Degradation of Southeast Asian tropical peatlands and integrated strategies for their better management and restoration

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REVIEW

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Degradation of Southeast Asian tropical peatlands and integrated strategies for their better management and restoration

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Abstract

1. About half of the world's tropical peatlands occur in Southeast (SE) Asia, where they serve as a major carbon (C) sink. Nearly 80% of natural peatlands in this region have been deforested and drained, with the majority under plantations and agriculture. This conversion increases peat oxidation which contributes to rapid C loss to the atmosphere as greenhouse gas emissions and increases their vulnerability to fires which generate regional smoke haze that has severe impacts on human health. Attempts at restoring these systems to mitigate environmental problems have had limited success.
2. We review the current understanding of intact and degraded peatlands in SE Asia to help develop a way forward in restoring these ecosystems. As such, we critically examine them in terms of their biodiversity, C storage, hydrology and nutrients, paying attention to both above-ground and below-ground subsystems.
3. We then propose an approach for better management and restoration of degraded peatlands that involves explicit consideration of multiple interacting ecological factors and the involvement of local communities who rely on converted peatlands for their livelihood.
4. We make the case that as processes leading to peatland development involve modification of both above-ground and below-ground subsystems, an integrated approach that explicitly recognizes both subsystems and their interactions is key to successful tropical peatland management and restoration.
5. *Synthesis and applications*. Gaining a better understanding of not just carbon stores and their changes during peat degradation, but also an in-depth understanding of the biota, nutrient dynamics, hydrology and biotic and abiotic feedbacks, is key to developing better solutions for the management and restoration of peatlands in Southeast Asia. Through the application of science- and nature-based solutions

that recognize the interactions among the above-ground and below-ground subsystems, and taking into account the livelihood needs of local people, we propose a way to mitigate the ongoing environmental damage that is occurring in these iconic and unique ecosystems.

KEYWORDS

above-ground–below-ground, carbon storage, climate change, greenhouse gas emissions, nature-based solutions, restoration, revegetation, rewetting

1 | INTRODUCTION

Peatlands cover an estimated 500,000 km^2 across the tropics with a C pool of at least 100 Pg (Dargie et al., 2017; Page et al., 2011). About 35%–50% of these peatlands occur in SE Asia (including New Guinea) where they have formed on coastal and sub-coastal lowlands under forest vegetation (Dargie et al., 2017; Gumbricht et al., 2017; Page et al., 2011). In the past, these peatlands supported the livelihoods of small numbers of indigenous people through fisheries, hunting and the provision of non-timber forest products (Page et al., 2011). In recent decades, however, peat swamp forests have been extensively logged (Langner et al., 2007) and drained for monoculture plantation development of tree crops such as acacia *Acacia* spp. and oil palm *Elaeis guineensis*, as well as for agriculture, at both industrial and smaller scales (Miettinen et al., 2012). As a result, peatland forest cover in SE Asia has declined from 76% to 29% (11.9 to 4.6 Mha) between 1990 and 2015 (Miettinen et al., 2016).

Conversion to plantations or other forms of agriculture results in several long-term changes in the peatland ecosystem. Removal of the original forest cover combined with drainage results in high rates of peat decomposition and carbon dioxide (CO_2) emissions such that the ecosystem switches from a net C sink to a source (Miettinen et al., 2017). Drainage also increases susceptibility to fire which results in additional peat loss and greenhouse gas (GHG) emissions, and is accompanied by extreme episodes of smoke pollution (haze), which seriously impact human health (Marlier et al., 2013; Page et al., 2002). The 2015 fire season alone is estimated to have caused persistent exposure of 69 million people to unhealthy air quality conditions and nearly 12,000 premature deaths (Crippa et al., 2016). Emissions from biological oxidation of drained peat as CO_2 are estimated to be 132–159 Mt/C year with an additional 122 Mt C/year from peat fires (Miettinen et al., 2017), with additional losses of C as methane (CH_4) and dissolved organic carbon (DOC) from drainage channels (Cook et al., 2018; Manning et al., 2019; Moore et al., 2013). Furthermore, peatland subsidence following agricultural conversion and resultant peat oxidation typically occurs at rates of 3–4 cm/year (Evans et al., 2019; Hooijer et al., 2012), which over time will lead to flooding risks as the land surface reaches the gravitational drainage limit.

In addition to causing the loss of C, clearing and degradation of peat swamp forests have a negative impact on regional biodiversity. By 2015, only 6.4% of peat swamp forests in peninsular Malaysia, Sumatra and Borneo remained in a pristine condition (Miettinen et al., 2016), placing several obligate peat forest species under threat of extinction (Giesen et al., 2018; Thornton et al., 2018). In addition, the habitat quality of remaining areas of forested peatland is being reduced by habitat fragmentation, drainage impacts, fire from surrounding landscape and unsustainable (and often illegal) timber extraction (Miettinen et al., 2016).

In peatlands that have been converted to commercial plantations, current management practices involve maintaining sufficiently low water table depths to enable optimal growth of tree crops while also reducing water-table draw down during dry seasons (Couwenberg & Hooijer, 2013). Water-tables fluctuate according to variation in rainfall, and the magnitude of these fluctuations will be greatest at sites with the lowest water-table depths (Couwenberg & Hooijer, 2013). This leads to frequent drying-wetting cycles that may increase peat loss and subsidence (Hooijer et al., 2012), and increase the risk of fire, during drier months. In addition to peatland under some form of land cover, there is an extensive area (18%–25% of the total, Miettinen et al., 2017; Wijedasa et al., 2018) with a land cover of shrubs and ferns that has resulted from deforestation and regular fires.

In response to extensive fires in 2015, the Indonesian government set up an agency, the BRG (Badan Restorasi Gambut; Peatland Restoration Agency), tasked with restoring degraded peatlands to prevent future fires through peatland rewetting and revegetation, and livelihood revitalization (BRG, 2016). Similar initiatives are being implemented at a range of scales by the private sector (e.g. under Ecosystem Restoration licences) and by a number of conservation and restoration projects (Puspitaloka et al., 2020). However, understanding of the effectiveness of rewetting in reducing peatland subsidence and GHG emissions remains limited (Dohong et al., 2018; Jauhiainen et al., 2008; Wilson et al., 2016). Furthermore, little is known about how to most effectively revegetate and restore degraded peatlands, in part because of limited understanding of the ecology of the dominant peatland plant species.

In this review, we discuss the ecology of peatlands in SE Asia, environmental consequences of their conversion and degradation and challenges related to their management and restoration. In doing this, we first consider both intact and degraded peatlands in terms of their resident biota, C and nutrient dynamics, hydrology, and soil organisms and processes. We then use this knowledge to discuss how to best implement a holistic approach for managing and restoring degraded peatlands that integrates both above-ground and below-ground ecological processes, and that recognizes the need for engagement of local communities. Our underlying goal is to highlight the multiple environmental problems that result from peatland degradation and to propose mitigation approaches using science- and nature-based solutions. This is relevant to enable responsible and sustainable management of peatlands that is under production, and to reverse peatland degradation through ecosystem restoration.

2 | TROPICAL PEATLAND HABITATS AND THEIR DEGRADATION

In their undisturbed intact state, tropical peatlands form a unique habitat in the SE Asian region because of their high water-tables and deep layers of organic matter (OM) which often separate plant root systems from underlying parent materials. Peat swamp forests support a high diversity of biota, some of which are endemic to this ecosystem (Giesen et al., 2018; Posa et al., 2011; Thornton et al., 2018). In pristine conditions, around 45 higher plant species (~3% of the peat swamp forest flora), mainly trees such as *Dyera polyphylla*, *Palaquium cochleariifolium* and *Shorea* spp., are restricted to SE Asian peatlands (Giesen et al., 2018). Many of the tree species that dominate in peatlands have specific adaptations to the waterlogged conditions, for example, by having buttress or stilt roots that provide improved stability, and breathing roots (pneumatophores) that protrude above the peat surface (Page et al., 2006).

Peat swamp forests host at least 80 endemic species of fish (Posa et al., 2011), many of which live in the water-filled pores and channels within the peat itself. Additionally, as most of the lowland forests throughout SE Asia have been cleared, existing peat swamp forests serve as refugia for species that have historically used a wider range of forest types (Posa et al., 2011). For instance, pristine peat swamp forests are the most important remaining habitat for the iconic orangutan (Erb et al., 2018; Wich et al., 2008) and for several endangered gibbon species (Cheyne et al., 2008; Quinten et al., 2010). There are also likely to be other animal species that are partially or fully dependent on peat swamp forest habitats; for example, very little is known about their invertebrate and below-ground biota.

Commercial exploitation of peat swamp forests intensified from the 1980s onwards, with the allocation of timber concessions (Dommain et al., 2016). Valuable timber species including ramin (*Gonystylus bancanus*) and meranti (*Shorea* spp.) that were once abundant in intact forests were logged and transported or floated out of the forests using light rail or through a system of constructed canals. The disturbance to the forest canopy from logging and to the peat from canal development has altered the forest microclimate and hydrological conditions by disrupting evapotranspiration processes and lowering groundwater levels. This has left these forests more susceptible to desiccation and potential fires (Field et al., 2009; Langner et al., 2007). Subsequently, from the 1990s, conversion of peat swamp forests to plantation agriculture by drainage and deforestation accelerated, especially on Sumatra and Kalimantan (Miettinen et al., 2016). During and for some years after the economic downturn of the late 1990s, Indonesia's forests, including peat swamp forests, were subject to high levels of illegal logging, which not only increased loss of forest cover but also the risk of fire (Miettinen et al., 2017).

The transformation of peat swamp forests into either deforested, degraded peatlands or plantation and smallholder agriculture has had significant social, economic, environmental and human health consequences in the region and beyond. These impacts involve the loss of biodiversity, including rare and endangered animal species, for example, flat-headed cat *Prionailurus planiceps*, Sunda clouded leopard *Neofelis diardi*, marbled cat *Pardofelis marmorata*, orangutans *Pongo* spp., fresh-water turtles, large flying fox *Pteropus vampyrus natuna* and several peat swamp fish species (Giam et al., 2012; Posa et al., 2011). In the worst-case scenario, in which the rate of conversion across the region was to reach that of the most rapidly deforested river basin, 77% (79 of 102 species) of the narrowly adapted (stenotopic) fish species are likely to become extinct, a figure that would more than double known extinctions of the world's freshwater fishes (Giam et al., 2012). They have also led to significant reductions in plant species populations; for example, 8 uncommon and 20 rare species that are restricted to this habitat may be under threat (Giesen et al., 2018). Several economically valuable timber species found in peat swamp forests, including ramin (*Gonystylus* sp.; Barstow & Randi, 2018), and several meranti species (e.g. *Shorea platycarpa*; Ashton, 1998) are listed by IUCN as critically endangered.

The degradation and conversion of peatlands has led to the disruption of natural hydrological functions (Evers et al., 2017), thereby increasing drought severity (Taufik et al., 2020) and fire hazards (Taufik et al., 2019), which, in turn, cause a rapid destabilization of the peatland C store (Baird et al., 2017), as we discuss below. This results in loss of sequestered C to the atmosphere as CO_2, and is accompanied by land subsidence, which increases the risk and duration of flooding, particularly for low-lying, shallow peatlands close to the coastline or rivers (Hooijer et al., 2015). Finally, upon peatland conversion to industrial or smallholder plantations and agriculture, there is an increased migration of human populations to these areas, which creates further pressures on local communities living in and near to pristine forests that rely on these forests for their livelihood (Silvius & Suryadiputra, 2005).

3 | CARBON RELEASE FROM DEGRADED SOUTHEAST ASIAN PEATLANDS

As tropical peatland is a globally important C store, large-scale conversion and degradation generates high C emissions. These are associated with reduced Net Primary Production (NPP; Basuki et al., 2019), drainage-driven peat oxidation and fluvial losses, and losses through combustion of peat by fire (Figure 1). We now discuss how these C losses occur, to better understand how conversion and degradation affect the peatland C cycle.

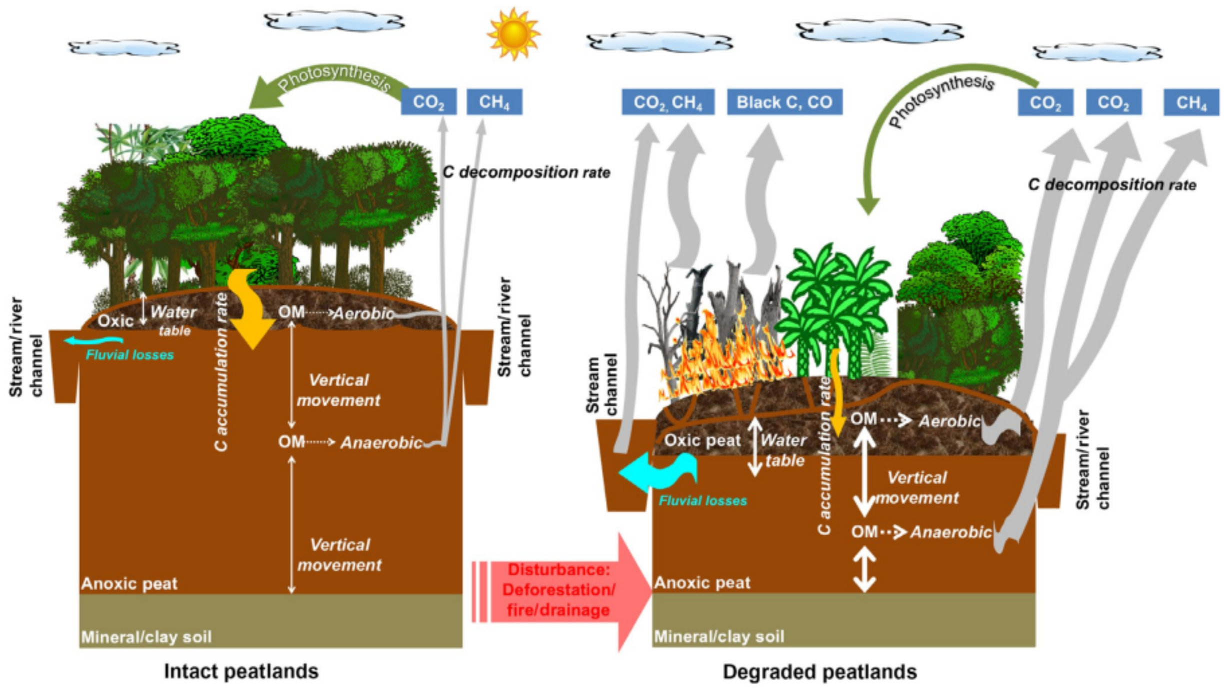

FIGURE 1 Carbon gains and losses from pristine and degraded peatlands. Thickness of arrows indicates relative magnitude of flux in pristine versus degraded peatlands. OM = organic matter; C = Carbon

3.1 | Carbon release from peatland conversion and degradation

Following peatland land-use change, oxidation of soil OM that has accumulated over millennia usually leads to a shift in the peatland C balance from a net sink to a net source of atmospheric C (Miettinen et al., 2017). This loss of C currently accounts for approximately 78% of SE Asia's total GHG emissions from managed land cover (i.e. 146 Mt/C year; Cooper et al., 2020; Miettinen et al., 2017). The balance of CO$_2$ versus CH$_4$ emissions varies considerably, and is driven by the redox potential of the peat and strongly affected by the long-term position of the water-table (Couwenberg et al., 2010). As such, water-table draw down increases greatly during peat forest degradation and peatland conversion, and has been shown to substantially increase CO$_2$ fluxes while reducing CH$_4$ emissions (Cooper et al., 2020; Jauhiainen et al., 2005; Wright et al., 2013). For example, eddy covariance measurements have revealed annual CH$_4$ exchanges over natural peat forest (i.e. 9.1 ± 0.9 gm CH$_4$ m^{-2} year^{-1}) to be twice that from plantations of *Acacia* spp. (i.e. 4.7 ± 1.5 gm CH$_4$ m^{-2} year^{-1}) on converted peatland with a low water table (Deshmukh et al., 2020). Furthermore, a clear diurnal variation in CH$_4$ exchange has been shown for natural forest (where the groundwater levels was higher) relative to *Acacia* plantations, which could be due to groundwater levels being consistently below the rooting zone in the latter (Deshmukh et al., 2020). Furthermore, in converted and degraded peatland, water saturation of dry peat upon rainfall inundation can actually support higher peat CO$_2$ release by promoting microbial respiration (Wright et al., 2013). Rainfall could also promote downward transport of labile C substrates that prime microbial communities to release C from deep peat layers, as has been demonstrated in other soil types (Fontaine et al., 2004; Kuzyakov et al., 2000).

In addition to evapotranspirative losses, water moves through tropical peatlands as subsurface flow within the peat profile and as overland flow (Baird et al., 2017; Kelly et al., 2014). As peatland deforestation and drainage result in conversion of peatland C to particulate organic C (POC) and DOC (Cook et al., 2018; Moore et al., 2011; Rixen et al., 2016), C enters the peatland hydrological system, including peatland canals and rivers. This enhanced C loss arises in a large part through organic material being metabolized by microbial activity (Cook et al., 2018; Moore et al., 2013). The fluvial losses as DOC from disturbed tropical peatlands in Indonesia have been shown to be around 50% greater than those from an adjacent intact peat swamp forest (Moore et al., 2011, 2013) and via CO$_2$ and CH$_4$ evasion from water surfaces (Manning et al., 2019), although this flux is smaller than gaseous C losses from the peat surface (Cooper et al., 2020; Hirano et al., 2012; Moore et al., 2013). In an El Ninõ year, when a lower discharge than usual is observed, TOC fluxes from oil palm plantation sites were found to be dominated (91%) by DOC and ranged from 34.4 ± 9.7 to 16.3 g C m^{-2} year^{-1}(Cook et al., 2018). In another study, levels of DOC from leaves in a degraded secondary peatland forests in a leaching experiment rose rapidly to levels far higher (up to 1,760 mg/L) than natural levels (74–84 mg/L) in the peat swamp (Yule & Gomez, 2009). More than 50% of the DOC that is lost through fluvial processes is believed to be subsequently mineralized either in rivers or after it enters the sea, where it is emitted to the atmosphere as CO_2 (Wit et al., 2015). Such losses represent an indirect contribution to GHG emissions, which are currently poorly quantified but may be an underappreciated contributor to the large total amounts of C that eventually reach the atmosphere due to peatland degradation and conversion (Cook et al., 2018).

3.2 | Carbon release following fire events

Undisturbed peat swamp forest is highly fire resistant, because it has a water-table close to the surface of the forest floor for most of the year and a humid microclimate beneath the closed canopy. Over recent decades, however, peatland fires have been substantial in terms of area burnt, severity of combustion and the scale of emissions of greenhouse and trace gases, aerosols and particulates (Huijnen et al., 2016; Page et al., 2002; Stockwell et al., 2016). Across SE Asia, fire has long been used as a cheap, fast and effective means to clear forest debris, regrowth and crop residues. But wherever anthropogenic disturbance has interfered with the natural hydrologic self-regulation of the peatland, drying of the peat surface has transformed a formerly fire-resistant ecosystem into one that can be highly flammable (Page & Hooijer, 2016; reviewed in Evers et al., 2017). In addition, increased human access and activities have escalated the number of both accidental and intentional fire ignitions (Cattaua et al., 2016).

Peatland fires are dominated by smouldering combustion (Hu et al., 2018) that persists for long time periods, and can penetrate from several centimetres to several tens of centimetres below the peat surface depending on moisture conditions (Ballhorn et al., 2009; Page et al., 2002; Simpson et al., 2016). Low oxygen availability at the fire front results in incomplete combustion, meaning that CO_2 release is accompanied by emissions of CH_4 and other toxic gaseous products, such as carbon monoxide, hydrogen cyanide and formaldehyde. Furthermore, high levels of small particulates ($PM_{2.5}$—particulate matter with diameter less than 2.5 µm) are released, which are injurious to human health (e.g. Kiely et al., 2020; Stockwell et al., 2016; Wiggins et al., 2018). This type of air pollution impacts negatively on livelihoods and economies over great distances because smoke plumes can be transported over tens or even hundreds of kilometres (Marlier et al., 2015). Some of the most severe fires in recent years (e.g. in 1997/98, 2006, 2009 and 2015) have occurred in conjunction with El Niño–Southern Oscillation (ENSO) and Indian Ocean Dipole (IOD) events (Field et al., 2009; Hong et al., 2008; Reid et al., 2013) which create drought conditions across western Indonesia and Malaysia. Ongoing peatland disturbance has, however, seen fire occurrence becoming decoupled from climate: in 2013, for example, extensive tracts of peatland in central Sumatra burnt during a short, 2-month dry spell in an otherwise wet year (Gaveau et al., 2015). Meteorological drought (low rainfall) leads to hydrological drought (decline in peatland water-table), causing soil moisture deficits and dry litter on the peat surface, which is highly combustible and serves as a fuel layer (Taufik et al., 2017). Hence, human-induced ignition can initiate wildfires even in non-drought years, and with amplification during drought years (Taufik et al., 2020), with major implications for biogeochemical cycles in peatlands through loss of nutrients bound in organic matter and thus ecosystem nutrient capital.

Fires in SE Asian peatlands are a major source of GHG emissions to the atmosphere, with average annual emissions of around 100 Tg/C year according to the Global Fire Emission Database (GFED; www.globalfiredata.org/index.html). Higher emissions occur during extreme fire years: for example, estimates for 2015 range from 227 Tg C (Huijnen et al., 2016) to 510 Tg C (Yin et al., 2016). The greatest uncertainty for estimating emissions is the depth of burn, that is, the volume of combusted peat (Ballhorn et al., 2009) with only a limited number of studies estimating fire fuel consumption (e.g. Simpson et al., 2016). Other studies have indicated significant variability in fire gaseous emissions factors between sampled fires and sites, reporting a 91% variation in CH_4 emissions from peat fires at sites in Malaysia which are likely to be due, at least in part, to differences in peat bulk density (Smith et al., 2018).

In addition to direct emissions, peat fires bring about local environmental impacts, including changes in peat chemistry and microbiology, although again, the research base is limited (but see Dhandapani, Ritz, et al., 2020). Combustion consumes surface peat, which is rich in labile C compounds, and exposes deeper peat layers that are at a more advanced stage of decomposition and contain more recalcitrant compounds (Könönen et al., 2015). The microbial abundance and phosphatase activity in burned peat soils are significantly less than that of unburned soils (Sazawa et al., 2018). In addition, burning and consequent loss of OM lowers the peat surface, bringing it closer to the groundwater level. Some studies have shown that successive fires can reduce the rate of soil respiration (Hirano et al., 2014) and thereby also the rate of peat surface subsidence (Konecny et al., 2016). However, other studies have shown that the loss of vegetation cover may increase mean surface soil temperature, thereby enhancing the rate of peat decomposition (Jauhiainen et al., 2014). In a comparison of fire and non-fire affected peatland in Brunei, no significant decrease in soil respiration at the burnt location was found, although CH_4 effluxes were enhanced in the fire-affected sites, due to a prolonged higher water table which provided more optimal conditions for methanogenesis (Lupascu et al., 2020).

4 | CURRENT STATUS OF NUTRIENT DYNAMICS IN SOUTHEAST ASIAN PEATLANDS

Tropical peat swamp forests in SE Asia generally have low nutrient availability especially when peat deposits are deep, as the surface layers become disconnected from the underlying parent material (Page et al., 2006; Wüst & Bustin, 2003). The dome-shaped topography in pristine peat swamps limits inflow of surface water and therefore nutrient input from rivers and streams is restricted to riparian zones (Gandois et al., 2014; Page et al., 2006). Nutrient inputs are low and depend primarily on rainfall, dusts and marine aerosols from outside the system (Weiss et al., 2002), and litterfall and microbial decomposition of litter inputs and organic materials within the system (Page et al., 1999; Sjögersten et al., 2011; Yule, 2010). Additionally, in peatlands of low nutrient availability, nitrogen (N)-fixing trees (e.g. *Koompassia malaccensis*) can serve as a source of N to the system (Ong et al., 2015). The plant–soil system cycles and retains nutrients near the peat surface, and studies in Malaysia (Too et al., 2018; Wüst & Bustin, 2003) and Indonesia (Lampela et al., 2014; Neuzil et al., 1993; Weiss et al., 2002) have shown nutrients to be highest in the top of the peat profile (0–80 cm) because of biological accumulation (Sulistiyanto, 2005). This decline in nutrient availability with depth, together with shifts in peat redox and substrate quality, contributes to explaining the strong depth-wise declines in microbial activity, and the capacity of the microbial community to mineralize nutrients (Jackson et al., 2009; Too et al., 2018).

In intact peat swamp forests, nutrients are tightly retained, both through being bound in relatively recalcitrant forms and through impairment of nutrient mineralization resulting from the high water table and low oxygen availability (Hoyos-Santillan et al., 2019). When peatlands are drained, logged and converted to agriculture, nutrients become more labile, leading to short-term increases in their availability upon mineralization, followed by longer-term decline in nutrient capital in disturbed, secondary and cleared forest (Dhandapani et al., 2019; Könönen et al., 2015). For example, studies in Malaysia (Dhandapani et al., 2019; Yule & Gomez, 2009) and Indonesia (Könönen et al., 2015) provide evidence that degradation, drainage and conversion of peat swamp forests can lead to decline in major nutrients such as N, P, base cations and micronutrients due to enhanced organic decomposition (Tonks et al., 2017), although some cations can also increase (Könönen et al., 2015). Furthermore, it has been shown that most macronutrients in the top 0–5 cm layer of peat were about 50% lower in the secondary forest than pristine peat swamps, because of reduced leaf litter addition, with P as a notable exception (Dhandapani et al., 2019). Conversion to agriculture also leads to changes in nutrient balances when land management involves inputs of new nutrients, such as additions of mineral (NPK) fertilizers and manure, or the planting of N-fixing trees such as *Acacia* spp. This contributes to nutrient runoff and eutrophication of nearby canals and rivers, and to gaseous losses of nitrogen as nitrous oxide (N_2O) emissions (Jauhiainen et al., 2012). In summary, clearing of peat forest and its degradation or conversion to agriculture leads to nutrient cycling shifting from a system in which nutrients are conserved to a system with greater rates of nutrient turnover and net losses.

5 | CURRENT HYDROLOGICAL MANAGEMENT PRACTICES IN TROPICAL PEATLANDS

Most major crops will not thrive in the waterlogged conditions of natural peat swamps (Andriesse, 1988). The two most common commercial peatland crops, oil palm and *Acacia* spp., both require well-drained peat to provide good yields. Furthermore, high water tables hinder fertilizer application, and interfere with access to planting areas by workers and machinery. Therefore, the strategy for plantation agriculture on peatlands in SE Asia has been to lower the water-table to improve conditions for managing and growing crops. For example, the Roundtable on Sustainable Palm Oil 2012 Best Management Practices for oil palm cultivation recommend that the water table be maintained at a depth of 40–60 cm below the peat surface (Lim et al., 2012). Water regulation is achieved by a hierarchical network of drains (Andriesse, 1988; Tay, 1969). This is essentially passive regulation, with the drainage network providing an easy route for water added by precipitation to leave the peatland.

Water management and site preparation in plantations alter the hydraulic properties of the peat. Shrinkage and compaction of surface peat reduce average pore size in the peat matrix and increase peat bulk density of surface peat in the first 1–5 years after drainage (Hooijer et al., 2012; Tonks et al., 2017). Furthermore, lower groundwater levels in drained peatlands lead to changes in the temperature regime of surface peat (Hoyt et al., 2019), and OM decomposition, like most metabolic processes, is affected strongly by temperature (Sjögerstena et al., 2018). Specifically, low groundwater levels are associated with larger fluctuations in peat surface temperature and larger diurnal fluctuations in efflux of CO_2 from the peat surface (Hirano et al., 2014; Jauhiainen et al., 2012, 2014). In addition, sites with low water table regimes are more vulnerable to drought event and will release greater amounts of CO_2, CH_4 and DOC compared to pristine peatlands, as a result of enhanced oxidative enzyme activities and microbial respiration (Kwon et al., 2013). The Soil Water Atmosphere Plant (SWAP) hydrological model (Kroes et al., 2017), which is based on two key variables (i.e. drainage resistance which is driven by the density of canals and canal water level), demonstrated that the drainage resistance of drained acacia plantation is less than half that of pristine forest (Taufik et al., 2019, 2020). Therefore, groundwater drains more easily from acacia plantations than from pristine forest, leading to lower groundwater levels and amplification of groundwater drought. The frequency of groundwater drought of transformed peatland is at least three times that of pristine peatland (Taufik et al., 2020).

In tropical peatland hydrological management, 'rewetting' involves measures to reverse the hydrologic effects of drainage for agriculture by blocking the ditches and canals to maintain higher water levels (Jaenicke et al., 2010). The effectiveness of rewetting in reducing peat decomposition has been evaluated by experiments involving either controlled mesocosms or direct hydrological interventions in the field (Jauhiainen et al., 2008), and by modelling approaches (Jaenicke et al., 2010). The outcomes of these studies are, however, inconclusive. Although rewetting can reduce the rate of peat decomposition due to water-saturation and anaerobic conditions, this might be in part offset by fermentation or anaerobic respiration (Cooper et al., 2019). Sites under high water-table regimes (Hirano et al., 2012; Jauhiainen et al., 2005, 2012, 2014) or complete water-saturation upon rewetting (Wilson et al., 2016) are associated with lower surface CO_2 efflux across a variety of land

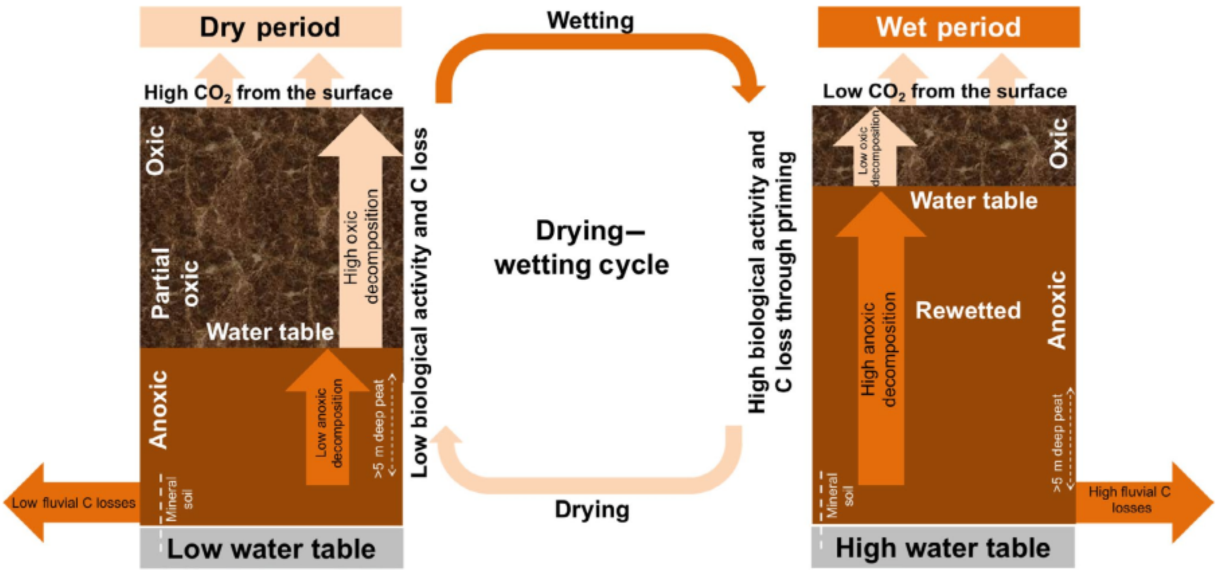

FIGURE 2 Carbon fluxes in the dry versus wet period in managed or degraded peatlands. In these peatlands, there are continual wetting–drying cycles that drive temporal variation in the water table and therefore the processes that lead to carbon loss from the peatlands

uses. Conversely, while subsidence rates (associated with peat decomposition) slow down, they do not necessarily halt completely, when water tables are at or above the peat surface (Couwenberg & Hooijer, 2013). This suggests that even with rewetting decomposition may still continue in the water-saturated zone through anaerobic respiration, with C being lost either through fluvial networks (Cook et al., 2018; Gandois et al., 2014; Moore et al., 2013) or as CH$_4$ (Deshmukh et al., 2020). Though not well studied in peat, it is likely that drying-wetting cycles can stimulate microbial activity by priming (described earlier) and therefore continued decomposition of organic C (Fontaine et al., 2004; Kuzyakov et al., 2000; Figure 2). Finally, litterfall contributes to the open pore structure of surface peat during its formation, and in plantations or degraded peatlands there are reduced litter inputs which leads to loss of this pore structure (Tonks et al., 2017); and this can hamper hydrological recovery even if peat is rewetted. In total, because of the persistent effects of drainage on its structure and chemistry, a rewetted peatland cannot be expected to behave like a natural peatland in its hydrology, soil ecology or gas exchange.

6 | BELOW-GROUND BIOTA AND PEATLAND FUNCTIONING

Accumulation of peat occurs due to production of OM from plants exceeding loss of OM through decomposition (Figure 3). The low OM decomposition rates in tropical peatlands are likely mostly due to unfavourable conditions for decomposers, including low pH, anoxia and low nutrient availability (Jackson et al., 2009; Page et al., 1999). Slow rates of decomposition are also likely driven by low quality of litter inputs through dominant plants producing leaf, root and wood litter that is high in tannins and lignin and low in nutrients (Yule, 2010; Yule & Gomez, 2009). Upon drainage and deforestation, decomposer processes are promoted by a lower water table and greater oxygen availability, which leads to more C release and thus loss of peat and greater subsidence (Furukawa et al., 2005; Hirano et al., 2012; Jauhiainen et al., 2005). However, the ecology of the below-ground biota in tropical peatlands and its role in ecosystem functioning, or in contributing to peat loss and land subsidence following peatland conversion, is poorly understood.

Decomposition in all ecosystems is carried out by prokaryotic microbes and fungi, with their activity then modulated by soil fauna. In intact peat swamp forests, the prokaryotic community is dominated by taxa characteristic of acidic, waterlogged conditions such as Proteobacteria, Acidobacteria, Verrucomicrobia and Planctomycetes (Kanokratana et al., 2011; Liu et al., 2020; Mishra et al., 2014). The highest prokaryotic diversity occurs near the peat surface (Ong et al., 2015; Too et al., 2018) and decreases sharply with depth due to anoxia and lack of fresh organic inputs; at depth, Acidobacteria and Crenarchaeota become the dominant bacterial and archaeal taxa, respectively (Jackson et al., 2009). With depth, there is also an increased abundance of anaerobic methanogenic bacteria (notably Methanobacteriaceae) and methanotrophic bacteria (notably Methylocystaceae) which are favoured by persistently saturated conditions (Finn et al., 2020). Conversion of pristine conditions to drained secondary forest has been shown to cause large changes in prokaryotic communities, for example by favouring Gram-positive bacteria over Gram-negative bacteria, which contribute to increased aerobic (vs. anaerobic) microbial activity and therefore more than doubled emissions of CO$_2$ (factor of 2.15; Dhandapani et al., 2019). The relative abundance of Gram-positive bacteria, and associated aerobic microbial activity, is further increased by a higher

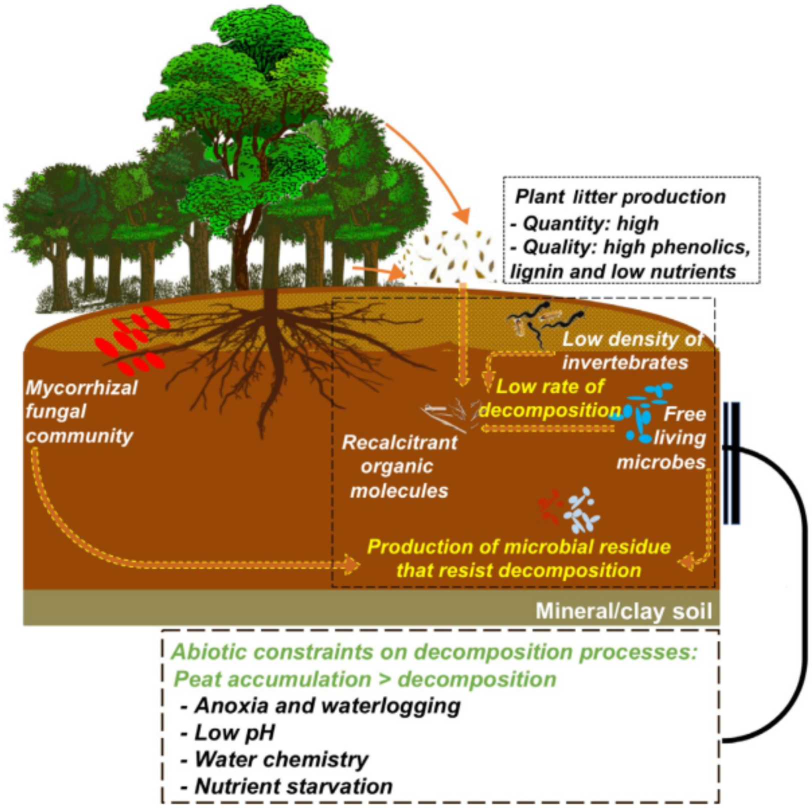

FIGURE 3 Mechanisms by which peat accumulates in intact, non-degraded peatlands. These include high production of low-quality litter, low densities of decomposer microbes and invertebrates, low soil biological activity, recalcitrant fungal residues and abiotic constraints. Several of these mechanisms may be impaired during peatland degradation which in turn prevents carbon from accumulating, but appropriate management or restoration of peatlands may reverse this impairment

intensity of peat land use, for example from diverse cropping systems to monoculture oil palm plantations (Dhandapani et al., 2020). Although we have an unclear understanding about the mechanistic basis by which changes in the structure of the peatland microbiome lead to shifts in its functioning during peat forest clearing, conversion and land-use intensification, the available evidence reveals a strong shift towards aerobic prokaryotic communities which likely contribute to rapid loss of peat.

Intact tropical peatlands support their own communities of saprophytic fungi, but very little is known about their composition or roles in ecosystem processes. A handful of studies have explored how fungal communities may be changed upon conversion and clearance of tropical peatlands. For example, studies in Malaysia have shown that the diversity within each of several fungal groups (Ascomycetes, Basidiomycetes, Chytridiomycota and Zygomycota) can be reduced through land conversion from peat swamp forests to oil palm plantation (Kusai et al., 2018; Shuhada et al., 2017). Similarly, cellulolytic and viable fungal communities were shown to be reduced upon peatland conversion from forest to paddy–soybean rotation and paddy field in Indonesia (Hadi et al., 2001). In another study in Indonesia, higher fungal species richness was found in natural peat swamps than in degraded peat soil, with Ascomycota as the most abundant phylum, followed by Basidiomycota, Zygomycota and Glomeromycota (Talukder & Sun, 2019). Furthermore, pathotrophic fungi (e.g. *Metarhizium* and *Pestalotiopsis*) can greatly increase in abundance in degraded peatlands, suggesting the potential of land conversion to increase the likelihood of plant diseases (Liu et al., 2020). A recurrent theme in these studies is a reduction of diversity of fungal communities upon conversion, but we know little about how the remaining community contributes to decomposition processes, or whether those fungi that are favoured by conversion are better capable of decomposing peat, thereby contributing to loss of peat and its subsidence.

Mycorrhizal fungi play a key role in the formation of OM, both as decomposers and through the accumulation of their necromass (Clemmensen et al., 2013), and with the necromass of ectomycorrhizal (EcM) fungi contributing more to OM formation than that of arbuscular mycorrhizal (AM) fungi (Averill et al., 2014). Southeast Asian peatlands are often dominated by trees of the *Dipterocarpaceae* which potentially support EcM and a variety of other families that all support AM. However, only three studies have explored the mycorrhizal status of trees in intact peat swamp forests in SE Asia, and they all report significant colonization of tree or seedling roots by both EcM (Moyersoen et al., 2001; Mulyani et al., 2014) and AM (Tawaraya et al., 2003). As such, Moyersoen et al. (2001) found no difference in the relative basal area abundance of EcM trees, the proportion of EcM roots in monoliths or fractional colonization of EcM roots, between peat forest and adjacent heath forest in Brunei, despite the very high water table in the peat forest. The extent to which accumulation of necromass of EcM associated with Dipterocarps contributes to the build-up of peat, in the manner shown for other EM-dominated ecosystems at higher latitudes

(Clemmensen et al., 2013), is unknown. When intact peat swamp forests undergo drainage and deforestation, mycorrhizal fungi will be lost along with their host trees, but the extent to which this may contribute to the loss of OM and subsidence of peat remains unexplored.

While the below-ground fauna mediate the decomposition processes carried out by bacteria and fungi, we know virtually nothing about the ecology or role of most of this fauna in tropical peat swamp forests. However, termites, which are major decomposers in most tropical and subtropical forests (Eggleton et al., 1996), have been investigated in SE Asian peatlands in a handful of studies. These have shown that while termite densities are often high in intact peat swamp forests, their diversity is typically low (Bourguignon et al., 2018; Neoh et al., 2016; Vaessen et al., 2011), suggesting that a limited number of taxa are adapted to an ecosystem with a persistently high water table. It has also been shown that conversion of peatlands and intensification of their management leads to termite diversity declining further, with the lowest diversity in plantations of oil palm, rubber and *Acacia* (Hidayat et al., 2018; Vaessen et al., 2011). Conversely, a study from peat swamp forests in Brunei showed that selective logging without drainage had limited impact on termite species richness, suggesting that peatland conversion is likely to alter the termite community only when it involves changes to the water table and a loss of forest cover (Bourguignon et al., 2018). Burning of degraded peatlands leads to further losses of termite species but particular genera (*Coptotermes*, *Schedorhinotermes* and *Parrhinotermes*) are adapted for surviving fire and dominate in the post-fire environment when most other termites have been lost (Neoh et al., 2016). The contribution of termites to decomposition processes remains unexplored, and it is unknown as to whether and how those taxa that do remain after peatland degradation, contribute to peat decomposition, losses and subsidence.

7 | ABOVE-GROUND AND BELOW-GROUND APPROACHES FOR IMPROVED RESTORATION AND SOCIO-ECONOMIC OUTCOMES

Protection of the remaining peatswamp forests across the SE Asian region has to be given the highest possible priority in any conservation activity, given their essential role in protecting large C stocks and hosting forest biodiversity while also mitigating GHG emissions and fire risk (Evers et al., 2017; Page et al., 2009; Vernimmen et al., 2020). Land-use planning and policies need to take into account the gamut of valuable and often irrecoverable ecosystem services that are provided by tropical peatlands in their natural state (Uda et al., 2017; Wijedasa et al., 2016), and the high costs and challenges associated with ecosystem restoration (Budiharta et al., 2018). Within private concession lands, companies are pledging to a 'No Deforestation, No Peat, No Exploitation (NDPE)' policies while also applying the High Conservation Value (HCV) and High Carbon Storage (HCS) approach to determine which areas of high C density should be excised from their concession or set aside for protection (HCSA, 2019; Padfield et al., 2016). Some companies have moved further to undertake active rehabilitation of degraded peatland and peat swamp forest, for example under Ecosystem Restoration Concession licences in Indonesia (Harrison et al., 2020). Nevertheless, extensive areas of degraded peat swamp forest and deforested peatland have no formal protection or restoration mandate despite them making a globally significant contribution to land-based greenhouse gas emissions (Cooper et al., 2020; Miettinen et al., 2017; Wijedasa et al., 2018).

7.1 | Restoration efforts that recognize above-ground and below-ground ecological principles

In the past two decades, a growing number of mostly small-scale peatland restoration initiatives have been attempted in SE Asia, notably in Indonesia (Dohong et al., 2018). In response to extensive forest and peatland fires in 2015, Indonesia aimed to restore 2 million hectares of peatlands (Republic of Indonesia, 2016). Restoration initiatives usually involve a combination of rewetting and revegetation to re-establish the water-saturated and forested conditions that more closely resemble the natural peatland environment and that can mitigate the ongoing release of C from peat oxidation and fires, as well as the loss of biodiversity (Evans et al., 2014; Wijedasa et al., 2018). They may also involve revitalization activities to promote suitable alternative livelihood options for local communities (Harrison et al., 2020). However, peatland restoration initiatives are challenging (Harrison et al., 2020; Zeng et al., 2020), and face a high risk of failure when their outcomes are poorly defined (Puspitaloka et al., 2020), socio-economic factors are not adequately considered (Zeng et al., 2020), and efforts to rewet and revegetate peatlands are uncoordinated (Giesen & Sari, 2018). Here, we discuss current solutions and the issues involved in implementing them, and propose an integrated, multidisciplinary approach to the restoration of these unique environments which focuses on combining ecological knowledge of the above-ground and below-ground peatland ecosystem, with consideration of the socio-economics of land use and management (Figure 4).

A key component of any peatland restoration activity is hydrological restoration (Figure 4), which has a primary aim of obstructing or limiting water flow to raise water levels in canals and ditches, and thereby rewet the adjacent peatland (Dohong et al., 2018). Several projects have evaluated different approaches to dam construction in terms of their cost, and life span, and susceptibility to damage by hydraulic forces, decay and vandalism (Dohong et al., 2018; Page et al., 2009; Ritzema et al., 2014). Evaluating the extent to which canal blocking is effective in raising the water table in the peat itself is difficult, because it requires assessment of how much drier the peatland would have been, had the canals not been blocked or were blocked differently, against the background of fluctuations (Reid et al., 2012) and long-term trends in rainfall (Dommain et al., 2011). The effects of canal blocking on peatland water levels have been studied using field measurements (Jauhiainen et al., 2008; Ritzema

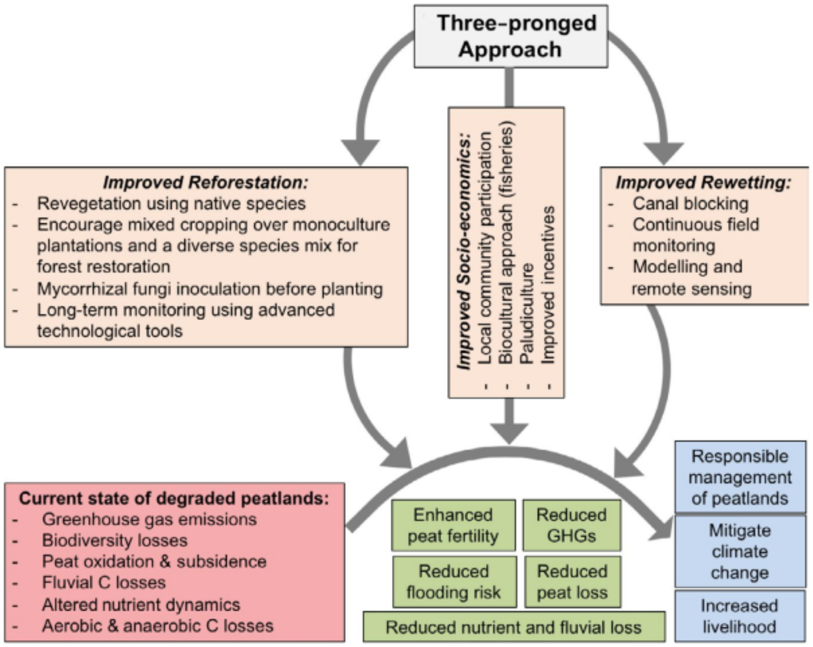

FIGURE 4 A three-pronged approach to better management and restoration of degraded peatlands which incorporates reforestation, rewetting and socio-economic consideration

et al., 2014), remote sensing (Jaenicke et al., 2011) and modelling approaches (Jaenicke et al., 2010). However, none of these methods in isolation can directly estimate the benefits of canal blocking because changes in water levels are complicated by fluctuating inputs from rainfall, and because models require data for parameterization and validation (Jaenicke et al., 2010). Further limitations stem from attempts to correlate remotely sensed soil moisture information (e.g. derived from radar sensors) with groundwater levels, because soil moisture may change rapidly in response to rainfall without reflecting a longer-term change in the peatland water table (FAO, 2020). There is, therefore, a need for improved approaches that integrate data from field monitoring, modelling and remote sensing to evaluate the benefits of rewetting (Figure 4) and to better understand seasonal fluctuations, which are a key driver of below-ground processes, including microbial functions involved in C and nutrient cycling.

For improved reforestation efforts, both above-ground and below-ground ecological principles can be exploited to meet the overall goal for effective restoration of degraded tropical peatlands (Figure 4). Restoring closed-canopy forest vegetation on degraded peatland has the potential to deliver a number of benefits: enhanced biodiversity, a supply of plant litter to the uppermost peat, stabilization of microclimate (e.g. increased humidity and lower maximum surface temperatures) and reduced fire risk due to increased moisture levels in both surface and below-ground fire fuels. In less disturbed peatlands, revegetation may rely solely on natural regeneration, but on more degraded sites enrichment planting is needed (Giesen & Sari, 2018), though this is constrained by the species-specific survivorship of planted seedlings (Lampela et al., 2017; Nuyim, 2005). A major barrier to natural regeneration is the distance from the existing forest edge, with increasing distance reducing the potential for seed dispersal by birds and mammals but also negatively influencing microclimate, hydrology, nutrient availability and fire frequency (Blackham et al., 2014; Graham et al., 2017; Wijedasa et al., 2020). Even where natural regeneration does occur near the forest edge, limited seed dispersal may warrant tree planting at greater distances, for example, beyond 500 m (Wijedasa et al., 2020). Enrichment plantings at greater distances from the forest edge pose challenges, however, because young trees are exposed to extremes of temperature, drought and flooding, and there is competition from the existing plant community which often consists of a dense cover of light-tolerant weedy ferns, sedges and shrubs.

To date, interventions to restore tree cover on degraded peatlands have been relatively small scale and have met with varied success. Frequently, the emphasis has been on initial tree planting rather than on long-term monitoring, and often with a focus on the number of trees planted rather than the number that survive (Harrison et al., 2020). But it is only through ongoing monitoring that we can gain valuable information on restoration effectiveness across a range of sites and conditions. Ideally, site monitoring data will include plant species composition and biomass, tree species survivorship and other key variables that can provide knowledge of changes in site hydrology, C and nutrient cycling over time. A major gap in understanding is how the choice of plant species for restoration impacts on peatland ecosystem functioning, and how the functional traits of different plant species used during restoration impact on the quality of above-ground and below-ground litter inputs to the peat, C and nutrient fluxes in the peat profile, and mitigation of C loss from the peat. In addition, peat swamp forest trees appear to allocate relatively large proportions of their biomass into root systems, which make an important contribution to below-ground peat accumulation (Brady, 1997; Lampela et al., 2014). Relatively little is known about differences among peat swamp tree species in root biomass allocation or rates of decomposition of their root, stem and leaf litter. More knowledge on these topics would be useful in identifying tree species for restoration programmes that have the potential to make the greatest contribution to below-ground litter inputs and hence to C accumulation under suitable hydrological conditions.

The below-ground soil biota is a key driver of ecosystem restoration because of its role in regulating the build-up of OM, and the establishment of vegetation (Kardol & Wardle, 2010). Our understanding of the role that the peatland microbiota plays in restoration efforts remains limited, but some work suggests that mixed cropping can support a high microbial diversity that is associated with reduced peat oxidation when compared with monocultures (Dhandapani, Girkin, et al., 2020; Mishra et al., 2014). This suggests that the diverse microbiota associated with more diverse plant communities may reduce peat oxidation and thus C loss and subsidence during restoration, implying that the C balance could be improved by mixed cropping a wider range of tree species within existing plantations, and encouraging a diverse mix of tree species in forest restoration (Figure 4). Furthermore, there have been a handful of studies that have shown how mycorrhizal fungi and their inocula may stimulate the growth and establishment of tree species on degraded peatland (e.g. Mulyani et al., 2014; Tawaraya et al., 2003; Yuwati & Putri, 2020). For example, inoculation of *Shorea balangeran* seedlings with EcM fungi (*Strobilomyces*, *Calvatia*, *Boletus* and *Scleroderma*) has positive effects on their early growth when planted into peatland (Turjaman et al., 2011). Furthermore, some studies have shown beneficial effects of inoculation by AM fungi (*Glomus* spp.) upon transplantation of *D. polyphylla* (Graham et al., 2013) and *Shorea balangeran* (Yuwati & Putri, 2020) into peatland. Such studies point to the importance of understanding the mycorrhizal associations of plant species being targeted for restoration, and of knowing which fungal inoculants are most likely to contribute to their success. Overall, there is a strong need for research to better understand the role of the peat microbiome and linkages between above-ground and below-ground biota in both mitigating peat loss and enabling vegetation establishment during restoration of tropical peatlands, as has been shown for other environments (Kardol & Wardle, 2010).

7.2 | Restoration efforts for livelihoods and improved socio-economic outcomes

One approach to the restoration of peatlands, especially peatlands which are in use by local communities for their livelihoods, entails cultivation at higher or high (near or at surface) water tables, that is, paludiculture, or the 'sustainable production of biomass on wet and rewetted peatlands' (Wichtmann & Joosten, 2007). The concept of paludiculture (Figure 4) is loosely defined and encompasses various aspects that are relevant for restoration, such as the type of ecosystem services provided (e.g. mitigation of greenhouse gases), the hydrological condition of the peat and the source of vegetation for cultivation (Tan, Lupascu, et al., 2020). Increasing efforts are being made to identify suitable approaches for high water table agriculture on tropical peatlands which integrate environmental and socio-economic goals, but this is a field that is much less developed for tropical than for temperate environments (Tan, Lupascu, et al., 2020). A number of timber species and non-timber commodities, such as jelutong *D. polyphylla*, sago *Metroxylon sagu*, nypa *Nypa fruticans*, illipe nut *Shorea macrophylla* and candle nut (*Aleurites moluccanus*), have been proposed as candidate paludiculture species to support livelihoods of communities during the restotation of degraded peatlands (Giesen & Sari, 2018; Widayati et al., 2016). While some of these crops (e.g. sago) are able to withstand varying degrees of peat rewetting, others that are in use in peat-livelihood projects are dryland crops (e.g. pineapple and coffee) which still require peat drainage. Current economic returns from candidate high water level crops are lower than those from conventional drainage-based crops such as oil palm *E. guineensis*; Tan, Lupascu, et al., 2020), which makes the switch for local communities to paludiculture challenging (Giesen & Sari, 2018). In addition, there has been a lack of market development for paludiculture crops beyond general statements that certain crops could have some future economic value.

Given the large numbers of people now living in or adjacent to peatland in SE Asia, it is increasingly recognized that the participation of local communities in peatland restoration needs to be embraced as an integral part of the process (Graham et al., 2017; Page et al., 2009), whether for governmental land or for concessions. This involves engagement of stakeholders at all stages of defining and addressing the peatland management 'problem', including exploration, goal setting, planning and monitoring phases, to anticipate whether environmental interventions such as restoration programmes could be culturally insensitive or negatively impact livelihoods (Stringer et al., 2006). Failure to understand the relevance and use of peatlands and the exclusion (or superficial inclusion) of local communities in peatland restoration plans can result in resistance by communities. This occurred, for example, in Central Kalimantan, where dams constructed for peatland restoration obstructed transport in canals by resident communities (Dohong, 2016; Page et al., 2009). It is therefore crucial to consider the lived experience of local communities and their interactions with peatlands, as well as the cultural and socio-economic factors that predispose local communities to aspire to restoring degraded peatlands (Puspitaloka et al., 2020; Uda et al., 2020).

Despite a growing body of work on identifying best practices for peatland hydrology and revegetation, there has been a lack of attention on the social dimensions for peatland restoration (Puspitaloka et al., 2020). However, these social dimensions encompass the definition of what peatland restoration means for different actors involved in the process of managing restoration projects, and how restoration activities are defined, executed and sustained for the long term (Uda et al., 2020; Ward et al., 2020). Adopting a biocultural approach (Figure 4) that can assist with integrating these social dimensions by placing emphasis on local values and traditions (Thornton et al., 2020). For example, fisheries are a critical resource for communities living by the edge of peat rivers, but their value extends beyond the utilitarian, with cultural and spiritual values of catching, consuming and selling fish all requiring consideration in the design and implementation of effective restoration strategies (Thornton et al., 2020).

Restoration of degraded tropical peatlands will necessarily require long time periods, given the need to restore both a closed-canopy forest and the key below-ground processes involved in C sequestration and storage (Harrison et al., 2020). These long time periods reflect the slow growth and high longevity tropical peatland tree species (King et al., 2005; Lieberman et al., 1985) and lengthy (but currently undefined) time periods may be required for the restoration of below-ground microbial processes. These extended time periods are, however, incompatible with the current pace of ecosystem degradation and, in particular, fire events which are occurring on tropical peatlands with increasing frequency (Vetrita & Cochrane, 2019). Average fire-return intervals are shorter over non-forest peat areas, which are the primary targets for replanting and reforestation (Adrianto et al., 2019). A single fire event in a peatland with unregulated drainage can result in an average burn depth of anywhere between 17 and 50 cm (Ballhorn et al., 2009; Konecny et al., 2016; Page et al., 2002), which, even assuming that suitable conditions for peat formation could be re-established, would require several centuries for the lost peat layer to re-accumulate (Harrison et al., 2020). Any restoration strategy for tropical peatlands must therefore incorporate fire prevention alongside rewetting, revegetation and the promotion of appropriate livelihood strategies. Multiple social drivers are at the root of the tropical peatland fire phenomenon (Santika et al., 2020; Sze et al., 2019; Tan, Carrasco, et al., 2020) and in order to limit fire occurrence, holistic restoration strategies will be needed to maintain and enhance any remaining natural forest cover and condition while also develop alternative non-burning methods for land management, raise public awareness and promote behavioural change (Harrison et al., 2020).

Our three-pronged approach (Figure 4) emphasizes that the restoration of degraded tropical peatland ecosystems will benefit from an integrated, multidisciplinary approach that involves contributions from researchers and practitioners with expertise in hydrology, microbial ecology, soil science, plant ecology, fire ecology and social sciences, among other topics. Developing these types of collaborative activities will require an in-depth understanding of the ecological and social challenges that need to be faced as well as the types of interventions and science required to address them. It will also involve gaining the support of local communities, with the recognition that these communities are diverse in culture, traditions and land-use history, have varied abilities to cope with peatland degradation and fires, and differ in their capacity and willingness to be involved in peatland restoration. Community willingness may be limited if the costs (both financial and societal) of adopting new land management practices remain unclear and the benefits poorly understood. As such, communities that have relied on oil palm cultivation will find it challenging to participate in peatland restoration since rewetting may entail lower productivity and financial return for their crops (Ward et al., 2020). This may require active exploration of viable economic alternatives (Tan, Lupascu, et al., 2020), including monetary or business incentives for farmers to commit to more sustainable practices (Figure 4). Most importantly, there is a pressing need to encourage research that involves and has a buy-in from multiple stakeholders and different actors (NGOs, government agencies or private sector, local people) to work towards a common goal of peatland restoration and achieve the benefits that restoration may deliver (Puspitaloka et al., 2020) in terms of increased livelihood, carbon sequestration and responsible management of peatlands (Figure 4).

8 | CONCLUSIONS AND FUTURE DIRECTIONS

We have reviewed current understanding of the intact and degraded peatlands in SE Asia, and critically examined them in terms of their biodiversity, C storage, hydrology, nutrients and the below-ground subsystem. We have then proposed an integrated approach for better management and restoration of degraded peatlands, which involves explicit consideration of multiple interacting ecological factors, and that addresses the involvement of local communities who rely on converted peatlands for their livelihood. In doing this, we emphasize that as processes leading to peatland formation involve several mechanisms that take place in both above-ground and below-ground subsystems, an integrated approach that explicitly recognizes both subsystems and the interactions between them is key for any attempt to successfully manage and restore these ecosystems.

Finally, we highlight some critical research and management needs that, if addressed, would provide much needed information on how to better restore degraded peatlands. First, we need better knowledge of the above-ground and below-ground processes that lead to C accumulation in intact peatlands so that we can understand how to reduce further peat loss in degraded peatlands and how to re-establish the appropriate living machinery (e.g. tree species composition) to maximize peat accumulation during peatland restoration. Second, hydrological and revegetation initiatives aimed at restoration have to some extent been undertaken in isolation. Integration is needed among these initiatives to deliver more robust science and nature-based solutions that take account of the linked ecological and hydrological processes that underpin restoration efforts. Third, in-depth research is required on the functional role of below-ground biota in peatlands, their role in C and nutrient cycling, and their effects on and feedbacks with the plant community. Fourth, we highlight that there is a need for all relevant stakeholders including governmental bodies and local communities to be actively engaged in the development and implementation of any attempts at improved restoration and rehabilitation of these ecosystems. And finally, adopting an integrated management and restoration approach to tropical peatlands is highly relevant to attempting to meet the targets set by the 2030

United Nations Agenda for Sustainable Development and the Paris Agreement on climate change.

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AUTHORS' CONTRIBUTIONS

S.M., D.A.W. and S.E.P. conceived the idea of the manuscript; S.M. wrote the draft of the manuscript in close consultation with D.A.W. and S.E.P. All other authors contributed input and text on topics that most closely aligned with their area of expertise and provided inputs for improving manuscript drafts.

DATA AVAILABILITY STATEMENT

This review was completed entirely using published sources, all of which are cited within the manuscript.

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REFERENCES

Adrianto, H. A., Spracklen, D. V., Arnold, S. R., Sitanggang, I. S., & Syaufina, L. (2019). Forest and land fires are mainly associated with deforestation in Riau province. *Indonesia. Remote Sensing*, *12*(1), 3. https://doi.org/10.3390/rs12010003

Andriesse, J. (1988). *Nature and management of tropical peat soils*. FAO, United Nations.

Ashton, P. (1998). Shorea platycarpa. In *The IUCN Red List of Threatened Species*. IUCN. https://doi.org/10.2305/IUCN.UK.1998.RLTS.T33135A9761348.en

Averill, C., Turner, B. L., & Finzi, A. C. (2014). Mycorrhiza-mediated competition between plants and decomposers drives soil carbon storage. *Nature*, *505*(7484), 543–545. https://doi.org/10.1038/nature12901

Baird, A. J., Low, R., Young, D., Swindles, G. T., Lopez, O. R., & Page, S. (2017). High permeability explains the vulnerability of the carbon store in drained tropical peatlands. *Geophysical Research Letters*, *44*, 1333–1339. https://doi.org/10.1002/2016GL072245

Ballhorn, U., Siegert, F., Mason, M., & Limin, S. (2009). Derivation of burn scar depths and estimation of carbon emissions with LIDAR in Indonesian peatlands. *Proceedings of the National Academy of Sciences of the United States of America*, *106*, 21213–21218. https://doi.org/10.1073/pnas.0906457106

Barstow, M., & Randi, A. (2018). *Gonystylus velutinus*. The IUCN Red List of Threatened Species: e.T38047A121997240.

Basuki, I., Kauffman, J. B., Peterson, J., Anshari, G., & Murdiyarso, D. (2019). Land cover changes reduce net primary production in tropical coastal peatlands of West Kalimantan, Indonesia. *Mitigation and Adaptation Strategies for Global Change*, *24*(4), 557–573.

Blackham, G. V., Webb, E. L., & 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.

Bourguignon, T., Dahlsjö, C. A. L., Salim, K. A., & Evans, T. A. (2018). Termite diversity and species composition in heath forests, mixed dipterocarp forests, and pristine and selectively logged tropical peat swamp forests in Brunei. *Insectes Sociaux*, *65*, 439–444.

Brady, M. A. (1997). *Organic matter dynamics of coastal peat deposits in Sumatra, Indonesia*. Thesis, University of British Columbia.

BRG. (2016). Pedoman Pelaksanaan Kerangka Pengaman Sosial dalam Restorasi Gambut, P.K.B.R.G.P.K. BRG-SB/12/2016), Editor. 2017, Badan Restorasi Gambut.

Budiharta, S., Meijaard, E., Gaveau, D. L. A., Struebig, M. J., Wilting, A., Kramer-Schadt, S., Niedballa, J., Raes, N., Maron, M., & Wilson, K. A. (2018). Restoration to offset the impacts of developments at a landscape scale reveals opportunities, challenges and tough choices. *Global Environmental Change*, *52*, 152–161. https://doi.org/10.1016/j.gloenvcha.2018.07.008

Cattau, M. E., Harrison, M. E., Shinyo, I., Tungau, S., Uriarte, M., & DeFries, R. (2016). Sources of anthropogenic fire ignitions on the peat-swamp landscape in Kalimantan. *Indonesia. Global Environmental Change*, *39*, 205–219. https://doi.org/10.1016/j.gloenvcha.2016.05.005

Cheyne, S. M., Thompson, C. J. H., Phillips, A. C., Hill, R. M. C., & Limin, S. H. (2008). Density and population estimate of gibbons (*Hylobates albibarbis*) in the Sabangau catchment, Central Kalimantan, Indonesia. *Primates*, *49*(1), 50–56. https://doi.org/10.1007/s10329-007-0063-0

Clemmensen, K. E., Bahr, A., Ovaskainen, O., Dahlberg, A., Ekblad, A., Wallander, H., Stenlid, J., Finlay, R. D., Wardle, D. A., & Lindahl, B. D. (2013). Roots and associated fungi drive long-term carbon sequestration in boreal forest. *Science*, *339*(6127), 1615–1618. https://doi.org/10.1126/science.1231923

Cook, S., Whelan, M. J., Evans, C. D., Gauci, V., Peacock, M., Garnett, M. H., Kho, L. K., Teh, Y. A., & Page, S. E. (2018). Fluvial organic carbon fluxes from oil palm plantations on tropical peatland. *Biogeosciences*, *15*, 7435–7450.

Cooper, H. V., Evers, S., Aplin, P., Crout, N., Dahalan, M. P. B., & Sjogersten, S. (2020). Greenhouse gas emissions resulting from conversion of peat swamp forest to oil palm plantation. *Nature Communications*, *11*, 407. https://doi.org/10.1038/s41467-020-14298-w

Cooper, H. V., Vane, C. H., Evers, S., Aplin, P., Girkin, N. T., & Sjögersten, S. (2019). From peat swamp forest to oil palm plantations: The stability of tropical peatland carbon. *Geoderma*, *342*, 109–117. https://doi.org/10.1016/j.geoderma.2019.02.021

Couwenberg, J., Dommain, R., & Joosten, H. (2010). Greenhouse gas fluxes from tropical peatlands in south-east Asia. *Global Change Biology*, *16*, 1715–1732. https://doi.org/10.1111/j.1365-2486.2009.02016.x

Couwenberg, J., & Hooijer, A. (2013). Towards robust subsidence-based soil carbon emission factors for peat soils in south-east Asia, with special reference to oil palm plantations. *Mires and Peat*, *12*, 1–13.

Crippa, P., Castruccio, S., Archer-Nicholls, S., Lebron, G. B., Kuwata, M., Thota, A., Sumin, S., Butt, E., Wiedinmyer, C., & Spracklen, D. V. (2016). Population exposure to hazardous air quality due to the 2015 fires in Equatorial Asia. *Scientific Reports*, *6*, 37074. https://doi.org/10.1038/srep37074

Dargie, G. C., Lewis, S. L., Lawson, I. T., Mitchard, E. T. A., Page, S. E., Bocko, Y. E., & Ifo, S. A. (2017). Age, extent and carbon storage of the central Congo Basin peatland complex. *Nature*, *542*, 86–90. https://doi.org/10.1038/nature21048

Deshmukh, C. S., Julius, D., Evans, C. D., Nardi, Susanto, A. P., Page, S. E., Gauci, V., Laurén, A., Sabiham, S., Agus, F., Asyhari, A., Kurnianto, S., Suardiwerianto, Y., & Desai, A. R. (2020). Impact of forest plantation on methane emissions from tropical peatland. *Global Change Biology*, *26*, 2477–2495. https://doi.org/10.1111/gcb.15019

Dhandapani, S., Girkin, N. T., Evers, S., Ritz, K., & Sjögersten, S. (2020). Is intercropping an environmentally-wise alternative to established oil palm monoculture in tropical peatlands? *Frontiers in Forests and Global Change*, *3*, 70. https://doi.org/10.3389/ffgc.2020.00070

Dhandapani, S., Ritz, K., Evers, S., Cooper, H., Tonks, A., & Sjögersten, S. (2020). Land-use changes associated with oil palm plantations impact PLFA microbial phenotypic community structure throughout the depth of tropical peats. *Wetlands*, *40*, 2351–2366. https://doi.org/10.1007/s13157-020-01342-0

Dhandapani, S., Ritz, K., Evers, S., Yule, C. M., & Sjögersten, S. (2019). Are secondary forests second-rate? Comparing peatland greenhouse gas emissions, chemical and microbial community properties between primary and secondary forests in peninsular Malaysia. *Science of the Total Environment*, *655*, 220–231. https://doi.org/10.1016/j.scitotenv.2018.11.046

Dohong, A. (2016). *An assessment of the restoration efforts of degraded peatland in central Kalimantan, Indonesia* (PhD thesis). The University of Queensland.

Dohong, A., Abdul Aziz, A., & Dargusch, P. (2018). A review of techniques for effective tropical peatland restoration. *Wetlands*, *38*, 275–292. https://doi.org/10.1007/s13157-018-1017-6

Dommain, R., Couwenberg, J., & Joosten, H. (2011). Development and carbon sequestration of tropical peat domes in south-east Asia: Links to post-glacial sea-level changes and Holocene climate variability. *Quaternary Science Reviews*, *30*, 999–1010. https://doi.org/10.1016/j.quascirev.2011.01.018

Dommain, R., Dittrich, I., Giesen, W., & Wibisono, I. (2016). Ecosystem services, degradation and restoration of peat swamps in the South East Asian tropics. In *Peatland Restoration and Ecosystem Services: Science, Policy and Practice. Ecological Reviews* (pp. 253–288).

Eggleton, P., Bignell, D. E., Sands, W. A., & Bignell, N. C. (1996). The diversity, abundance and biomass of termites under differing levels of disturbance in the Mbalmayo Forest Reserve, southern Cameroon. *Philosophical Transactions: Biological Sciences*, *35*, 51–68.

Erb, W. M., Barrow, E. J., Hofner, A. N., Utami-Atmoko, S. S., & Vogel, E. R. (2018). Wildfire smoke impacts activity and energetics of wild Bornean orangutans. *Scientific Reports*, *8*, 7606. https://doi.org/10.1038/s41598-018-25847-1

Evans, C. D., Page, S. E., Jones, T., Moore, S., Gauci, V., Laiho, R., Hruška, J., Allott, T. E. H., Billett, M. F., Tipping, E. D., Freeman, C., & Garnett, M. H. (2014). Contrasting vulnerability of drained tropical and high-latitude peatlands to fluvial loss of stored carbon. *Global Biogeochemical Cycles*, *28*, 1215–1234. https://doi.org/10.1002/2013GB004782

Evans, C. D., Williamson, J. M., Kacaribu, F., Irawan, D., Suardiwerianto, Y., Hidayat, M. F., Laurén, A., & Page, S. E. (2019). Rates and spatial variability of peat subsidence in *Acacia* plantation and forest landscapes in Sumatra, Indonesia. *Geoderma*, *338*, 410–421. https://doi.org/10.1016/j.geoderma.2018.12.028

Evers, S., Yule, C. M., Padfield, R., O'Reilly, P., & Varkkey, H. (2017). Keep wetlands wet: The myth of sustainable development of tropical peatlands - implications for policies and management. *Global Change Biology*, *23*(2), 534–549. https://doi.org/10.1111/gcb.13422

FAO. (2020). *Peatland mapping and monitoring–Recommendations and technical overview*. Ch-7.

Field, R. D., van der Werf, G. R., & Shen, S. S. P. (2009). Human amplification of drought-induced biomass burning in Indonesia since 1960. *Nature Geoscience*, *2*, 185–188.

Finn, D. R., Ziv-El, M., van Haren, J., Park, J. G., del Aguila-Pasquel, J., Urquiza–Muñoz, J. D., & Cadillo-Quiroz, H. (2020). Methanogens and methanotrophs show nutrient-dependent community assemblage patterns across tropical peatlands of the Pastaza-Marañón Basin, Peruvian amazonia. *Frontiers in Microbiology*, *11*, 746. https://doi.org/10.3389/fmicb.2020.00746

Fontaine, S., Bardoux, G., Abbadie, L., & Mariotti, A. (2004). Carbon input to soil may decrease soil carbon content. *Ecology Letters*, *7*, 314–320. https://doi.org/10.1111/j.1461-0248.2004.00579.x

Furukawa, Y., Inubushi, K., Ali, M., Itang, A. M., & Tsuruta, H. (2005). Effect of changing groundwater levels caused by land-use changes on greenhouse gas fluxes from tropical peat lands. *Nutrient Cycling in Agroecosystems*, *71*, 81–91.

Gandois, L., Teisserenc, R., Cobb, A. R., Chieng, H. I., Lim, L., Kamariah, A. S., Hoyt, A., & Harvey, C. F. (2014). Origin, composition, and transformation of dissolved organic matter in tropical peatlands. *Geochimica et Cosmochimica Acta*, *137*, 35–47. https://doi.org/10.1016/j.gca.2014.03.012

Gaveau, D. L. A., Salim, M. A., Hergoualc'h, K., Locatelli, B., Sloan, S., Wooster, M., Marlier, M. E., Molidena, E., Yaen, H., DeFries, R., Verchot, L., Murdiyarso, D., Nasi, R., Holmgren, P., & Sheil, D. (2015). Major atmospheric emissions from peat fires in Southeast Asia during non-drought years: Evidence from the 2013 Sumatran fires. *Scientific Reports*, *4*, 6112. https://doi.org/10.1038/srep06112

Giam, X., Koh, L. P., Tan, H. H., Miettinen, J., Tan, H. T. W., & Ng, P. K. L. (2012). Global extinctions of freshwater fishes follow peatland conversion in Sundaland. *Frontiers in Ecology and the Environment*, *10*(9), 465–470. https://doi.org/10.1890/110182

Giesen, W., & Sari, E. N. (2018). *Tropical peatland restoration report: The Indonesian case*. Technical report.

Giesen, W., Wijedasa, L. S., & Page, S. E. (2018). Unique Southeast Asian peat swamp forest habitats have relatively few distinctive plant species. *Mires and Peat*, *22*, 1–13.

Graham, L. L. B., Giesen, W., & Page, S. E. (2017). A common-sense approach to tropical peat swamp forest restoration in Southeast Asia. *Restoration Ecology*, *25*, 312–321. https://doi.org/10.1111/rec.12465

Graham, L. L. B., Turjaman, M., & Page, S. E. (2013). *Shorea balangeran* and *Dyera polyphylla* (syn. *Dyera lowii*) as tropical peat swamp forest restoration transplant species: Effects of mycorrhizae and level of disturbance. *Wetlands Ecology and Management*, *21*, 307–321.

Gumbricht, T., Roman-Cuesta, R. M., Verchot, L., Herold, M., Wittmann, F., Householder, E., Herold, N., & Murdiyarso, D. (2017). An expert system model for mapping tropical wetlands and peatlands reveals South America as the largest contributor. *Global Change Biology*, *23*, 3581–3599. https://doi.org/10.1111/gcb.13689

Hadi, A., Haridi, M., Inubushi, K., Purnomo, E., Razie, F., & Tsuruta, H. (2001). Effects of land-use change in tropical peat soil on the microbial population and emission of greenhouse gases. *Microbes and Environments*, *16*, 79–86. https://doi.org/10.1264/jsme2.2001.79

Harrison, M. E., Ottay, J. B., D'Arcy, L. J., Cheyne, S. M., Anggodo, Belcher, C., Cole, L., Dohong, A., Ermiasi, Y., Feldpausch, T., Gallego-Sala, A., Gunawan, A., Höing, A., Husson, S. J., Kulu, I. P., Soebagio, S. M., Mang, S., Mercado, L., Morrogh-Bernard, H. C., … Veen, F. J. F. (2020). Tropical forest and peatland conservation in Indonesia: Challenges and directions. *People and Nature*, *2*, 4–28. https://doi.org/10.1002/pan3.10060

HCSA. (2019). *High carbon stock approach*. Annual Report. Retrieved from http://highcarbonstock.org/wp-content/uploads/2020/04/HCSA-Annual-Report-2019_Final.pdf

Hidayat, M. R., Endris, W. M., & Dwiyanti, Y. (2018). Effect of a rubber plantation on termite diversity in Melawi, West Kalimantan, Indonesia. *Agriculture and Natural Resources*, *52*, 439–444. https://doi.org/10.1016/j.anres.2018.10.016

Hirano, T., Kusin, K., Limin, S., & Osaki, M. (2014). Carbon dioxide emissions through oxidative peat decomposition on a burnt tropical peatland. *Global Change Biology*, *20*, 555–565. https://doi.org/10.1111/gcb.12296

Hirano, T., Segah, H., Kusin, K., Limin, S., Takahashi, H., & Osaki, M. (2012). Effects of disturbances on the carbon balance of tropical peat swamp forests. *Global Change Biology*, *18*, 3410–3422. https://doi.org/10.1111/j.1365-2486.2012.02793.x

Hong, C. C., Li, T., LinHo, & Kug, J. S. (2008). Asymmetry of the Indian Ocean dipole. Part I: Observational analysis. *Journal of Climate*, *21*, 4834–4848.

Hooijer, A., Page, S., Jauhiainen, J., Lee, W. A., Lu, X. X., Idris, A., & Anshari, G. (2012). Subsidence and carbon loss in drained tropical peatlands. *Biogeosciences*, *9*, 1053–1071. https://doi.org/10.5194/bg-9-1053-2012

Hooijer, A., Vernimmen, R., Visser, M., & Mawdsley, N. (2015). *Flooding projections from elevation and subsidence models for oil palm plantations in the Rajang Delta peatlands*, Deltares report (p. 76).

Hoyos-Santillan, J., Lomax, B. H., Large, D., Turner, B. L., Lopez, O. R., Boome, A., Sepulveda-Jauregui, A., & Sjögersten, S. (2019). Evaluation of vegetation communities, water table, and peat composition as drivers of greenhouse gas emissions in lowland tropical peatlands. *Science of the Total Environment*, *688*(2019), 1193–1204. https://doi.org/10.1016/j.scitotenv.2019.06.36

Hoyt, A. M., Gandois, L., Eri, J., & Cobb, A. R. (2019). CO_2 emissions from an undrained tropical peatland: Interacting influences of temperature, shading and water table depth. *Global Change Biology*, *25*, 2885–2899.

Hu, Y., Fernandez-Anez, N., Smith, T. E. L., & Rein, G. (2018). Review of emissions from smouldering peat fires and their contribution to regional haze episodes. *International Journal of Wildland Fire*, *27*, 293. https://doi.org/10.1071/WF17084

Huijnen, V., Wooster, M. J., Kaiser, J. W., Gaveau, D. L. A., Flemming, J., Parrington, M., Inness, A., Murdiyarso, D., Main, B., & van Weele, M. (2016). Fire carbon emissions over maritime southeast Asia in 2015 largest since 1997. *Scientific Reports*, *6*, 26886. https://doi.org/10.1038/srep26886

Jackson, C. R., Liew, K. C., & Yule, C. M. (2009). Structural and functional changes with depth in microbial communities in a tropical Malaysian peat swamp forest. *Microbial Ecology*, *57*, 402–412. https://doi.org/10.1007/s00248-008-9409-4

Jaenicke, J., Englhart, S., & Siegert, F. (2011). Monitoring the effect of restoration measures in Indonesian peatlands by radar satellite imagery. *Journal of Environmental Management*, *92*, 630–638. https://doi.org/10.1016/j.jenvman.2010.09.029

Jaenicke, J., Wösten, H., Budiman, A., & Siegert, F. (2010). Planning hydrological restoration of peatlands in Indonesia to mitigate carbon dioxide emissions. *Mitigation and Adaptation Strategies for Global Change*, *15*, 223–239. https://doi.org/10.1007/s11027-010-9214-5

Jauhiainen, J., Kerojoki, O., Silvennoinen, H., Limin, S., & Vasander, H. (2014). Heterotrophic respiration in drained tropical peat is greatly affected by temperature—A passive ecosystem cooling experiment. *Environmental Research Letters*, *9*, 105013. https://doi.org/10.1088/1748-9326/9/10/105013

Jauhiainen, J., Limin, S., Silvennoinen, H., & Vasander, H. (2008). Carbon dioxide and methane fluxes in drained tropical peat before and after hydrological restoration. *Ecology*, *89*, 3503–3514. https://doi.org/10.1890/07-2038.1

Jauhiainen, J., Silvennoinen, H., Hämäläinen, R., Kusin, K., Limin, S., Raison, R. J., & Vasander, H. (2012). Nitrous oxide fluxes from tropical peat with different disturbance history and management. *Biogeosciences*, *9*, 1337–1350. https://doi.org/10.5194/bg-9-1337-2012

Jauhiainen, J., Takahashi, H., Heikkinen, J. E. P., Martikainen, P. J., & Vasander, H. (2005). Carbon fluxes from a tropical peat swamp forest floor. *Global Change Biology*, *11*, 1788–1797. https://doi.org/10.1111/j.1365-2486.2005.001031.x

Kanokratana, P., Uengwetwanit, T., Rattanachomsri, U., Bunterngsook, B., Nimchua, T., Tangphatsornruang, S., Plengvidhya, V., Champreda, V., & Eurwilaichitr, L. (2011). Insights into the phylogeny and metabolic potential of a primary tropical peat swamp forest microbial community by metagenomic analysis. *Microbial Ecology*, *61*, 518–528. https://doi.org/10.1007/s00248-010-9766-7

Kardol, P., & Wardle, D. A. (2010). How understanding aboveground–belowground linkages can assist restoration ecology. *Trends in Ecology & Evolution*, *25*, 670–679. https://doi.org/10.1016/j.tree.2010.09.001

Kelly, T. J., Baird, A. J., Roucoux, K. H., Baker, T. R., Honorio Coronado, E. N., Ríos, M., & Lawson, I. T. (2014). The high hydraulic conductivity of three wooded tropical peat swamps in northeast Peru: Measurements and implications for hydrological function. *Hydrological Processes*, *28*, 3373–3387. https://doi.org/10.1002/hyp.9884

Kiely, L., Spracklen, D. V., Wiedinmyer, C., Conibear, L., Reddington, C. L., Arnold, S. R., Knote, C., Khan, M. F., Latif, M. T., Syaufina, L., & Adrianto, H. A. (2020). Air quality and health impacts of vegetation and peat fires in Equatorial Asia during 2004–2015. *Environmental Research Letters*, *15*, 094054. https://doi.org/10.1088/1748-9326/ab9a6c

King, D. A., Davies, S. J., Supardi, M. N. N., & Tan, S. (2005). Tree growth is related to light interception and wood density in two mixed dipterocarp forests of Malaysia. *Functional Ecology*, *19*(3), 445–453. https://doi.org/10.1111/j.1365-2435.2005.00982.x

Konecny, K., Ballhorn, U., Navratil, P., Jubanski, J., Page, S. E., Tansey, K., Hooijer, A., Vernimmen, R., & Siegert, F. (2016). Variable carbon losses from recurrent fires in drained tropical peatlands. *Global Change Biology*, *22*, 1469–1480. https://doi.org/10.1111/gcb.13186

Könönen, M., Jauhiainen, J., Laiho, R., Kusin, K., & Vasander, H. (2015). Physical and chemical properties of tropical peat under stabilised land uses. *Mires and Peat*, *16*, 1–13.

Kroes, J. G., van Dam, J. C., Bartholomeus, R. P., van Walsum, P. E. V. (2017). *Wageningen Environmental Research*. SWAP version 4.

Kusai, N. A., Ayob, Z., Maidin, M. S. T., Safari, S., & Ahmad Ali, S. R. (2018). Characterization of fungi from different ecosystems of tropical peat in Sarawak, Malaysia. *Rendiconti Lincei. Scienze Fisiche e Naturali*, *29*, 469–482. https://doi.org/10.1007/s12210-018-0685-8

Kuzyakov, Y., Friedel, J. K., & Stahr, K. (2000). Review of mechanisms and quantification of priming effects. *Soil Biology*, *14*, 1485–1498. https://doi.org/10.1016/S0038-0717(00)00084-5

Kwon, M. J., Haraguchi, A., & Kang, H. (2013). Long-term water regime differentiates changes in decomposition and microbial properties in tropical peat soils exposed to the short-term drought. *Soil Biology and Biochemistry*, *60*, 33–44. https://doi.org/10.1016/j.soilbio.2013.01.023

Lampela, M., Jauhiainen, J., Sarkkola, S., & Vasander, H. (2017). Promising native tree species for reforestation of degraded tropical peatlands. *Forest Ecology and Management*, *394*, 52–63. https://doi.org/10.1016/j.foreco.2016.12.004

Lampela, M., Jauhiainen, J., & Vasander, H. (2014). Surface peat structure and chemistry in a tropical peat swamp forest. *Plant and Soil*, *382*, 329–347. https://doi.org/10.1007/s11104-014-2187-5

Langner, A., Miettinen, J., & Siegert, F. (2007). Land cover change 2002–2005 in Borneo and the role of fire derived from MODIS imagery. *Global Change Biology*, *13*, 2329–2340. https://doi.org/10.1111/j.1365-2486.2007.01442.x

Lieberman, D., Lieberman, M., Hartshorn, G., & Peralta, R. (1985). Growth rates and age-size relationships of tropical wet forest trees in Costa Rica. *Journal of Tropical Ecology*, *1*(2), 97–109. https://doi.org/10.1017/S026646740000016X

Lim, K. H., Lim, S. S., Parish, F., & Suharto, R. (Eds). (2012). *RSPO manual on best management practices (BMPs) for existing oil palm cultivation on peat*. RSPO. ISBN 978-967-11292-1-0 . http://www.rspo.org/

Liu, B., Talukder, M. J. H., Terhonen, E., Lampela, M., Vasander, H., Sun, H., & Asiegbu, F. (2020). The microbial diversity and structure in peatland forest in Indonesia. *Soil Use and Management*, *36*, 123–138. https://doi.org/10.1111/sum.12543

Lupascu, M., Akhtar, H., Smith, T. E. L., & Sukri, R. S. (2020). Post-fire carbon dynamics in the tropical peat swamp forests of Brunei reveal long-term elevated CH_4 flux. *Global Change Biology*, *26*, 5125–5145.

Manning, F. C., Kho, L. K., Hill, T. C., Cornulier, T., & Teh, Y. A. (2019). Carbon emissions from oil palm plantations on peat soil. *Frontiers in Forests and Global Change*, *2*, 37. https://doi.org/10.3389/ffgc.2019.00037

Marlier, M. E., DeFries, R. S., Kim, P. S., Koplitz, S. N., Jacob, D. J., Mickley, L. J., & Myers, S. S. (2015). Fire emissions and regional air quality impacts from fires in oil palm, timber, and logging concessions in Indonesia. *Environmental Research Letters*, *10*, 085005. https://doi.org/10.1088/1748-9326/10/8/085005

Marlier, M. E., DeFries, R. S., Voulgarakis, A., Kinney, P. L., Randerson, J. T., Shindell, D. T., Chen, Y., & Faluvegi, G. (2013). El Niño and health risks from landscape fire emissions in southeast Asia. *Nature Climate Change*, *3*, 131–136. https://doi.org/10.1038/nclimate1658

Miettinen, J., Hooijer, A., Shi, C., Tollenaar, D., Vernimmen, R., Liew, S. C., Malins, C., & Page, S. E. (2012). Extent of industrial plantations on Southeast Asian peatlands in 2010 with analysis of historical expansion and future projections. *GCB Bioenergy*, *4*, 908–918. https://doi.org/10.1111/j.1757-1707.2012.01172.x

Miettinen, J., Hooijer, A., Vernimmen, R., Liew, S. C., & Page, S. E. (2017). From carbon sink to carbon source: Extensive peat oxidation in insular Southeast Asia since 1990. *Environmental Research Letters*, *12*, 024014.

Miettinen, J., Shi, C., & Liew, S. C. (2016). Land cover distribution in the peatlands of peninsular Malaysia, Sumatra and Borneo in 2015 with changes since 1990. *Global Ecology and Conservation*, *6*, 67–78. https://doi.org/10.1016/j.gecco.2016.02.004

Mishra, S., Lee, W. A., Hooijer, A., Reuben, S., Sudiana, I. M., Idris, A., & Swarup, S. (2014). Microbial and metabolic profiling reveal strong influence of water table and land-use patterns on classification of degraded tropical peatlands. *Biogeosciences*, *11*, 1727–1741. https://doi.org/10.5194/bg-11-1727-2014

Moore, S., Evans, C. D., Page, S. E., Garnett, M. H., Jones, T. G., Freeman, C., Hooijer, A., Wiltshire, A. J., Limin, S. H., & Gauci, V. (2013). Deep instability of deforested tropical peatlands revealed by fluvial organic carbon fluxes. *Nature*, *493*, 660–663. https://doi.org/10.1038/nature11818

Moore, S., Gauci, V., Evans, C. D., & Page, S. E. (2011). Fluvial organic carbon losses from a Bornean blackwater river. *Biogeosciences*, *8*, 901–909. https://doi.org/10.5194/bg-8-901-2011

Moyersoen, B., Becker, P., & Alexander, I. J. (2001). Are ectomycorrhizas more abundant than arbuscular mycorrhizas in tropical heath forests? *New Phytologist*, *150*, 591–599. https://doi.org/10.1046/j.1469-8137.2001.00125.x

Mulyani, R. B., Sastrahidayat, I. R., Abadi, A. L., & Djauhari, S. (2014). Exploring ectomycorrhiza in peat swamp forest of Nyaru Menteng Palangkaraya Central Borneo. *Journal of Biodiversity and Environmental Sciences*, *5*, 133–145.

Neoh, K.-B., Bong, L.-J., Muhammad, A., Itoh, M., Kozan, O., Takematsu, Y., & Yoshimura, T. (2016). The impact of tropical peat fire on termite assemblage in Sumatra, Indonesia: Reduced complexity of community structure and survival strategies. *Environmental Entomology*, *45*, 1170–1177. https://doi.org/10.1093/ee/nvw116

Neuzil, S. G., Cecil, C. B., Kane, J. S., & Soedjono, K. (1993). Inorganic geochemistry of domed peat in Indonesia and its implication for the origin of mineral matter in coal. *Geological Society of America Special Papers*, *286*, 23–44.

Nuyim, T. (2005). Guideline on peat swamp forests rehabilitation and planting in Thailand. In *Global Environment Centre and Wetlands International*. ISBN 983-3377-01-7. https://agris.fao.org/agris-search/search.do?recordID=US201300140438

Ong, C. S. P., Juan, J. C., & Yule, C. M. (2015). Litterfall production and chemistry of *Koompassia malaccensis* and *Shorea uliginosa* in a tropical peat swamp forest: Plant nutrient regulation and climate relationships. *Trees*, *29*, 527–537. https://doi.org/10.1007/s00468-014-1130-y

Padfield, R., Drew, S., Syayuti, K., Page, S., Evers, S., Campos-Arceiz, A., Kangayatkarasu, N., Sayok, A., Hansen, S., Schouten, G., Maulidia, M., Papargyropoulou, E., & Tham, M. H. (2016). Landscapes in transition: An analysis of sustainable policy initiatives and emerging corporate commitments in the palm oil industry. *Landscape Research*, *41*, 744–756. https://doi.org/10.1080/01426397.2016.1173660

Page, S. E., & Hooijer, A. (2016). In the line of fire: the peatlands of Southeast Asia. *Philosophical Transactions of the Royal Society B: Biological Sciences*, *371*(1696), 20150176. https://doi.org/10.1098/rstb.2015.0176

Page, S., Hoscilo, A., Wösten, H., Jauhiainen, J., Silvius, M., Rieley, J., Ritzema, H., Tansey, K., Graham, L., Vasander, H., & Limin, S. (2009). Restoration ecology of lowland tropical peatlands in Southeast Asia: Current knowledge and future research directions. *Ecosystems*, *12*, 888–905. https://doi.org/10.1007/s10021-008-9216-2

Page, S. E., Rieley, J. O., & Banks, C. J. (2011). Global and regional importance of the tropical peatland carbon pool. *Global Change Biology*, *17*, 798–818. https://doi.org/10.1111/j.1365-2486.2010.02279.x

Page, S. E., Rieley, J. O., Shotyk, Ø. W., & Weiss, D. (1999). Interdependence of peat and vegetation in a tropical peat swamp forest. *Changes and Disturbance in Tropical Rainforest SE-Asia*, 161–173. https://doi.org/10.1098/rstb.1999.0529

Page, S. E., Rieley, J. O., & Wüst, R. (2006). Ch-7: Lowland tropical peatlands of Southeast Asia. *Developments in Earth Surface Processes*, *9*, 145–172.

Page, S. E., Siegert, F., Rieley, J. O., Boehm, H.-D., Jaya, A., & Limin, S. (2002). The amount of carbon released from peat and forest fires in Indonesia during 1997. *Nature*, *420*, 61–65. https://doi.org/10.1038/nature01131

Posa, M. R. C., Wijedasa, L. S., & Corlett, R. T. (2011). Biodiversity and conservation of tropical peat swamp forests. *BioScience*, *61*, 49–57. https://doi.org/10.1525/bio.2011.61.1.10

Puspitaloka, D., Kim, Y., Purnomo, H., & Fulé, P. Z. (2020). Defining ecological restoration of peatlands in Central Kalimantan, Indonesia. *Restoration Ecology*, *28*, 435–446. https://doi.org/10.1111/rec.13097

Quinten, M. C., Waltert, M., Syamsuri, F., & Hodges, J. K. (2010). Peat swamp forest supports high primate densities on Siberut Island, Sumatra. *Indonesia. Oryx*, *44*, 147. https://doi.org/10.1017/S0030605309990718

Reid, J. S., Hyer, E. J., Johnson, R. S., Holben, B. N., Yokelson, R. J., Zhang, J., Campbell, J. R., Christopher, S. A., Di Girolamo, L., Giglio, L., Holz, R. E., Kearney, C., Miettinen, J., Reid, E. A., Turk, F. J., Wang, J., Xian, P., Zhao, G., Balasubramanian, R., … Liew, S. C. (2013). Observing and understanding the Southeast Asian aerosol system by remote sensing: An initial review and analysis for the Seven Southeast Asian Studies (7SEAS) program. *Atmospheric Research*, *122*, 403–468. https://doi.org/10.1016/j.atmosres.2012.06.005

Reid, J. S., Xian, P., Hyer, E. J., Flatau, M. K., Ramirez, E. M., Turk, F. J., Sampson, C. R., Zhang, C., Fukada, E. M., & Maloney, E. D. (2012). Multi-scale meteorological conceptual analysis of observed active fire hotspot activity and smoke optical depth in the Maritime Continent. *Atmospheric Chemistry and Physics*, *12*, 2117–2147. https://doi.org/10.5194/acp-12-2117-2012

Republic of Indonesia. (2016). First nationally determined contribution, Report submitted to UNFCCC

Ritzema, H., Limin, S., Kusin, K., Jauhiainen, J., & Wösten, H. (2014). Canal blocking strategies for hydrological restoration of degraded tropical peatlands in Central Kalimantan, Indonesia. *Catena*, *114*, 11–20. https://doi.org/10.1016/j.catena.2013.10.009

Rixen, T., Baum, A., Wit, F., & Samiaji, J. (2016). Carbon leaching from tropical peat soils and consequences for carbon balances. *Frontiers in Earth Science*, *4*, 74. https://doi.org/10.3389/feart.2016.00074

Santika, T., Budiharta, S., Law, E. A., Dennis, R. A., Dohong, A., Struebig, M. J., Medrilzam, Gunawan, H., Meijaard, E., & Wilson, K. A. (2020). Interannual climate variation, land type and village livelihood effects on fires in Kalimantan. *Indonesia. Global Environmental Change*, *64*, 102129. https://doi.org/10.1016/j.gloenvcha.2020.102129

Sazawa, K., Wakimoto, T., Fukushima, M., Yustiawati, Y., Syawal, M. S., Hata, N., Taguchi, S., Tanaka, S., Tanaka, D., & Kuramitz, H. (2018). Impact of peat fire on the soil and export of dissolved organic carbon in tropical peat soil, Central Kalimantan, Indonesia. *ACS Earth and Space Chemistry*, *2*, 692–701. https://doi.org/10.1021/acsearthspacechem.8b00018

Shuhada, S. N., Salim, S., Nobilly, F., Zubaid, A., & Azhar, B. (2017). Logged peat swamp forest supports greater macrofungal biodiversity than large-scale oil palm plantations and smallholdings. *Ecology and Evolution*, *7*, 7187–7200. https://doi.org/10.1002/ece3.3273

Silvius, M. J., & Suryadiputra, N. (2005). *Review of policies and practices in tropical peat swamp forest management in Indonesia*. Wetlands International report.

Simpson, J., Wooster, M., Smith, T., Trivedi, M., Vernimmen, R., Dedi, R., Shakti, M., & Dinata, Y. (2016). Tropical peatland burn depth and combustion heterogeneity assessed using UAV photogrammetry and airborne LiDAR. *Remote Sensing*, *8*, 1000. https://doi.org/10.3390/rs8121000

Sjögersten, S., Aplin, P., Gauci, V., Peacock, M., Siegenthaler, A., & Turner, B. L. (2018). Temperature response of ex-situ greenhouse gas emissions from tropical peatlands: Interactions between forest type and peat moisture conditions. *Geoderma*, *324*, 47–55. https://doi.org/10.1016/j.geoderma.2018.02.029

Sjögersten, S., Cheesman, A. W., Lopez, O., & Turner, B. L. (2011). Biogeochemical processes along a nutrient gradient in a tropical ombrotrophic peatland. *Biogeochemistry*, *104*, 147–163. https://doi.org/10.1007/s10533-010-9493-7

Smith, T. E. L., Evers, S., Yule, C. M., & Gan, J. Y. (2018). In-situ tropical peatland fire emission factors and their variability, as determined by field measurements in peninsula Malaysia. *Global Biogeochemical Cycles*, *32*, 18–31. https://doi.org/10.1002/2017GB005709

Stockwell, C. E., Jayarathne, T., Cochrane, M. A., Ryan, K. C., Putra, E. I., Saharjo, B. H., Nurhayati, A. D., Albar, I., Blake, D. R., Simpson, I. J., Stone, E. A., & Yokelson, R. J. (2016). Field measurements of trace gases and aerosols emitted by peat fires in Central Kalimantan, Indonesia, during the 2015 El Niño. *Atmospheric Chemistry and Physics*, *16*, 11711–11732.

Stringer, L. C., Dougill, A. J., Fraser, E., Hubacek, K., Prell, C., & Reed, M. S. (2006). Unpacking "Participation" in the adaptive management of social-ecological systems: A critical review. *Ecology and Society*, *11*, 39.

Sulistiyanto, Y. (2005). *Nutrient dynamics in different sub-types of peat swamp forest in central Kalimantan, Indonesia* (PhD thesis). University of Nottingham.

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

Talukder, M., & Sun, H. (2019). The microbial diversity and structure in peat land forest in Indonesia. *Journal of Biodiversity Conservation and Bioresource Management*, *5*, 133–144. https://doi.org/10.3329/jbcbm.v5i1.42193

Tan, Z. D., Carrasco, L. R., & Taylor, D. (2020). Spatial correlates of forest and land fires in Indonesia. *International Journal of Wildland Fire*, *29*(12), 1088. https://doi.org/10.1071/WF20036

Tan, Z. D., Lupascu, M., & Wijedasa, L. S. (2020). Paludiculture as a sustainable land use alternative for tropical peatlands: A review. *Science of the Total Environment*, *753*, 142111. https://doi.org/10.1016/j.scitotenv.2020.142111

Taufik, M., Minasny, B., McBratney, A. B., Van Dam, J. C., Jones, P. D., & Van Lanen, H. A. J. (2020). Human-induced changes in Indonesian peatlands increase drought severity. *Environmental Research Letters*, *15*(8), 084013. https://doi.org/10.1088/1748-9326/ab96d4

Taufik, M., Setiawan, B. I., & Van Lanen, H. A. J. (2019). Increased fire hazard in human-modified wetlands in Southeast Asia. *Ambio*, *48*(4), 363–373. https://doi.org/10.1007/s13280-018-1082-3

Taufik, M., Torfs, P. J. J. F., Uijlenhoet, R., Jones, P. D., Murdiyarso, D., & Van Lanen, H. A. J. (2017). Amplification of wildfire area burnt by hydrological drought in the humid tropics. *Nature Climate Change*, *7*(6), 428–431. https://doi.org/10.1038/nclimate3280

Tawaraya, K., Takaya, Y., Turjaman, M., Tuah, S. J., Limin, S. H., Tamai, Y., Cha, J. Y., Wagatsuma, T., & Osaki, M. (2003). Arbuscular mycorrhizal colonization of tree species grown in peat swamp forests of Central Kalimantan, Indonesia. *Forest Ecology and Management*, *182*, 381–386. https://doi.org/10.1016/S0378-1127(03)00086-0

Tay, T. H. (1969). The distribution, characteristics, uses and potential of peat in west Malaysia. *Journal of Tropical Geography*, *29*, 57–63.

Thornton, S. A., Dudin, Page, S. E., Caroline, U., & Harrison, M. E. (2018). Peatland fish of Sebangau, Borneo: Diversity, monitoring and conservation. *Mires and Peat*, *22*, 1–25.

Thornton, S. A., Setiana, E., Yoyo, K., Dudin, Yulintine, Harrison, M. E., Page, S. E., & Upton, C. (2020). Towards biocultural approaches to peatland conservation: The case for fish and livelihoods in Indonesia. *Environmental Science & Policy*, *114*, 341–351. https://doi.org/10.1016/j.envsci.2020.08.018

Tonks, A. J., Aplin, P., Beriro, D. J., Cooper, H., Evers, S., Vane, C. H., & Sjögersten, S. (2017). Impacts of conversion of tropical peat swamp forest to oil palm plantation on peat organic chemistry, physical properties and carbon stocks. *Geoderma*, *289*, 36–45. https://doi.org/10.1016/j.geoderma.2016.11.018

Too, C. C., Keller, A., Sickel, W., Lee, S. M., & Yule, C. M. (2018). Microbial community structure in a Malaysian tropical peat swamp forest: The influence of tree species and depth. *Frontiers in Microbiology*, *9*, 2859. https://doi.org/10.3389/fmicb.2018.02859

Turjaman, M., Santoso, E., Susanto, A., Gaman, S., Limin, S. H., Tamai, Y., Osaki, M., & Tawaraya, K. (2011). Ectomycorrhizal fungi promote growth of *Shorea balangeran* in degraded peat swamp forests. *Wetlands Ecology and Management*, *19*, 331–339. https://doi.org/10.1007/s11273-011-9219-1

Uda, S. K., Hein, L., & Sumarga, E. (2017). Towards sustainable management of Indonesian tropical peatlands. *Wetlands Ecology and Management*, *25*(6), 683–701. https://doi.org/10.1007/s11273-017-9544-0

Uda, S. K., Schouten, G., & Hein, L. (2020). The institutional fit of peatland governance in Indonesia. *Land Use Policy*, *99*, 103300. https://doi.org/10.1016/j.landusepol.2018.03.031

Vaessen, T., Verwer, C., Demies, M., Kaliang, H., & van der Meer, P. J. (2011). Comparison of termite assemblages along a land-use gradient on peat areas in Sarawak, Malaysia. *Journal of Tropical Forest Science*, *23*, 196–203.

Vernimmen, R., Hooijer, A., Akmalia, R., Fitranatanegara, N., Mulyadi, D., Yuherdha, A., Andreas, H., & Page, S. (2020). Mapping deep peat carbon stock from a LiDAR based DTM and field measurements, with application to eastern Sumatra. *Carbon Balance and Management*, *15*(1), 4. https://doi.org/10.1186/s13021-020-00139-2

Vetrita, Y., & Cochrane, M. A. (2019). Fire frequency and related land-use and land-cover changes in Indonesia's Peatlands. *Remote Sensing*, *12*(1), 5. https://doi.org/10.3390/rs12010005

Ward, C., Stringer, L. C., Warren-Thomas, E., Agus, F., Hamer, K., Pettorelli, N., Hariyadi, B., Hodgson, J., Kartika, W. D., Lucey, J., McClean, C., Nurida, N. L., Saad, A., & Hill, J. K. (2020). Wading through the swamp: What does tropical peatland restoration mean to national-level stakeholders in Indonesia? *Restoration Ecology*, *28*, 817–827. https://doi.org/10.1111/rec.13133

Weiss, D., Shotyk, W., Rieley, J., Page, S., Gloor, M., Reese, S., & Martinez-Cortizas, A. (2002). The geochemistry of major and selected trace elements in a forested peat bog, Kalimantan, SE Asia, and its implications for past atmospheric dust deposition. *Geochimica et Cosmochimica Acta*, *66*, 2307–2323. https://doi.org/10.1016/S0016-7037(02)00834-7

Wich, S. A., Meijaard, E., Marshall, A. J., Singleton, I. (2008). Distribution and conservation status of the orangutan (Pongo spp.) on Borneo and Sumatra: How many remain? *Oryx*, *42*, 329–339.

Wichtmann, W., & Joosten, H. (2007). Paludiculture: Peat formation and renewable resources from rewetted peatlands. *IMCG Newsletter*. https://www.researchgate.net/profile/Hans-Joosten/publication/265622374_Paludiculture_Peat_formation_and_renewable_resources_from_rewetted_peatlands/links/541999140cf25ebee98875ee/Paludiculture-Peat-formation-and-renewable-resources-from-rewetted-peatlands.pdf

Widayati, A., Tata, H. L., & Noordwijk, M. V. (2016). *Agroforestry on peatlands: Combining productive and protective functions as part of Restoration*. World Agroforestry Centre (ICRAF) Policy Brief No. 70.

Wiggins, E. B., Czimczik, C. I., Santos, G. M., Chen, Y., Xu, X., Holden, S. R., Randerson, J. T., Harvey, C. F., Kai, F. M., & Yu, L. E. (2018). Smoke radiocarbon measurements from Indonesian fires provide evidence for burning of millennia-aged peat. *Proceedings of the National Academy of Sciences of the United States of America*, *115*, 12419–12424. https://doi.org/10.1073/pnas.1806003115

Wijedasa, L. S., Jauhiainen, J., Könönen, M., Lampela, M., Vasander, H., Leblanc, M. C., Evers, S., Smith, T. E., Yule, C. M., Varkkey, H., Lupascu, M., Lupascu, M., Parish, F., Singleton, I., Clements, G. R., Aziz, S. A., Harrison, M. E., Cheyne, S., Anshari, G. Z., … Andersen, R. (2016). Denial of long-term issues with agriculture on tropical peatlands will have devastating consequences. *Global Change Biology*, *23*(3).

Wijedasa, L. S., Sloan, S., Page, S. E., Clements, G. R., Lupascu, M., & Evans, T. A. (2018). Carbon emissions from South-East Asian peatlands will increase despite emission-reduction schemes. *Global Change Biology*, *24*, 4598–4613. https://doi.org/10.1111/gcb.14340

Wijedasa, L. S., Vernimmen, R., Page, S. E., Mulyadi, D., Bahri, S., Randi, A., Evans, T. A., Lasmito, Priatna, D., Jensen, R. M., & Hooijer, A. (2020). Distance to forest, mammal and bird dispersal drive natural regeneration on degraded tropical peatland. *Forest Ecology and Management*, *461*, 117868. https://doi.org/10.1016/j.foreco.2020.117868

Wilson, D., Blain, D., Couwenberg, J., Evans, C. D., Murdiyarso, D., Page, S. E., Renou-Wilson, F., Rieley, J. O., Sirin, A., Strack, M., & Tuittila, E. S. (2016). Greenhouse gas emission factors associated with rewetting of organic soils. *Mires and Peat*, *17*, 1–28.

Wit, F., Müller, D., Baum, A., Warneke, T., Pranowo, W. S., Müller, M., & Rixen, T. (2015). The impact of disturbed peatlands on river outgassing in Southeast Asia. *Nature Communications*, *6*, 10155. https://doi.org/10.1038/ncomms10155

Wright, E. L., Black, C. R., Turner, B. L., & Sjögersten, S. (2013). Environmental controls of temporal and spatial variability in CO_2 and CH_4 fluxes in a neotropical peatland. *Global Change Biology*, *19*, 3775–3789.

Wüst, R. A. J., & Bustin, R. M. (2003). Opaline and Al–Si phytoliths from a tropical mire system of west Malaysia: Abundance, habit, elemental composition, preservation and significance. *Chemical Geology*, *200*, 267–292. https://doi.org/10.1016/S0009-2541(03)00196-7

Yin, Y. I., Ciais, P., Chevallier, F., van der Werf, G. R., Fanin, T., Broquet, G., Boesch, H., Cozic, A., Hauglustaine, D., Szopa, S., & Wang, Y. (2016). Variability of fire carbon emissions in equatorial Asia and its nonlinear sensitivity to El Niño. *Geophysical Research Letters*, *43*, 10472–10479. https://doi.org/10.1002/2016GL070971

Yule, C. M. (2010). Loss of biodiversity and ecosystem functioning in Indo-Malayan peat swamp forests. *Biodiversity and Conservation*, *19*, 393–409. https://doi.org/10.1007/s10531-008-9510-5

Yule, C. M., & Gomez, L. N. (2009). Leaf litter decomposition in a tropical peat swamp forest in Peninsular Malaysia. *Wetlands Ecology and Management*, *17*, 231–241. https://doi.org/10.1007/s11273-008-9103-9

Yuwati, T. W., & Putri, W. S. (2020). Diversity of arbuscular mycorrhiza spores under Shorea balangeran (Korth.) Burck. Plantation as bioindicator or the revegetation success. *Jurnal Galam*, *1*, 15–26.

Zeng, Y., Sarira, T. V., Carrasco, L. R., Chong, K. Y., Friess, D. A., Lee, J. S. H., Taillardat, P., Worthington, T. A., Zhang, Y., & Koh, L. P. (2020). Economic and social constraints on reforestation for climate mitigation in Southeast Asia. *Nature Climate Change*, *10*, 842–844. https://doi.org/10.1038/s41558-020-0856-3

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