Author/s:
Volkova, L;Krisnawati, H;Qirom, MA;Adinugroho, WC;Imanuddin, R;Hutapea, FJ;McCarthy, MA;Di Stefano, J;Weston, CJ

Title:
Fire and tree species diversity in tropical peat swamp forests

Date:
2023-02-01

Citation:
Volkova, L., Krisnawati, H., Qirom, M. A., Adinugroho, W. C., Imanuddin, R., Hutapea, F. J., McCarthy, M. A., Di Stefano, J. & Weston, C. J. (2023). Fire and tree species diversity in tropical peat swamp forests. Forest Ecology and Management, 529, https://doi.org/10.1016/j.foreco.2022.120704.

Persistent Link:
https://hdl.handle.net/11343/334269

Fire and tree species diversity in tropical peat swamp forests

Liubov Volkova[1*], Haruni Krisnawati[2], Muhammad A. Qirom[2], Wahyu C. Adinugroho[2], Rinaldi Imanuddin[2], Freddy Jontara Hutapea[1,2], Michael A. McCarthy[3], Julian Di Stefano [1], Christopher J. Weston[1]

1 - School of Ecosystem and Forest Sciences, Faculty of Science, The University of Melbourne, Creswick, Victoria, 3363, Australia

2 - Research Center for Ecology and Ethnobiology, Research Organization for Life Sciences and Environment, National Research and Innovation Agency (BRIN), Jl. Raya Jakarta-Bogor KM. 46, Cibinong, Bogor, 16911, Indonesia

3 - School of Ecosystem and Forest Sciences, Faculty of Science, The University of Melbourne, Parkville, Victoria, 3052, Australia

*Corresponding author: lubav@unimelb.edu.au

Abstract

Tropical peat swamp forests contain diverse plant communities that support many endangered flora and fauna species. These forests are increasingly threatened by disturbances such as drainage, logging, and subsequent fire, yet little is known about tree species recovery from these threats.

Tree species richness and diversity were measured in peat swamp forests of Central Borneo regenerating naturally after fires from 139 chronosequence plots. Our findings indicated that two to three decades after fire, similar levels of richness and diversity to relatively undisturbed reference forest are reached within sites. However, when averaged across landscapes our modelling shows that fires of average frequency of 50 years, or even 100 years, can substantially suppress tree species richness and diversity. Thus, even rare fires can reduce the species richness and diversity of tropical peat swamp forests.

Two groups of tree species drove the difference in richness and diversity between disturbed and reference forests: 'decreasers' (occupying recently burnt forest and declining with fire exclusion) and 'increasers' that were only present in less disturbed forests and absent or rare in recently burnt forests. The decreasers were more common than the increasers.

We conclude that peat swamp forests can recover adequately from fires through natural regeneration, and it is possible that artificial planting may speed up this process. Where possible, excluding fire from tropical peat forests will help to maintain the diversity of these threatened ecosystems and promote their recovery.

Keywords: south-eastern Asia, Kalimantan, Species richness, Shannon diversity, peatland, *Syzygium garcinifolia, Stemonourus scorpioides*, *Litsea oppositifolia*

Introduction

Approximately 13.4 million ha of tropical peatlands occurs in Indonesia, with exceptionally deep peatlands in Kalimantan on the island of Borneo (Anda *et al.* 2021). Peat swamp forests, one of the categories of peatlands (Page *et al.* 1999), contain distinctive plant communities and provide critical habitat for numerous animals, including threatened species such as orangutans, *Pongo spp*., Sumatran tiger, *Panthera tigris sumatrae*, and Bornean agile gibbon, *Hylobates albibarbis* (Morrogh-Bernard *et al.* 2003; Husson *et al.* 2018).

In recent decades large areas of tropical peat swamp forest have been degraded through timber removal, agricultural expansion (including burning), and through associated access and drainage activities (Rieley *et al.* 2008). The drainage of water from peat domes has allowed periodic drying of surface peat and the initiation of cycles of human burning to maintain areas tree-free for future agricultural development (Morrogh-Bernard *et al.* 2003; Fuller *et al.* 2004; Yule 2010; Hoscilo *et al.* 2011; Posa *et al.* 2011). The net result of these actions is that large areas of peatlands, that were once naturally resistant to fires, become fire-prone systems (Turetsky *et al.* 2015). In addition, climate change has increased the rate of peatland drying with several years of drought over the last two decades (Barber and Schweithelm 2000; Hirano *et al.* 2007). The large scale wildfires of 1997-98 throughout the tropics in Amazonia and Southeast Asia were linked to changes in land use and extended dry seasons that were likely caused by climate change (Cochrane 2002; Page *et al.* 2002). In the Southeast Asian region, fires in 2002 and 2006 affected more that 8 million ha of peatlands (Page *et al.* 2009). Further extensive fires in 2015 in Indonesia burnt more than 2.5 million ha of forests and caused economic losses up to USD 16.1 billion (World Bank 2016). Tropical peat swamp forests are particularly susceptible to fires because peat is extremely flammable when dry (Posa *et al.* 2011). Throughout the last two decades more than 14% of the area of peat swamp forests in Sumatra and Kalimantan has been burnt at least once, and 23% has burnt more than twice, with an average fire return interval of less than 50 years (Vetrita and Cochrane 2020). In some areas of the former Mega Rice project of Central Kalimantan fires have burnt consecutively over five years (Page *et al.* 2009). Once affected by fires it is unknown how quickly and completely peat swamp forests can recover. Reduced canopy cover, decreased species richness, and high tree mortality are among the effects of fires on peat swamp forests (Yeager *et al.* 2003). Further, the high combustibility of peat and its capacity to smoulder for months destroys seedbanks and underground perennating organs (Page *et al.* 2009) and causes collapse of mature trees. While the detrimental effects of disturbances (such as logging and fire) on plant communities in tropical peat swamp forests have been reported at a plot scale (Yule 2010; Posa *et al.* 2011; Wijedasa *et al.* 2020), landscape scale changes at different fire return intervals has not been predicted. This is an important knowledge gap, as understanding how plant communities recover naturally after fire will help determine the impact of repeated fires in this ecosystem and stimulate the development of rehabilitation strategies after future fire events.

The global ecological importance of peat swamp forests, including acting as carbon sinks and contributing significantly to the global carbon cycle (Page *et al.* 2006), combined with their threatened status and the increasing rate of anthropogenic disturbance, have generated a critical need to characterise their response and recovery from a variety of threatening processes. This study investigates the impact of fire on tree species recovery in the tropical peat swamp forests of Kalimantan, Indonesia. Specifically, the study aimed to: (a) determine the rate of post-fire recovery of species richness and diversity; (b) identify how changes in fire frequency alter tree species diversity and richness at the landscape level; (c) identify tree species most at risk from fire and, (d) provide recommendations for current and future restoration projects.

Material and Methods

Study sites and data measurements

Peat swamp forests growing on either the *Mawas* or the *Sebangau* peat domes near Palangka Raya, in Central Kalimantan, on Borneo Island were selected for this study (Fig.1). Areas of peat swamp forest within and near the Sebangau National Park (NP) with no known fire history were designated as reference forest, noting that Sebangau NP had a history of selective logging prior to declaration of the park in 2004. The peat swamp forests near Tumbang Nusa research station had a history of past logging, and some were burnt in one, two or multiple fires over 1997 - 2015 years (Fig. 1). All burnt peat swamp forests sampled for this study have regenerated naturally. For the analysis, our chronosequence dataset contained 139 plots randomly located, but with time since fire (TSF) corresponding to 7 elapse times: 4 years=14 plots, 5 years=9 plots, 7 years=14 plots, 10 years=19 plots, 16 years=30 plots, 20 years=11 plots, 22 years=12 plots. The other 30 plots were the reference plots (unburnt). The unequal number of plots reflected the difficulties of establishing plots for some TSF categories, e.g., inaccessible flooded areas, or areas converted to plantations or too close to roads or canals. The TSF was determined from fire history data of the Ministry of Environment and Forestry of Indonesia and from consultation with local authorities. All plots were established at least 50 - 100 metres from roads and canals. In each plot, larger trees with a diameter at breast height (DHB) at 1.3 m ≥10 cm were measured within 300 - 400 m^2 plot boundaries, and smaller trees and saplings with DBH <10 cm were measured in smaller sub-plots (40 m^2). All measured trees were identified with either local or scientific names. Among the 126 tree species encountered, about 17 species could not be identified beyond family, so a morphospecies approach that allocated each unknown species a unique code (e.g. *Litsea oppositifolia* and *Litsea sp*. 1) was applied to facilitate the analysis.

Theil's uncertainty coefficient (Theil 1972) was used to calculate the correlation between the TSF and both the fire frequency and inter-fire interval classes, resulting in high correlations in both cases (0.73 and 0.77). This was due to expected associations among the three variables; for example, most recently burnt plots were also frequently burnt and had short inter-fire intervals. Given the confounding among the three fire variables, TSF alone was used due to its simplicity and management relevance for building a better understanding of the recovery of these forests from novel logging and burning disturbance.

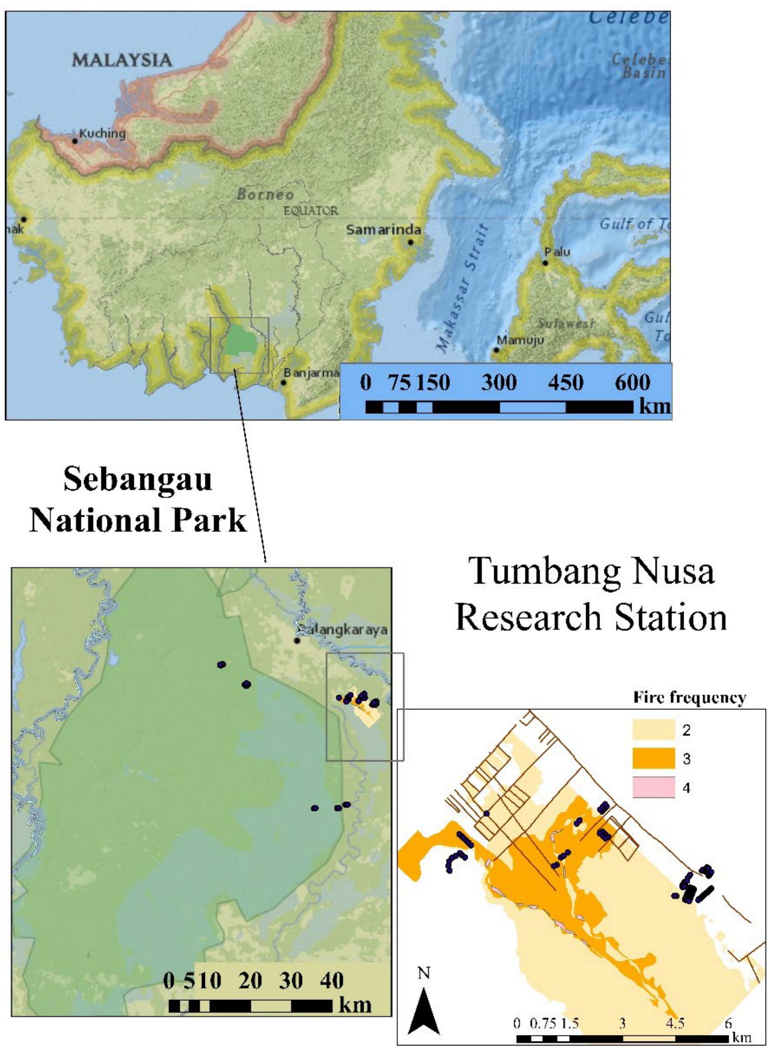

Figure 1. Study site location in Central Kalimantan, Indonesia. Black dots represent study sites.

Data analysis

Tree species richness (number of species) and Shannon diversity were modelled as a function of time since fire (t) by fitting a logistic function given by:

$$f(t) = K/[1 + ((K - P_0)/P_0)\exp[-r \times t]] \qquad \text{Eq (1)}$$

where K is the maximum species richness (or diversity), P_0 is the species richness (or diversity) at zero years since fire, and r controls the rate of increase in species richness (or diversity) with time towards the asymptotic value K. Thus, we used two different versions of equation (1) – one for analysing species diversity and one for analysing species richness.

Statistical models were fitted using Bayesian Markov chain Monte Carlo methods within OpenBUGS. Errors (residuals) were assumed to have a normal distribution. Vague (flat) priors were specified to limit the priors' influence on the estimated parameters. These were uniform on the range 0 to 10 for the parameters r and P_0, and on the range 0 to 100 for the parameter K. The upper limits of these uniform distributions were chosen to ensure they did not constrain the posterior. The precision (reciprocal of the variance) of the residuals was given a gamma prior with both parameters equal to 0.001 (mean=1, variance=1000).

For unburnt sites with a history of selective logging these values were treated as being censored (with a minimum value of 50 years), with times since fire of all sites assumed to be drawn from a common exponential distribution. A statistical model in which the residuals were spatially correlated was also analysed (using the spatial.exp function in OpenBUGS); this model generated qualitatively similar results to the model that ignored spatial correlation, so it will not be discussed further.

The above analysis examines how species richness and diversity vary with time since fire. We also examined how the average species richness and diversity across sites (i.e., averaged over a landscape) is expected to vary with the mean fire interval occurring in the landscape. The expected species richness and diversity across landscapes was predicted by modelling the expected age structure of landscapes subject to random fires and then calculating the average diversity and richness across this landscape. When fires occur randomly with respect to time (independent of time since fire), the time since fire for plots in the landscape is expected to follow an exponential distribution (McCarthy *et al.* 2001), with the probability density of the age structure equal to $\exp(-t/\mu)/\mu$, where μ is the mean fire interval and t is time since fire. The expected species richness and diversity are obtained by multiplying this age structure by the estimated functions of recovery following fire and integrating over time since fire (t):

$$E(R) = \int_0^\infty R(t) \exp(-t/\mu)/\mu \, dt \qquad \text{Eq (2)}$$

$$E(D) = \int_0^\infty D(t) \exp(-t/\mu)/\mu \, dt \qquad \text{Eq (3)}$$

The resulting integrals that define the expected species richness and diversity are functions of the model parameters for $R(t)$ and $D(t)$ (estimated from the fitted values of Eq (1) for each of $R(t)$ and $D(t)$ and the mean fire interval (μ)). These integrals were calculated for mean fire intervals up to 100 years to determine the degree to which random fires will reduce the expected richness and diversity of landscapes below the maximum attainable values (the asymptote K in Eq 1).

For species that were detected at more than 20 sites (23 species), we fitted models of how their recorded abundance changed with time since fire. The log of the expected abundance was modelled as a linear function of time since fire (i.e., abundance changed exponentially with time since fire, either increasing or decreasing). The observed abundances were modelled as being drawn from a Poisson distribution, with the mean given by the expected abundance. Vague priors were assumed for the regression coefficients.

As for the analysis of species richness and diversity, for sites where time since fire was not known precisely, these values were treated as being censored and drawn from an exponential distribution. Species were classified as either increasers or decreases when the 95% credible interval coefficient was either fully above or below zero. Species responses to time since fire were classified as being unclear when the 95% credible interval of the regression coefficient spanned zero. These include cases where the species might be most abundant at intermediate times since fire, although there were no species where such a hump-shaped relationship was clearly appropriate.

Results

One hundred and twenty-six species were identified among a sample of 5226 trees collected across 139 plots. Time since fire had a marked influence on peat swamp forest tree species richness and diversity. Species richness and Shannon diversity were lowest in recently burnt forest and highest in the reference forests (Fig. 2). Both richness and diversity recovered rapidly over time, demonstrating substantial increases from 5 to 15 years since fire, with richness increasing by a factor of three and diversity approximately doubling over the same period. The increase was less pronounced 20 years after fires (Fig. 2).

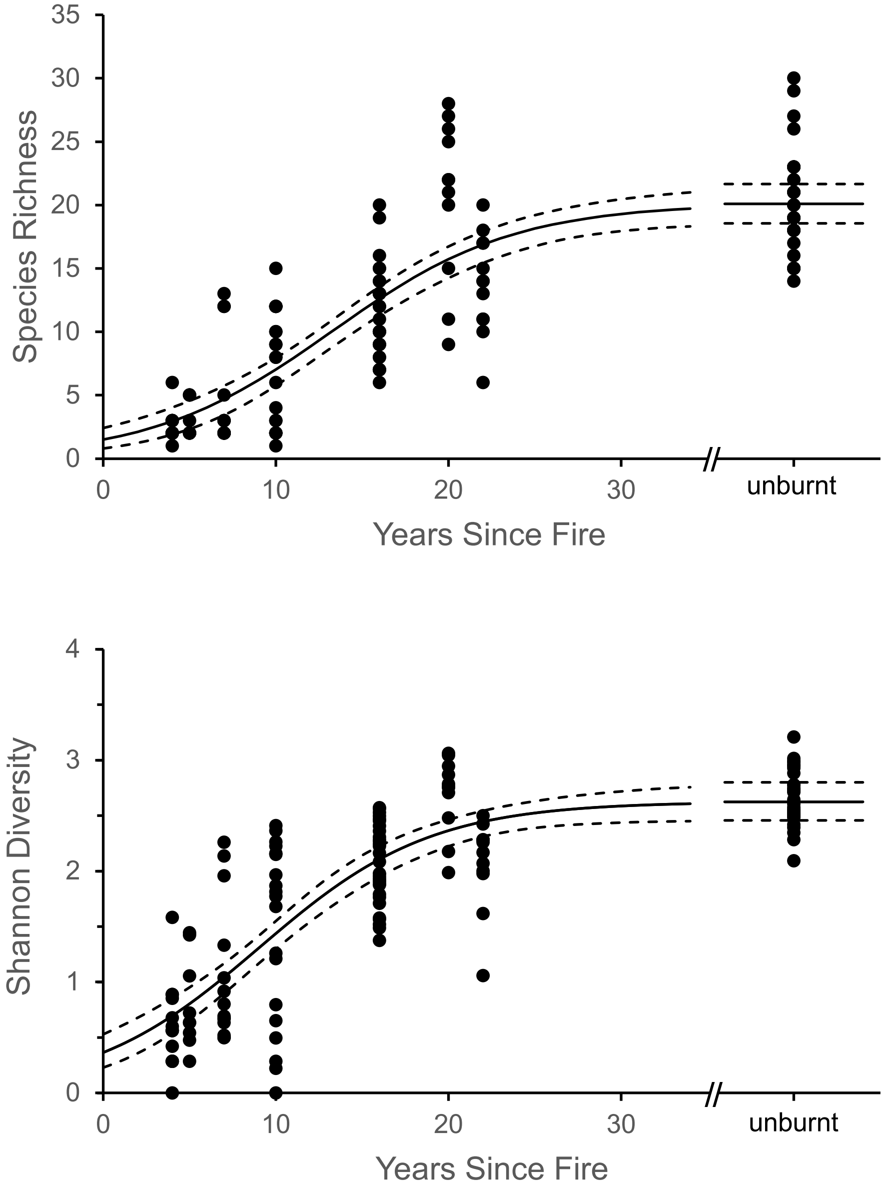

Fig 2. Estimated recovery of species richness and species diversity with time since fire. Dots are the values measured at each plot, the lines are the estimated relationships, and the dashed lines are 95% credible intervals of the estimate.

Despite the observed post-fire recovery at our study sites, modelled species richness and diversity averaged across the whole landscape was reduced substantially by fires occurring relatively infrequently. For example, a mean fire interval of 100 years reduced landscape-scale richness by 12% and landscape-scale diversity by 9% below the maximum attainable values, while a mean fire return interval of 50 years reduced richness by 23% and diversity by 17%. Moreover, expected richness and diversity continued to increase even as the mean fire interval increased beyond 100 years (Fig. 3).

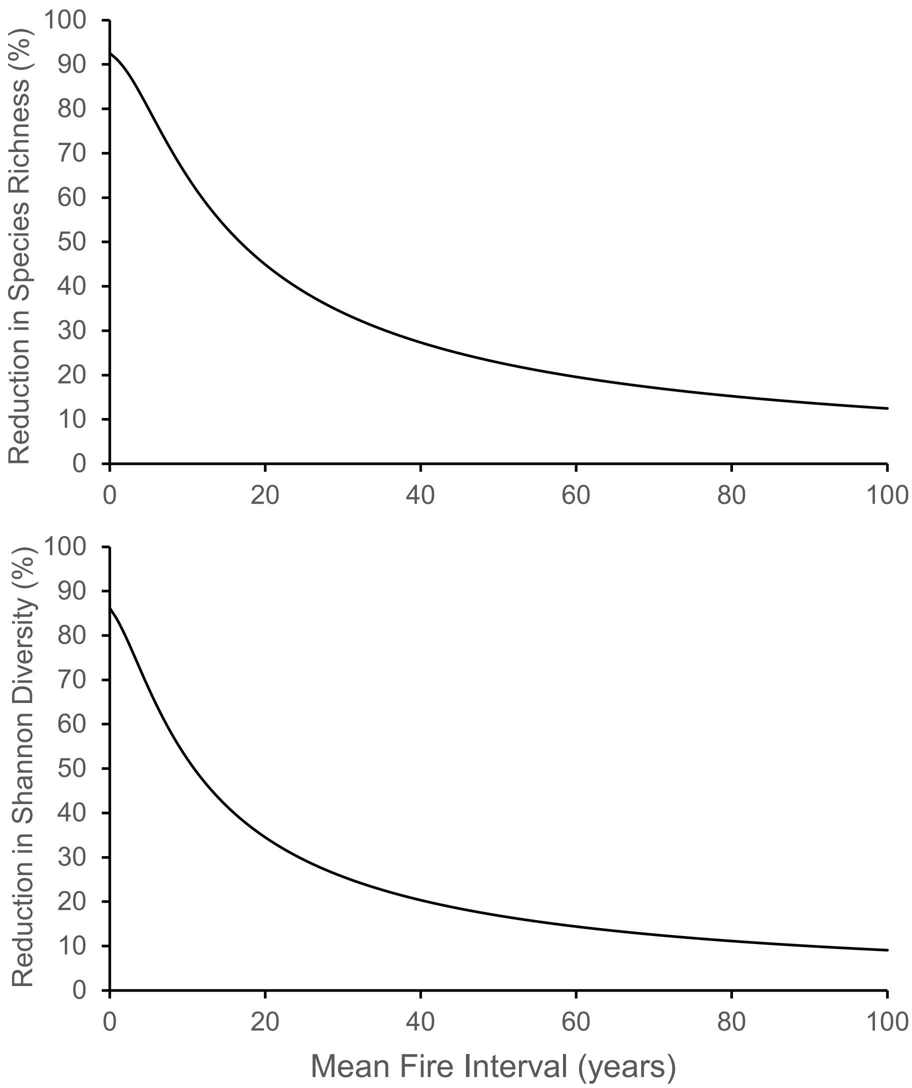

Fig 3. Reduction in species richness and Shannon diversity averaged across a landscape subject to different mean fire intervals.

Two patterns of individual species recovery were evident; some species became very numerous immediately after fire and then decreased over time (‘decreasers’ – early-successional species) while others were absent or rare immediately after fire and then increased over time (‘increasers’, Fig. 4). A third group of species did not have a clear patten of recovery which was possibly due to low sample number (Supplementary, Fig S1).

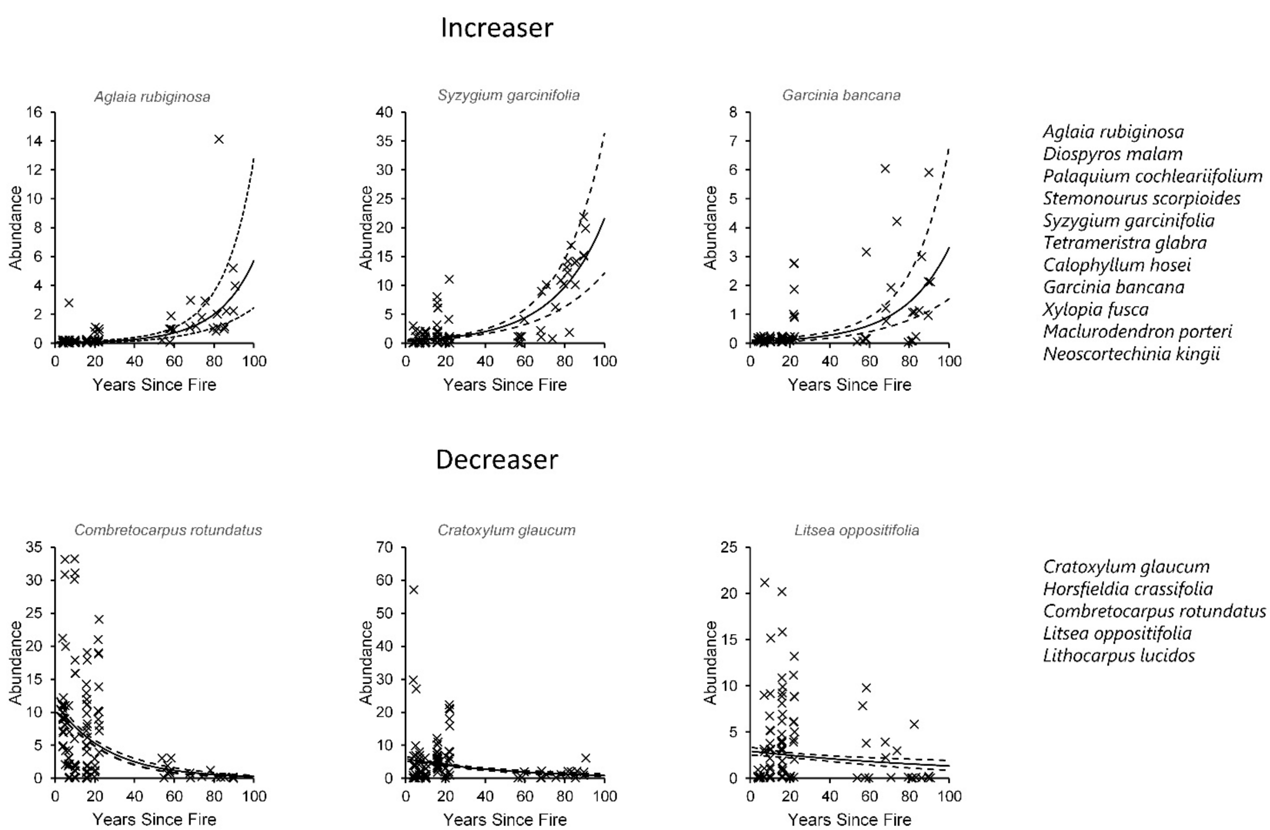

Fig 4. Examples of species' responses to time since fire. The datapoints have been jittered to reveal instances where multiple datapoints occupy the same values.

Discussion

In this study we assessed tree species richness and diversity of tropical peat swamp forests recovering naturally from fires. A key result of this study that is perhaps counter-intuitive, is that even though recovery was observed within 20-30 years at the site level, at a landscape scale, fires returning at that interval would reduce biodiversity substantially – and even fires occurring at much longer intervals (50-80 years) are predicted to result in substantial decline of biodiversity. This is an alarming result, giving that average fire return interval in peat swamp forests of Kalimantan and Sumatra is between 28 and 45 years (Vetrita and Cochrane 2020). Changing climate, land use change, mass relocation of people from densely populated Java to Kalimantan and associated with it clearance of land, increase the chances of fires becoming more extensive, severe, and frequent.

The current United Nations (UN) decade on ecosystem restoration 'aims to prevent, halt and reverse the degradation of ecosystems on every continent' (UN 2022a). Recently burnt forests were dominated by early-successional species, 'decreasers', *Combretocarpus rotundatus and Cratoxylum glaucum,* which are well known wind dispersed species (Blackham *et al.* 2014). Invasion of burnt areas is favoured by their small, wind-dispersed seeds which are produced year-round (Blackham *et al.* 2013). These two species are recommended for peatland rehabilitation (Harrison *et al.* 2011; Istomo and Valentino 2012; Shiodera *et al.* 2012; Wibisono and Dohong 2017) (Table S2). However, we believe that these species do not require artificial replanting as they easily resprout and dominate forest stands 4-5 years after fires (Fig. 4).

Moreover, stimulating regeneration of these species could lead to a 'pioneer desert' (Martínez-Garza and Howe 2003) dominated by species that will not attract the larger frugivores needed for promoting diversity.

A restoration initiative "seeding 100 Million trees to restore Borneo's rainforests" recruits new farmers to participate in the restoration of degraded forests by planting fast-growing tree species (UN 2022b). The species recommended for this project were unavailable, but we suggest that species from the "increaser" group should be the focus of revegetation efforts as these species are most at risk of decline and disappearance if fires become more frequent and more extensive. The increasers such as *Syzygium garcinifolia* and *Stemonourus scorpioides* were not present at recently burnt sites, yet these species have important values for wildlife, the latter providing food and nest sites for orangutans, *Pongo pygmaeus* (Gibson 2005; Cheyne 2007; Fauzy *et al.* 2017; Atmoko *et al.* 2018). *Syzygium* species of family Myrtaceae are generally dispersed by medium size and larger birds and mammals (Blackham *et al.* 2014), some of which are not found in recently burnt and degraded forests. Several species recorded in our study were on the International Union for Conservation of Nature (IUCN) red list as 'near threatened' or 'vulnerable' with abundances declining (Centre 1998; Ly *et al.* 2017; Olander and Wilkie 2019). Moreover, information on the conservation status of most of the species documented in our study sites was out of date or not available (Table S1), pointing to the urgent need for addressing this important knowledge gap.

The lack of information on interactions and interdependence between the two groups of plants identified in our study - 'decreases' possibly play an important role for succession of peat swamp forests regeneration. For instance, 'decreasers' may facilitate the establishment of late-successional species, by providing shade and altering soil properties (Chazdon 2014). Both groups of plants are important during revegetation. Given the potential interaction between increasers and decreasers and the need to establish the conditions for the increaser group, we suggest that a few of the early-successional species ('decreaser' group) should be incorporated into peat swamp forest revegetation processes: *Litsea oppositifolia* was only present as a small tree with diameter at breast height <10 cm even in long unburnt forests (20 years since fire), yet this species provides food for the proboscis monkey, *Nasalis larvatus* (Mukhlisi *et al.* 2018). The age at which this species bear fruits is unavailable in the literature, yet medium sized trees or larger are necessary to provide both food and habitat structure. With a mean annual increment of 1 cm per year (Yahya *et al.* 2004), it would require at least 20 years before this species can provide habitat for mammals. *Horsfieldia crassifolia*, a critically endangered decreaser (Government 2022), was only observed in sites more than 7-10 years after fires. This species has orange-coloured fruits and is a food source for mammals and birds such as hornbills (Bucerotidae). Revegetating recently burnt peat swamp forests with these 'decreasers' will enhance biodiversity, potentially creating conditions for the 'increasers' and protect these species from extinction.

The unburnt peat swamp forests within and near the Sebangau National Park were the baseline (reference) for our study, yet even these forests have been disturbed by humans in the last 50 years. Although the impact of selective logging on species richness and diversity are smaller than fires (Tata and Pradjadinata 2013), selective logging makes peat swamp forests more susceptible to fires (Harrison *et al.* 2021), resulting in synergistic negative effects. A history of past selective logging in these forests has possibly caused near loss of important commercial tree species (Astiani 2016), such as *Gonystylus bancanus* (Hadisuparto 1996; van Nieuwstadt and Sheil 2005), which then leads to low abundance of the species with forest recovery. In our study *G. bancanus* was present in low abundance in long unburnt and reference forests. It is believed that limited distance of seed dispersal due to large and heavy fruits might be one of the reasons why

G. bancanus tends to grow near mother trees (Hamzah *et al.* 2010). It is highly likely that removal of large mother trees during the era of selective logging removed the seed source for this species leading to its current low abundance in regenerating forests.

We acknowledge several limitations in our study. Firstly, analyses were based on a chronosequence approach, not repeated measures, which is difficult to achieve in practice across a large landscape for up to 22 years. Peat swamp forests are spatially variable due to both position on the peat dome and prior disturbance history. We acknowledge that chronosequence studies have limitations when applied to studying successional trajectories in species-rich and highly disturbed communities because of often major difficulties in determining temporal linkages between stages (Walker *et al.* 2010). However, chronosequences usefully clarify ecological processes in a manner that cannot be achieved in any other way – and our study sites were of similar biotic and abiotic conditions and classified as peat swamp forests (Foster and Tilman 2000). While measurements were taken at different times, the species identification was completed by the same team. The size of sampling plots varied slightly between measurements, but the sampling approach was consistent (Table S1). Our modelling assumed that fires occur randomly with respect to time (independent of time since fire). This is our best approximation of climate change, continued land use change (Sari *et al.* 2022), and at this stage there are no models that can predict peat fire occurrence. Our personal experience supports the random occurrence of fire – a peat fire in 2019 near Tumbang Nusa research camp started randomly and smouldered for months (Volkova *et al.* 2021). A single fire event may eliminate some species, and repeated fires at the same location will further reduce species diversity by altering habitat structure and eliminating intolerant species.

Tree species encountered in this study are virtually non-studied and there is very little or non-existent information available about their habitat preference, resprouting characteristics, flowering time, seed dispersal mechanisms, threatened status and other biophysical characteristics. Therefore, it is extremely difficult to provide sound practical advice on planting strategies; targeted research is required to improve our understanding of plant physiology of key species we recommend for replanting.

Conclusion

Our study in naturally regenerating tropical peat swamp forests after fires signals the need to manage for the persistence of key tree species by excluding fire. Although tree species richness and diversity recover after 20-30 years at a site level, our modelling shows that across landscapes fires occurring much less frequently (e.g., every 50-80 years) are expected to substantially reduce tree species diversity and richness. Protecting peat swamp forests from fire will be increasingly important as risks associated with urbanisation, agriculture development and climate change increase. Nevertheless, future fire is likely, and when fires occur, active planting may assist forest recovery. Further research on revegetation strategies with species proposed in this study is of critical importance for biodiversity preservation, maintaining habitat for endangered animals, and for the livelihood and wellbeing of indigenous communities relying on the ecosystem services peat swamp forests provide.

Funding

This work was supported by the Australian Centre for International Agricultural Research; the Department of Industry, Science, Energy and Resources (International Climate Change and Energy Innovation Division), grant number SLAM/2020/140, and the Asia-Pacific Network for Sustainable Forest Management and Rehabilitation (APFNet) through the project “Improving capacities towards reducing greenhouse gas emissions from peat swamp forest fires in Indonesia” (Project ID: 2018P5-IND).

CRediT authorship contribution statement

LV - Conceptualization, Funding acquisition, Investigation, Methodology, Project administration, Data curation, Validation, Formal analysis, Visualization, Writing – original draft, Writing – review & editing

HK - Conceptualization, Funding acquisition, Investigation, Methodology, Project administration, Data curation, Validation, Formal analysis, Visualization, Writing – review & editing

MAQ - Data curation, Validation, Formal analysis, Visualization, Writing – review & editing

WCA - Data curation, Validation, Formal analysis, Visualization, Writing – review & editing

RI - Data curation, Formal analysis, Visualization, Writing – review & editing

FJH - Data curation, Formal analysis, Writing – review & editing

JD – Methodology, Data curation, Validation, Formal analysis, Visualization, Writing – review & editing

MAM – Methodology, Data curation, Validation, Formal analysis, Visualization, Writing – review & editing

CJW - Conceptualization, Funding acquisition, Investigation, Methodology, Project administration, Data curation, Validation, Formal analysis, Visualization, Writing – review & editing

Declaration of the completing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgment

Authors acknowledge help of Banjarbaru and Tumbang Nusa personnel: Wawan Halwany, Purwanto B. Santosa, Supriyadi, Arif Susianto, Ariel, Yusnan, our brilliant field crew Agung Pujinegoro, Ariyanto, Hendra,Wahyu Saputra, Junaidi, Leo Saputra, Ipan Sunandar. The Sebangau NP people: Suyoko, Tatang Suwardi, Faulan Rahman, Sugianur, Madi, Mahriadi, Fauzianor.

L Volkova and CJ Weston acknowledge the foreign research permit from the Ministry of Research, Technology and Higher Education Republic of Indonesia (64/E5/E5.4/SIP/2020).

Code used in the analysis can be found in the Supplementary section.

References

Anda M., Ritung S., Suryani E., Sukarman, Hikmat M., Yatno E., Mulyani A., Subandiono R. E., Suratman, and Husnain. (2021). Revisiting tropical peatlands in Indonesia: Semi-detailed mapping, extent and depth distribution assessment. *Geoderma,* **402**, 115235.

Astiani D. (2016). Tropical peatland tree-species diversity altered by forest degradation. *Biodiversitas,* **17**(1), 102-109.

Atmoko T., Rifqi M. A., Mukhlisi, Muslim T., Purnomo, and Ma'ruf A. (2018). Wisata alam Wehea-Kelay. Bogor: FORDA Press.

Barber C. V., and Schweithelm J. (2000). Trial by fires: forest fires and forestry policy in Indonesia's era of crisis and reform. Washington: World Resources Institute.

Blackham G. V., Thomas A., Webb E. L., and Corlett R. T. (2013). Seed rain into a degraded tropical peatland in Central Kalimantan, Indonesia. *Biological Conservation,* **167**, 215-223.

Blackham G. V., Webb E. L., and Corlett R. T. (2014). Natural regeneration in a degraded tropical peatland, Central Kalimantan, Indonesia: Implications for forest restoration. *Forest Ecology and Management,* **324**, 8-15.

Centre W. C. M. (1998). Horsfieldia crassifolia. The IUCN Red List of Threatened Species 1998: e.T34541A9874777. https://dx.doi.org/10.2305/IUCN.UK.1998.RLTS.T34541A9874777.en. Downloaded on 31 May 2021.

Chazdon R. L. (2014). Second growth : the promise of tropical forest regeneration in an age of deforestation / Robin L. Chazdon: The University of Chicago Press.

Cheyne S. M. (2007). Asian apes - protecting the last great refuge of gibbons (*Hylobates albibarbis*) and orang-utans (*Pongo pygmaeus* Wurmbii): Final Report (No. Final report orangutans and gibbons compared: implications of differences in feeding and behavioural ecology for conservation). Oxford.

Cochrane M. A. (2002). Spreading like wildfire :tropical forest fires in Latin America and the Caribbean : prevention, assessment and early warning (Vol. 1). Mexico :: UNEP.

Fauzy F., Penyang, and Hidayat N. (2017). Identifikasi jenis pohon sarang dan pakan orangutan (*Pongo pygmaeus*) di Arboretum Nyaru Menteng, Palangka Raya. *Jurnal Huta Tropika,* **12**(2), 51-60.

Foster B. L., and Tilman D. (2000). Dynamic and static views of succession: Testing the descriptive power of the chronosequence approach. *Plant Ecology,* **146**(1), 1-10.

Fuller D. O., Jessup T. C., and Salim A. (2004). Loss of forest cover in Kalimantan, Indonesia since the 1997-1998 El niño. *Conservation Biology,* **18**(1), 249-254.

Gibson A. (2005). *Orangutan nesting preferences in a disturbed tropical deep-peat swamp forest, Central Kalimantan, Indonesia.* Unpublished Masters thesis, University of East Anglia, Norwich.

Government S. (2022). Floura and Fauna Web, National Parks, https://www.nparks.gov.sg/florafaunaweb/flora/2/9/2963, accessed 21 March 2022.

Hadisuparto H. (1996). The effect of timber harvesting and forest conversion on peat swamp forest dynamics and environment in West Kalimantan. In D. S. Edwards, W. E. Booth & S. C. Choy (Eds.), *Tropical Rainforest Research - Current Issues*. Dordrecht: Kluwer Academic Publishers.

Hamzah K. A., Parlan I., Sulaiman A., Faidi M. A., Yong H., and Kamarazaman I. S. (2010). *Gonystylus bancanus* jewel of the peat swamp forest. Malaysia: Forest Research Institute Malaysia.

Harrison M. E., Kursani, Santiano, Hendri, Purwanto A., and Husson S. J. (2011). Baseline flora assessment and preliminary monitoring protcol in the Katingan Peat Swamp, Central Kalimantan, Indonesia (No. Report produced by the Orangutan Tropical Peatland Project for PT. Rimba Makmur Utama/ PT. Starling Asia). Palangkaraya.

Harrison M. E., Nasir D., Healy W., Ici P., Kulu I. P., Husson S. J., Santiano, Purwanto A., Iwan, Page S. E., van Veen F., and Imron M. A. (2021). *The importance of monitoring research in assessing impacts* *of anthropogenic activities on tropical peatland biodiversity: examples from Central Kalimantan, Indonesia. Abstract Book: Oral Presentations. Proceedings of the 16th International Peatland Congress*, Tallinn, Estonia.

Hirano T., Segah H., Harada T., Limin S., June T., Hirata R., and Osaki M. (2007). Carbon dioxide balance of a tropical peat swamp forest in Kalimantan, Indonesia. *Global Change Biology,* **13**, 412-425.

Hoscilo A., Page S. E., Tansey K. J., and Rieley J. O. (2011). Effect of repeated fires on land-cover change on peatland in southern Central Kalimantan, Indonesia from 1973 to 2005. *International Journal of Wildland Fire,* **20**, 578-588.

Husson S. J., Limin S. H., Adul, Boyd N. S., Brousseau J. J., Collier S., Cheyne S. M., D'Arcy L. J., Dow R. A., Dowds N. W., Dragiewicz M. L., Smith D. A. E., Iwan, Hendri, Houlihan P. R., Jeffers K. A., Jarrett B. J. M., Kulu I. P., Morrogh-Bernard H. C., Page S. E., Perlett E. D., Purwanto A., Capilla B. R., Salahuddin, Santiano, Schreven S. J. J., Struebig M. J., Thornton S. A., Tremlett C., Yeen Z., and Harrison M. E. (2018). Biodiversity of the Sebangau tropical peat swamp forest, Indonesian Borneo. *Mires and Peat,* **22**, 1-50.

Istomo, and Valentino N. (2012). Pengaruh perlakuan kombinasi media terhadap pertumbuhan anakan tumih (*Combretocarpus rotundatus* (Miq.) Danser). *Jurnal Silvikultur Tropika,* **3**(2), 81-84.

Ly V., Nanthavong K., Pooma R., Luu H. T., Khou E., and Newman M. F. (2017). Cotylelobium lanceolatum. The IUCN Red List of Threatened Species 2017: e.T33069A2832191. https://dx.doi.org/10.2305/IUCN.UK.2017-3.RLTS.T33069A2832191.en. Downloaded on 01 June 2021.

Martínez-Garza C., and Howe H. F. (2003). Restoring tropical diversity: Beating the time tax on species loss. [Review]. *Journal of Applied Ecology,* **40**(3), 423-429.

McCarthy M. A., Gill A. M., and Bradstock R. A. (2001). Theoretical fire-interval distributions. *International Journal of Wildland Fire,* **10**(1), 73-77.

Morrogh-Bernard H., Husson S., Page S. E., and Rieley J. O. (2003). Population status of the Bornean orang-utan (*Pongo pygmaeus*) in the Sebangau peat swamp forest, Central Kalimantan, Indonesia. *Biological Conservation,* **110**, 141-152.

Mukhlisi, Atmoko T., and Priyono. (2018). Flora di habitat bekantan lahan basah Suwi, Kalimantan Timur. Bogor: FORDA Press.

Olander S. B., and Wilkie P. (2019). Madhuca crassipes. The IUCN Red List of Threatened Species 2019: e.T136448234A136470726. https://dx.doi.org/10.2305/IUCN.UK.2019-3.RLTS.T136448234A136470726.en. Downloaded on 31 May 2021.:

Page S., Hoscilo A., Langner A., Tansey K., Siegert F., Limin S., and Rieley J. O. (2009). Tropical peatland fires in Southeast Asia. In M. A. Cochrane (Ed.), *Tropical fire ecology. Climate Change, Land Use, and Ecosystem Dynamics* (pp. 263-287). Chichester, UK: Springer and Praxis Publishing.

Page S. E., Rieley J. O., Shotyk Ø. W., and Weiss D. (1999). Interdependence of peat and vegetation in a tropical peat swamp forest. *Philosophical Transactions B,* **354**, 1885-1897.

Page S. E., Rieley J. O., and Wüst R. (2006). Lowland tropical peatlands of Southeast Asia. In I. P. Martini, A. Martìnez Cortizas & W. Chesworth (Eds.), *Peatlands: evolution and records of environmental and climate changes* (pp. 145-172). Amsterdam: Elsevier.

Page S. E., Siegert F., Rieley J. O., Boehm H. D. V., Jaya A., and Limin S. (2002). The amount of carbon released from peat and forest fires in Indonesia during 1997. *Nature,* **420**(6911), 61-65.

Posa M. R. C., Wijedasa L. S., and Corlett R. T. (2011). Biodiversity and Conservation of Tropical Peat Swamp Forests. *Bioscience,* **61**(1), 49-57.

Rieley J., Notohadiprawiro T., Setiadi B., and Limin S. (2008). Restoration of tropical peatland in Indonesia: Why, where and how? . In H. Wosten, J. Rieley & S. Page (Eds.), *Restoration of tropical peatlands* (pp. 20-28): Restorpeat and International Peat Society.

Sari I. L., Weston C. J., Newnham G. J., and Volkova L. (2022). Developing Multi-Source Indices to Discriminate between Native Tropical Forests, Oil Palm and Rubber Plantations in Indonesia. *Remote Sensing,* **14**(1).

Shiodera S., Atikah T. D., Rahajoe J. S., Apandi I., Seino T., Haraguchi A., Afentina, and Kohyama T. (2012). Impact of peat-fire disturbance to forest structure in tropical peat forest. *The Boreal Forest Society,* **60**, 59-62.

Tata M. H. L., and Pradjadinata S. (2013). Regenerasi alami hutan rawa gambut terbakar dan lahan gambut terbakar di Tumbang Nusa, Kalimantan Tengah dan implikasinya terhadap konservasi. *Jurnal Penelitian Hutan dan Konservasi Alam,* **10**(3), 327-342.

Theil H. (1972). Statistical Decomposition Analysis. Amsterdam: North-Holland Publishing Company.

Turetsky M. R., Benscoter B., Page S., Rein G., van der Werf G. R., and Watts A. (2015). Global vulnerability of peatlands to fire and carbon loss. *Nature Geoscience,* **8**(1), 11-14.

UN. (2022a). Preventing, Halting And Reversing The Degradation of Ecosystems Worldwide, https://www.decadeonrestoration.org/, accessed 21 March 2022.

UN. (2022b). Seeding 100M trees to restore Borneo's rainforests; United Nations Decade on Ecosystem Restoration 2021-2030; https://implementers.decadeonrestoration.org/implementers/4/100-million-trees-for-borneo. Retrieved 28/01/2022

van Nieuwstadt M. G. L., and Sheil D. (2005). Drought, fire and tree survival in a Borneo rain forest, East Kalimantan, Indonesia. *Journal of Ecology,* **93**, 191-201.

Vetrita Y., and Cochrane M. A. (2020). Fire Frequency and Related Land-Use and Land-Cover Changes in Indonesia's Peatlands. *Remote Sensing,* **12**(1).

Volkova L., Adinugroho W. C., Krisnawati H., Imanuddin R., and Weston C. J. (2021). Loss and Recovery of Carbon in Repeatedly Burned Degraded Peatlands of Kalimantan, Indonesia. *Fire,* **4**(4).

Walker L. R., Wardle D. A., Bardgett R. D., and Clarkson B. D. (2010). The use of chronosequences in studies of ecological succession and soil development. *Journal of Ecology,* **98**(4), 725-736.

Wibisono I. T. C., and Dohong A. (2017). Panduan teknis revegetasi lahan gambut. Jakarta: Badan Restorasi Gambut.

Wijedasa L. S., Vernimmen R., Page S. E., Mulyadi D., Bahri S., Randi A., Evans T. A., Lasmito, Priatna D., Jensen R. M., and Hooijer A. (2020). Distance to forest, mammal and bird dispersal drive natural regeneration on degraded tropical peatland. *Forest Ecology and Management,* **461**, 117868.

World Bank. (2016). The cost of fire: an economic analysis of Indonesia's 2015 fire crisis. Jakarta: The World Bank.

Yahya A. Z., van Gardingen P. R., and Grace J. (2004). Diameter growth of naturally regenerated Dryobalanops aromatica in Peninsular Malaysia. *Journal of Tropical Forest Science,* **16**(1), 1-8.

Yeager C. P., Marshall A. J., Stickler C. M., and Chapman C. A. (2003). Effects of fires on peat swamp and lowland dipterocarp forests in Kalimantan, Indonesia. *Tropical Biodiversity,* **8**(1), 121-138.

Yule C. M. (2010). Loss of biodiversity and ecosystem functioning in Indo-Malayan peat swamp forests. *Biodiversity and Conservation,* **19**, 393-409.