondary forest should have at least 569 trees ha^{-1} and 15.69 m^2 ha^{-1} of basal area, while the results showed a mean of only 150 trees ha^{-1} and 11.53 m^2 ha^{-1} (KLHK 2014). In short, the sites were no longer considered forested areas, by this definition, and are in need of restoration/rehabilitation.

Re-entry of high-severity burns with short return intervals may result in multiple secondary successional trajectories (Nieuwstadt 2002, Donato et al. 2009). Postponed forest maturation may result, if fewer trees were to establish (Chua et al. 2013). This may be the case where abundant and aggressive grass cover outcompetes tree recruitment. Prolonged delay of early stage forest recovery, or the potential disappearance of native tree recruitment, can influence the natural post-fire forest trajectory towards alternative compositions and functional group dominance, such as grass-dominated conditions or monocultures of *Acacia decurrens* (Arroyo-Rodríguez et al. 2017). Given the possibility of conversion, frequent burning may create fireprone conditions that are more susceptible to subsequent disturbances (Dennis et al. 2001, Trumbore et al. 2015).

CONCLUSIONS

The study provided an increased understanding of early, post-fire recovery in tropical montane forests. Fire frequency is expected to increase as climate change and drought interact with humans in these landscapes. Greater understanding is needed if managers are to improve the post-fire forest recovery by implementing appropriate silvicultural techniques, e.g. reforestation (Chazdon 2003).

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