Anthropocene 40 (2022) 100354

Contents lists available at ScienceDirect

Anthropocene

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

Invited Review Article

Tropical peatlands in the Anthropocene: The present and the future

Nicholas T. Girkin [a,*], Hannah V. Cooper [b], Martha J. Ledger [c], Patrick O'Reilly [d], Sara A. Thornton [d,e], Christine M. Åkesson [f], Lydia E.S. Cole [f], K. Anggi Hapsari [g,h], Donna Hawthorne [f], Katherine H. Roucoux [f]

[a] School of Water, Energy and Environment, Cranfield University, UK
[b] Sustainable Soils and Crops, Rothamsted Research, UK
[c] School of Biosciences, University of Nottingham, UK
[d] School of Geology, Geography and the Environment. University of Leicester, UK
[e] Ecosystem Health & Social Dimensions, Wildfowl & Wetlands Trust, UK
[f] School of Geography and Sustainable Development, University of St Andrews, UK
[g] Albrecht-von-Haller Institute, University of Goettingen, Germany
[h] College of Life and Environmental Sciences, University of Exeter, UK

ARTICLE INFO

Key words:
Tropical peatlands
Land use change
Palm oil
Climate change
Biodiversity, carbon cycle

ABSTRACT

Tropical peatlands are a globally important carbon store. They host significant biodiversity and provide a range of other important ecosystem services, including food and medicines for local communities. Tropical peatlands are increasingly modified by humans in the rapid and transformative way typical of the "Anthropocene," with the most significant human—driven changes to date occurring in Southeast Asia. This review synthesizes the dominant changes observed in human interactions with tropical peatlands in the last 200 years, focusing on the tropical lowland peatlands of Southeast Asia. We identify the beginning of transformative anthropogenic processes in these carbon-rich ecosystems, chart the intensification of these processes in the 20th and early 21st centuries, and assess their impacts on key ecosystem services in the present. Where data exist, we compare the tropical peatlands of Central Africa and Amazonia, which have experienced very different scales of disturbance in the recent past. We explore their global importance and how environmental pressures may affect them in the future. Finally, looking to the future, we identify ongoing efforts in peatland conservation, management, restoration, and socio-economic development, as well as areas of fruitful research toward sustainability of tropical peatlands.

1. Introduction

Tropical peatlands are a carbon-rich wetland ecosystem of critical importance to global carbon cycles. Despite accounting for approximately 14% of total peatland area, tropical peatlands store 69.6 – 129.8 Gt C, equivalent to 19% of total peatland carbon stocks (Dargie et al., 2017; Page et al., 2011), with significant uncertainty remaining regarding their geographic extent and the volume of carbon they contain (Xu et al., 2018). By contrast, drained tropical peatlands can become a significant source of atmospheric greenhouse gases (GHGs), contributing annual emissions of up to 4540 Tg carbon dioxide (CO_2) and 90 Tg methane (CH_4) (Sjögersten et al., 2014); the scale of nitrous oxide (N_2O) fluxes remains to be clarified (Cooper et al., 2020; Leifeld and Menichetti, 2018). Substantial recent effort has focussed on mapping peatland ecosystems across the tropics, to quantify key ecological functions, monitor rates of loss to global environmental changes, and conserve peatland ecosystems. Within the latest decades effort has focused predominantly on South America, Central Africa, and Southeast Asia (Page et al., 2011), although tropical peatland ecosystems extend more broadly than this (Xu et al., 2018), covering approximately 58.7 Mha of peatlands between 23.5°N and 23.5°S (Leifeld and Menichetti, 2018; Cole et al., 2022). Large areas of tropical peatlands, particularly in Southeast Asia, are increasingly characterised by a high degree of disturbance, with rates of change drastically accelerating since the mid-20th century driven by rapid economic development (Page et al., 2006). This dramatic change in peatland condition has led to the heightened recognition of their importance at a regional (e.g. ASEAN, 2016) to global scale (e.g., GPI, 2021).

* Corresponding author.
E-mail address: N.T.Girkin@cranfield.ac.uk (N.T. Girkin).

https://doi.org/10.1016/j.ancene.2022.100354
Received 25 July 2022; Received in revised form 17 November 2022; Accepted 17 November 2022
Available online 19 November 2022

The scale of recent anthropogenic driven changes in Southeast Asian tropical peatlands is vast: for example, between 1995 and 1999, Indonesia's Mega-Rice project aimed to convert 1 Mha of peatlands to rice production. While this was never fully achieved, the resulting deforestation and drainage increased fire vulnerability and ultimately resulted in the loss of up to 2.57 Gt C during the 1997 El Niño driven fire events (Page et al., 2002). Between 2000 and 2010, over 2756 Mha of tropical peatlands were lost in Malaysia alone (19.7%), predominantly to expanding agriculture, with ongoing annual losses of 2.2% (Miettinen and Liew, 2010). As a result of drainage, El Niño driven fire events in peatlands across Southeast Asia in 2015 released the equivalent of the total CO_2 emissions of Europe for a three month period over an eight week period (Huijnen et al., 2016). Beyond Southeast Asia, tropical peatlands globally face a range of ongoing threats, including over-exploitation of non-timber forest products, logging, conversion to agriculture, modification of hydrological systems, fire, urban expansion, exploitation from mining, oil and gas extraction, as well as the impacts of climate change (Dargie et al., 2018; Roucoux et al., 2017). Cataloguing and understanding the extent of human modification of tropical peatlands is therefore critical.

Previous reviews discussing anthropogenic changes in tropical peatlands have reported on carbon storage and dynamics (Hergoualc'H and Verchot, 2011; Leifeld et al., 2019), fluxes of greenhouse gases in the context of land use change or management (Van Lent et al., 2015, Leifeld et al., 2019), fire events (Goldammer 2006; Page and Hooijer, 2016), or the impacts of climate change (Leng et al., 2018), or have been regional case studies (Goldammer 2006; Dargie et al., 2018; Roucoux et al., 2017). To date, an assessment of the effects of anthropogenic processes and impacts on tropical peatlands has not been available in a long-term historical and prehistorical context at a global scale. This paper synthesizes the literature on the development of human-driven transformation in tropical peatlands, consider the extent of anthropogenic changes, discuss future trends within this critically important ecosystem, and identify fruitful avenues of research for the sustainability of tropical peatlands.

2. Tracing human impacts on tropical peatlands

2.1. Defining the anthropocene in tropical peatlands

The precise definition of the "Anthropocene" has been debated at length elsewhere, but most descriptions share the premise that it reflects a time period in which the scale of human environmental impacts has induced Earth System changes that will persist for millennia or longer (Lewis and Maslin, 2015).

For the purposes of this review, we use the anthropocene as a term to refer to the period in which tropical peatlands have experienced an intensification of anthropogenic processes that has led to transformative change. These changes have occurred asynchronously in time and space across tropical peatlands. Contemporary observations show that anthropogenic impacts are most pronounced in Southeast Asia, primarily in Indonesia and Malaysia (Lilleskov et al., 2019). There is palaeoecological evidence to suggest that the human activities responsible for transformative impacts on tropical peatlands in Southeast Asia began *c.* 200 years ago (Cole et al., 2015), but further intensified within the last 50 years (Evers et al., 2017; Wijedasa et al., 2017). This recent intensification is in contrast to the lower intensity historical and pre-historical interactions that people have had with peatlands across the tropical latitudes throughout the Holocene. Information on these past interactions and peatland dynamics provides important context for understanding the relative changes in these ecosystems that we are experiencing today (Cole at al, 2022).

Anthropogenic impacts on tropical peatlands have both proximate and distal drivers. In tropical peatlands, proximate causes include changes in land and hydrological management, cropping and settlement (Cooper et al., 2020; Dargie et al., 2018; Evers et al., 2017; Roucoux et al., 2017), while distal causes include climate change and shifting global demand for natural resources and agricultural commodities (Evers et al., 2017; Wang et al., 2018). Proximate effects include loss of biodiversity and peat subsidence; while a key distal effect is the consequence of peatland degradation on global GHG budgets (Leifeld et al., 2019; Sjögersten et al., 2014). This review describes the anthropogenic processes that tropical peatlands are being exposed to in the anthropocene, the impacts they are having on the integrity of these ecosystems, and what this information could tell us about future tropical peatland resilience. Throughout, the spatial and temporal scale of processes and impacts are highlighted, and contemporary changes contextualised using available information on anthropogenic interactions with these ecosystems in the past. We refer to the anthropocene in this synthesis to emphasise the period in the recent past, i.e., within the last 200 years, when tropical peatland ecosystems, specifically, became under threat of irreversible damage from human activities. Thus, this review describes the anthropogenic processes that tropical peatlands are being exposed to in the anthropocene, the impacts they are having on the integrity of these ecosystems, and what this information could tell us about future tropical peatland resilience. Throughout, the spatial and temporal scale of processes and impacts are highlighted, and contemporary changes contextualised using available data on anthropogenic interactions with these ecosystems in the past. In using this term here, we also acknowledge the importance of critically reflecting on it, in particular because it implies the pervasive impacts of a global society, whilst neglecting the underlying power imbalances and vastly unequal contributions of different cultures and communities to the environmental impacts we associate with this period (Malm and Hornborg, 2014).

2.2. Tropical peatland formation and early anthropogenic activity

Peat is formed from partially decomposed organic matter that accumulates under waterlogged, acidic and anoxic conditions (Moore, 1989). Accumulation rates are spatially and temporally variable (e.g., Dommain et al., 2011), but are generally greater in the tropics than in temperate and boreal regions as higher temperatures enhance plant productivity and organic inputs to the peat. While temperate and boreal peatlands are frequently moss-dominated and often open or only sparsely forested, tropical peatlands are commonly forested, and can include broadleaved evergreen trees, mangroves, and palm dominated areas. For example, in the lowland peatlands of Amazonia in the Pastaza-Marañón Foreland Basin (Fig. 1), peatland 'pole forests' are the most carbon-dense ecosystem dominated by thin-stemmed tree species, such as *Pachira nitida* (Malvaceae), *Platycarpum loretense* (Rubiaceae) and *Hevea guianensis* (Euphorbiaceae), whilst palm swamps, dominated by *Mauritia flexuosa* (Arecaceae), can also contain thick peat deposits (Coronado et al., 2021). Peatlands in Central Africa are similarly dominated by broadleaved evergreen trees including *Carapa procera* (Meliaceae) and *Symphonia globulifera* (Clusiaceae) and canopy palms, frequently *Raphia laurentii* (Arecaceae). Coastal peatlands in across Central America and Southeast Asia can frequent be found under mangrove forests (Alongi, 2015; Phillips et al., 1997), with accumulation of up to several metres. Odum et al., (1982) highlighted that red mangrove (*Rhizophora mangle*) is the main driver of peat formation in these systems, although black mangrove (*Avicennia germinans*) has been reported in peat-forming mangroves in Cambodia, and Mexico (Lo et al., 2018).

In these forested tropical peatlands, the peat is predominantly formed from root, stem and leaf material (Hoyos-Santillan et al., 2015). The timing of peat initiation varies widely, depending on local climatic, hydrological, ecological, and geomorphological conditions (Anderson and Muller, 1975; Aucour et al., 1999; Dargie et al., 2017). Globally, tropical peatlands have accumulated 44 – 55 Gt C since the start of the Holocene, with a mean rate of accumulation of 12.8 g C m^{-2} yr^{-1}. Rates for the last two thousand years are higher, ranging from 20 to 50 g C m^{-2} yr^{-1} (Yu et al., 2010), with the majority of undisturbed peatlands

Fig. 1. Distribution of tropical peatlands and well-studied, exemplar peatland ecosystems within each region. Peatland map adapted from Xu et al., 2017. (1) Changuinola peat deposit, Panama; (2) Pastaza-Marañón Foreland Basin, Peru; (3) Central African Peatland Complex (Cuvette Centrale), Republic of Congo and Democratic Republic of Congo; (4) Kalimantan, Indonesia; (5) West Papua (Papua Barat), Indonesia.

continuing to accumulate carbon to the present day (Page et al., 2006).

Since the initiation of peat formation in a diversity of tropical peatlands, which commonly occurred in the Holocene (Cole et al., 2021), there is compelling evidence that human impacts on these ecosystems have been of relatively low intensity (Cole et al., 2022). In many cases, it is clear that human activity in these areas predates peatland initiation (Barker et al., 2007; Hunt and Premathilake, 2012; Roberts et al., 2018). Evidence from a peatland area in the northern Peruvian Amazonia suggests that pre-Columbian anthropogenic activity was associated with only minimal changes in forest cover (Kelly et al., 2018). Data from tropical peatland areas in the Congo Basin (Wosten et al., 2015), Indonesia (Hope et al., 2005) and Malaysia (Cole et al., 2019) suggest that human settlement was restricted to major watercourses or areas of mineral soil, avoiding the waterlogged grounds. In Southeast Asia, agriculture was dominated by swidden-systems employing small-scale slash and burn to temporarily clear land for the production of short-term crops, and the forest landscape was manipulated under longer-term husbandry for the provision of forest resources. Such systems were governed by diverse institutional arrangements concerning land management and ownership. Typically (though not exclusively) these systems involved a combination of activities including the production of crops for subsistence and local trading, the gathering and use of non-timber forest products, hunting and/or fishing (Barker et al., 2007; Posa et al., 2011; Thornton et al., 2018). These traditional uses of peatlands were generally facilitated by water transport due to the extensive network of rivers, lakes, and later canals found throughout Southeast Asian tropical peatland mosaic landscapes (Chokkalingam et al., 2005). It is only within the last millennium that fossil charcoal, indicating past vegetation burning, has been found more commonly in cores taken from inside peatlands at a notable distance from water transport routes (for example multiple sites in Borneo: Anshari et al., 2001; Hope et al., 2005; Cole et al., 2019). This limited evidence of burning within peatlands, prior to the last 1000 years, or much more recently in some locations (for example Cole et al., 2015), is also true of Central Congo and Amazonian peatlands (Cole et al., 2022).

Taken together, the data demonstrate that humans have been active in peatlands throughout the Holocene, but did not play a dominant role in shaping Earth system processes in the past. Any anthropogenic impacts were geographically scattered, uneven in intensity and temporary in nature, with as a result of a variety of human-peatland relationships in different localities. What is clear is that the impacts of these early human activities are in stark contrast to those observed within the most decades, notably in Southeast Asian peatlands (Cole et al., 2022). These contemporary human activities are inducing change in peatland ecosystems on a transformative scale, typically associated with the anthropocene.

2.3. The increasing scale and pace of anthropogenic activities in peatlands

Within the last 200 years, a series of technological, economic, and society advances and processes have brought about a fundamental change in the relationship between humans and peatlands in some tropical regions. We do not specify a single start date for the anthropocene in tropical peatlands, primarily as anthropogenic activities are dissimilar in their intensity and time scales across geographies. A key feature of this 200-year period has been an increasingly pronounced shift in the geographical and temporal scale and intensity of activities in these environments. This is most pronounced in Southeast Asia; however similar threats exist in other major tropical peatland regions (Dargie et al., 2018; Roucoux et al., 2017). This has led to the overshadowing of traditional, extensive agricultural and other livelihood practices in peatlands, and the adoption of more intensive patterns of agricultural and non-agricultural economic value generation in such areas (Evers et al., 2017).

In Southeast Asian peatlands, a process driving this change in human-peatland interactions has been the incorporation of peatland areas into new, often global supply chains as primary sources of value in the form of agricultural products and raw materials (Evers et al., 2017). These developments have not only been facilitated by a series of technical innovations in engineering, mechanisation, hydro-engineering, plant science and supply chain management, but also cultural, demographic, political and other socio-economic transformations. Collectively, these innovations and transformations have enhanced both the capacity and impetus of human actors to modify peatland landscapes. Chief among these has been the changing relationship between peatlands and local and global economies. In Southeast Asia in the 19th and early 20th centuries, the colonial administrations of the time were responsible for linking national economies into an extractive global network, through the exploitation and sale of each nation's raw materials and agricultural commodities in new markets tied to the colonial system. At this time, peatlands were still relatively unexploited as they were perceived to hold little value, economic or otherwise(Manzo et al., 2020).

In Southeast Asia, particularly Indonesia and Malaysia, new governance regimes emerged in the 1950 s and 1960 s, which embraced a series of development policies informed by anti-colonial and modernisation goals, and more latterly neo-liberalism (O'Reilly, in review). These fuelled an imperative to increase economic output by raising local production for domestic and global markets. Agriculture has been a key sector for development due to a combination of food security issues, and notions of "returning the land to the people" (Manzo et al., 2020; Varkkey and O'Reilly, 2019). Alongside policy changes, innovations in agricultural technologies resulted in the increased availability of affordable and mobile land-moving and clearance equipment which facilitated the deforestation and drainage of peatland. Simultaneously, various crops emerged which could generate a sufficient return to justify the costs associated with peatland conversion, with the most high-profile of these being the oil palm, *Elaeis guineensis* (Evers et al., 2017; Miettinen et al., 2012; Sheil et al., 2009; Wijedasa et al., 2017).

Oil palms were first brought to Java in 1848 as ornamental trees (Henderson and Osborne, 2000), with the first commercial plantations established in peninsular Malaysia by 1917. These remained small in scale and were confined to mineral soils. During the 1950 s and 1960 s the merits of palm oil as a source of vegetable oil became more widely recognised, as did the significantly greater yields produced by oil palms per unit area when compared to than other oil-producing crops (Murphy et al., 2021). Oil palm differs from the other major crops in that it does not grow productively outside of the tropical belt, and remains relatively labour intensive, thereby offering economic benefits to a variety of actors, from landless households through to large private sector entities. By 1966, driven by strong government support, Malaysia dominated the global trade in palm oil (Sheil et al., 2009). With Indonesia subsequently entering the industry, these countries now collectively account for 87% of global production (Koh et al., 2011). The industry is also expanding in Latin America and to a lesser extent Africa, although at present oil palm plantations are found peripheral to peatland areas (Dargie et al., 2018; Roucoux et al., 2017). Prior to the early 2000 s, the majority ($\simeq$ 90%) of oil palm plantation expansion occurred on predominantly mineral soils as opposed to peat, given the relative availability of land over mineral soils deemed underutilised until this point. Only $\simeq$ 6% of the peatlands in Peninsular Malaysia, Sumatra and Borneo were converted to plantations during this initial period of palm oil expansion (Fig. 2) (Koh et al., 2011). However, the increasingly limited supply of suitable and cheap land in Malaysia and Indonesia prompted a move onto peatlands, encouraged by a widespread perception of these ecosystems as 'empty' or waste lands (Manzo et al., 2020).

Between 1990 and 2010, oil palm plantation expansion accounted for over 10% of deforestation of Indonesian and Malaysian peatlands (Koh et al., 2011). Sumatra lost 41.3%, and Borneo 24.8% of peatland forest cover, the equivalent of 2.25% loss per year between 2000 and 2010 (Miettinen et al., 2011a, 2011b). By 2010, 5.1 Mha of forested peatlands were lost in Malaysia, with only 10% of the remaining peatlands considered to be in pristine condition, with plantations covering 2 Mha, equivalent to 15% of the total peatland area (Miettinen and Liew, 2010). In Indonesia, 1.2 Mha of oil palm plantations, equivalent to 14.6% of the total, were established on peatlands by 2012 (Miettinen et al., 2012; Oktarita et al., 2017). Expansion of palm oil is not limited alone to Malaysia and Indonesia in the region. Sristhunthon & Chawchai (2020) report the expansion of oil palm plantation in Southern Thailand also led to conversion of peatlands and mangroves (Srisunthon and Chawchai, 2020). Oil palm plantation expansion is also an imminent threat for mangroves in Indonesia and Malaysia, accounting for around 15% and 40%, respectively, of mangrove loss between 2000 and 2012 (Richards and Freiss, 2015).

Given the scale of impact of oil palm agriculture on tropical forest ecosystems on both peat and mineral soil it is unsurprising that palm oil has become a controversial commodity. Its overall role in tropical deforestation is disputed, as are the relative costs and benefits associated with palm oil production (Evers et al., 2017; Wijedasa et al., 2017). However, there is little doubt that it has played a key role in the acceleration of anthropogenic impacts on tropical peatlands both through direct conversion but also indirectly, by opening up previously peripheral land for development, stimulating ancillary activities, and increasing fire risk (Page and Hooijer, 2016; Evers et al., 2017; Wijedasa et al., 2017). In Indonesia in particular, substantial areas of peat swamp forest have been converted into land suitable for the cultivation of fast-growing timber trees, primarily *Acacia spp.*, for use in the paper pulp industry (Miettinen et al., 2012). Policies in many countries with tropical peatlands have encouraged the expansion of a range of intensive smallholder and commercial agriculture to bolster food security and meet economic and social policy goals. However, it remains the case that oil palm cultivation is a highly significant driver of human-induced modification of peatlands in Indonesia and Malaysia (Evers et al., 2017; Wijedasa et al., 2017).

Other significant peatland regions within tropical latitudes, such as in the Cuvette Centrale in Central Africa (Dargie et al., 2018) and in South America (Roucoux et al., 2017), have not as yet been significantly impacted by industrial agricultural practices. However, there is evidence that the experience of peatland exploitation in Southeast Asia is informing management and development strategies elsewhere. Peatlands and wetlands have been drained and converted in Colombia, the world's third largest producer of palm oil, and there has been significant expansion of palm oil production in Peru, raising concerns about vulnerable peatlands (Abram et al., 2014; Potter, 2015; Roucoux et al., 2017). Peatland drainage in Central America, particularly, Panama, has even longer history than that of Malaysian and Indonesian peatlands, including drainage in the early 20th Century for banana plantations, although such effectives were comparatively smaller in scale (Aronson et al., 2014) The degree to which peatlands are commercially exploited is greatly influenced by the proximity to or ease of accessing potential markets for peatland-based products. This can encourage the expansion

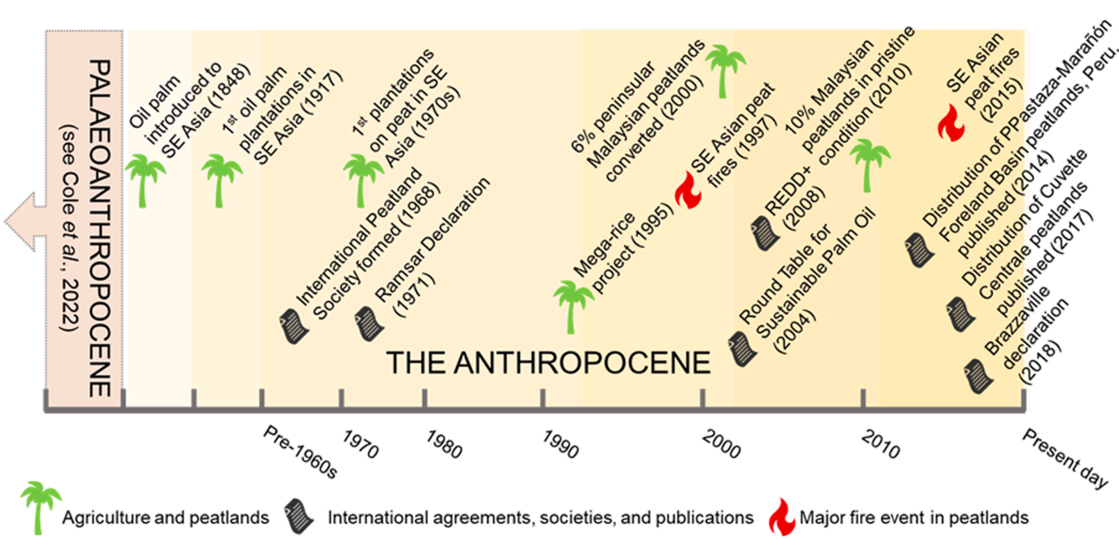

Fig. 2. Timeline of some key events in tropical peatlands in the anthropocene. For events in the palaeoanthropocene (the period of comparatively low anthropogenic impacts in tropical peatlands, preceding transformative change, occurring prior to c. 200 years ago), see Cole et al., 2022.

of road networks into peatlands, which can often be the precursor to wider changes in peatland use and condition (Roucoux et al., 2017; Lilleskov et al., 2019), as has been demonstrated in tropical forests more generally (Laurance et al., 2009). Anthropogenic drivers of transformative change in tropical peatlands are not solely limited to agriculture. Globally, the list of factors includes: the granting of logging, oil and gas concessions, the expansion of urban settlements and the unsustainable exploitation of forest products all within peatland areas (Gutiérrez-Vélez et al., 2011; Janovec et al., 2013; Marshall et al., 2018; Roucoux et al., 2017; Wijedasa et al., 2017). Concerns over the collective and adverse impacts of practices have contributed to a large and growing literature which discusses the need for sustainable peatland management, for restoring degraded peatlands and for prioritising the protection of those that remain intact (Dhandapani et al., 2020; Evers et al., 2017; Wijedasa et al., 2017).

3. Anthropogenic processes in peatlands

3.1. Direct and indirect contemporary anthropogenic processes

The relationships between people and peatlands are continually evolving. Until relatively recently, most direct anthropogenic processes were generally limited due to the challenging nature of the peatland terrain and the geographically peripheral position that peatlands occupied in relation to large-scale human resource exploitation. However, this has been changed by a combination of physical, technical and socio-cultural processes which have enabled easier exploitation of peatlands, and an intensification of anthropogenic processes (Evers et al., 2017; Cooper et al., 2019). Peatlands are also indirectly affected by human activities outside of peatland areas which drive large-scale environmental changes. Examples include increases in atmospheric concentrations of CO_2 and, which in turn is driving climate change (for example alterations in precipitation and climate warming) (IPCC, 2014). Here, we identify the key anthropogenic processes that are driving change in tropical peatlands, discuss the processes of transformation, and in which geographies the processes are most pronounced.

3.2. Land use change

Land use change has transformed peatland landscapes in Southeast Asia in recent decades. Conversion of forested peatlands to agricultural or other land uses generally involves a series of specific activities, resulting in a continuum of degradation. First, peat swamp forests are drained by digging drainage canals of various sizes. Lowering the water table is essential to create the conditions in which oil palms and other popular crops can grow, with most contemporary and commercial agricultural systems being developed for non-waterlogged areas. The drainage process fundamentally changes the hydrology of peatland areas, leading to the drying of the upper layers of the peat substrate and subsequently surface subsidence as a result of rapid aerobic microbial decomposition of peat (Anshari et al., 2010). This contributes to large-scale GHG emissions, changes in microbial communities and increased vulnerability to fire. Second, deforestation removes the natural surface vegetation, which in turn removes plant roots that provide structural support that enhances the accumulation and protection of the peat (Dommain et al., 2010). Deforestation can be mechanical but can also involve fire for rapid and less laborious clearance of vegetation (Purnomo et al., 2017). Due to low bulk density, prior to planting, the surface can be mechanically compacted to improve plant growth, driving further peat shrinkage. The peatland is then planted with new species, frequently with monocultures (Gaveau et al., 2014). Changes in surface vegetation consequently results in changes in root and litter inputs, further affecting the quality of peat carbon and pore size within the physical structure (Yule and Gomez, 2009). Peatlands are not only converted for agricultural expansion, but also for urban development, with various ground treatment works required for stabilisation of the ground prior to construction, although these are not always effective in preventing subsidence (Marshall et al., 2018).

Oil and gas activities also directly result in deforestation, hydrological damage, and landscape degradation through the development of new infrastructure. A recent assessment by Lawson et al. (2022) estimates 8.3% of all tropical peatlands have a potential overlap with a 30 km buffer zone around oil and gas infrastructure, including in Indonesia, the Niger Delta, and he Putumayo-Oriente-Marañón Basin. Oil spills frequently occur around pipelines and shipping routes and can have multiple adverse impacts on tropical peatland ecology, for example through direct toxicity for flora and fauna (Lawson et al., 2022).

3.3. Fire

Fire can be a natural process that influences peatland dynamics, particularly during periods of climatic drying and localised drought (Cole et al., 2022). But although the vegetation in intact tropical peatlands can burn, the ecosystem as a whole is naturally fire-resistant because of the waterlogged soils and moist below-canopy microclimate (Cooper et al., 2019; Turetsky et al., 2015). However, humans have substantially altered fire regimes in Southeast Asia, with often severe consequences for ecosystem function, including carbon storage (Shlisky et al., 2009). In Indonesia, small-scale land management fires have been used for centuries (Simorangkir, 2007). However, before the Colonial era, even dry periods in Indonesia and Malaysia likely only saw limited scale and intensity of forest and land fires due to the intrinsically fire-resistant properties of pristine tropical peat swamps (Turetsky et al., 2015; Page and Hooijer, 2016), and these had minimal consequences for the local environment (e.g., Hapsari et al., 2017; Cole et al., 2019). Since the 1980 s, the frequency and severity of fire events has sharply increased, especially on the islands of Sumatra and Borneo (Chokkalingam et al., 2005). Some of the most extensive fires over the past 25 years in Southeast Asia have been located in drained or deforested tropical peatland areas (Page and Hooijer, 2016). By comparison, carbon losses due to fires during the early Holocene did not exceed annual sequestration rates in accumulating peatlands (Dommain et al., 2014). Increased burning events also drive negative feedback loops, with smog reducing photosynthesis and hence potentially vegetation inputs to peatlands. In addition, smoke plumes have also been reported to increase peat oxidative losses (Hirano et al., 2012).

The majority of contemporary fires in tropical peatlands are started by people, either deliberately or accidentally. Fire ignition density in oil palm concession (0.055 ignitions km^2) is comparable to non-forest areas (0.060 ignitions km^{-2}) but both are approximately ten times ignition density of forests (0.006 ignitions km^{-2}) (Cattau et al., 2016a). Most fires started within oil palm concessions and which occur in relatively close proximity to settlements stay within those boundaries, with only 1–2% having a wider impact on the landscape (Cattau et al., 2016a). Fire is used as method to clear and maintain land for agricultural and plantation development, by both small-scale agricultural practitioners and companies (Simorangkir, 2007). It is also used by communities to improve access to locations within peat swamps where resources are available from fishing and hunting, and to decrease the occurrence of pests and diseases (Chokkalingam et al., 2005). The peatland 'fire problem' is therefore highly complex and evades simple policy solutions (Jefferson et al., 2020).

There are no known studies exploring recent fire incidence in or impacts on the peatlands of lowland Peruvian Amazonia or the Central Congo Basin, which may reflect a lack of quantification of such processes or their relatively low exposure to transformative anthropogenic activities that these peatland ecosystems are experiencing compared with those in Southeast Asia. Future climatic changes in Amazonia and Central African Basin, coupled with the additional threats of, for example, expanding industrial agriculture, road infrastructure and mining, may result in fire becoming a driver of change in these geographies as well (Cole et al., 2022).

3.4. Atmospheric carbon dioxide

Rising atmospheric CO_2 concentrations, driven by ongoing anthropogenic emissions from a range of processes, are likely to have an indirect impact on tropical peatlands, although precise effects, particularly in terms of the effects on plant community structure and abundance, and ongoing carbon sequestration, are uncertain. In an extrapolation from Free Air Carbon Dioxide Enrichment (FACE) studies in temperate forests and plantations, Malhi (2012), proposes an 18% increase in tropical forest productivity relative to pre-industrial values. Other evidence from tropical forests suggests an alteration in species composition, as some plant functional groups are better able to exploit elevated atmospheric CO_2 (Lewis et al., 2009). In tropical peatlands, these collective changes in productivity and ecosystem turnover would manifest in increased aboveground biomass and vegetation inputs to the peat substrate, which could result in higher peat accumulation rates and thus slow increases in atmospheric CO_2 (Malhi, 2010; Malhi et al., 2014). The expansion of boreal peatlands 12,000 – 8000 years ago likely contributed to a sustained peak in atmospheric CH_4 and slight decline in CO_2 (MacDonald et al., 2006). This matches similar evidence of high carbon accumulation rates in peatlands in Kalimantan, Indonesia, between 9530 and 8590 years ago (Page et al., 2004b). Changes in inputs would also directly alter peat GHG fluxes, as changes in plant species composition alters both the composition and concentration of plant root exudate profiles, affecting CO_2 and CH_4 efflux (Girkin et al., 2018a, 2018b).

3.5. Climate change

Rising atmospheric concentrations of greenhouse gases are driving climate change, including changes in precipitation and temperature, with potentially significant impacts on tropical peatlands globally (Loisel et al., 2021). The main threat climate change poses to peatlands comes from the increased duration or greater severity of drought, which drives rapid degradation of peat through oxidation, or increased likelihood of fire events (Loisel et al., 2020). This can result in substantial peat carbon loss, or prolonged periods when peat is not accumulating, as evidenced in the Central African peatland complex (Garcin et al., 2022). Under most future climate change scenarios, there will be strong increases in rainfall seasonality and intensity (IPCC, 2013; Li et al., 2007), but precise effects on peatlands will differ between regions. Our lack of ability to predict more exact future impacts on peatlands is mostly due to substantial uncertainty in regional precipitation models, and limited measurements of peatland hydrology (Baird et al., 2017). Peatlands in Southeast Asia are generally thought to be most at risk from future climate change due to the influence of El Niño on exacerbating drought events (Roucoux et al., 2017; Kondo et al., 2018). In contrast, climate models typically predict higher precipitation for peatlands in western Amazonia, meaning that even under conditions of higher temperatures, the water balance of these peatlands might avoid being severely affected by climate change in the coming decades (Roucoux et al., 2017).

Sea level changes are likely to affect mangroves over peat including reductions in total area, structural complexity, and functionality (Alongi, 2015).

Temperature changes will also have indirect effects on plant productivity and decomposition rates. Predictions for the tropics are more certain for temperature than for precipitation, and include approximately 4 °C higher mean annual temperatures for Southeast Asia and Amazonia, and potentially up to 6 °C for Central Africa under RCP8.5 scenarios (high emissions, "business-as-usual" climate scenarios) (IPCC, 2013). Predicting regional responses to global heating and the consequences for tropical peatlands is complex, and requires models which can account for large-scale atmospheric processes (e.g., the El Niño-Southern Oscillation (ENSO) and the Indian Ocean Dipole (IOD)). Moreover, different phases of these cycles affect continents at different times and by contrasting extents. The ENSO and IOD can interact in their effects: all of the major Southeast Asian atmospheric pollution events from 1960 to 2006 happened during years of drought induced by ENSO and/or IOD conditions (Gaveau et al., 2014). For example, the 1997–98 fires happened over a period that saw the strongest recorded ENSO and IOD events in the 20th century (Gaveau et al., 2014). With climate change, the frequency, duration and intensity of ENSO events have already increased since 1976 (Aiken, 2004).

4. Anthropogenic impacts

4.1. Tropical peatland ecosystem services

A range of anthropogenic activities have had serious consequences for ecosystem services provided by tropical peatlands, and many of these activities have intensified in recent decades. At present, exposure to anthropogenic impacts is not equal across tropical peatlands globally, with the most destructive impacts experienced in Southeast Asia ecosystems. However, there are drivers of change, for example, in the climate, that will have consequences for tropical peatlands everywhere. Many of these services underpin the livelihoods of communities living in and around tropical peatlands. For example, in Southeast Asia, communities living in peatlands are have diversified their sources of income (ranging from small-scale agricultural production to fishing in rivers), and it is also common to adopt livelihood strategies that involve multiple sources of income and food, which are flexible in relation to seasonal changes and demand for various resources (Thornton et al., 2020). Documented livelihood strategies in Southeast Asian and Amazonian peat-forming landscapes include farming, charcoal production, birds' nest farming, hunting, fishing, gathering non-timber forest products such as fruits and fibres, small-scale logging and employment in construction, oil extraction and other industries (Arce-Nazario, 2007; Chokkalingam et al., 2005; Thornton, 2017; Roucoux et al., 2017). These strategies would be variously impacted by transformative change in peatland ecosystems, with some more vulnerable to others. Here we discuss the impacts of intensive anthropogenic activities on fundamental ecosystem processes in tropical peatlands, drawing principally on evidence from Southeast Asian peatlands, which have experienced the most significant changes in the anthropocene. We discuss the underlying mechanisms of change and identify possible feedback pathways, as well as discussing potential impacts on local communities.

4.2. Carbon and nutrient cycling

Tropical peatlands store a huge volume of carbon, recently estimated at 129.8 Gt C (maximum value from Dargie et al., 2017) (Fig. 3). Carbon storage and sequestration can vary across vegetation types. For example, mixed forest over peat (including canopy palms and tropical hardwood trees) in Panama had mean carbon storage of 1884 Mg C ha^{-1}, followed by mangroves (*Rhizophora mangle*, 1771 Mg C ha^{-1}), broadleaved evergreen trees (1694 Mg C ha−1), and Cyperus (sawgrass) bog plain (1488 Mg C ha^{-1}), figures broadly comparable to carbon storage across other tropical peatland vegetation (Murdiyarso et al., 2009; Draper et al., 2014; Crezee et al., 2022). Localised processes, for example litter inputs, root turnover, and specific management practices, as well as variation in peat depth can result in a high degree of heterogeneity of peatland carbon across short distances (Dhandapani et al., 2021; Girkin et al., 2019). The protection of this carbon store is of critical importance for climate change mitigation (Leifeld and Menichetti, 2018; Page et al., 2011). In the absence of human intervention peatlands accumulate carbon at rates of 1 – 2 mm yr^{-1} (Page et al., 2004a). However, the extent of carbon storage varies spatially with both depth and dominant vegetation type within peatland sites, and in many regions remains poorly quantified due to the remote nature of many tropical peatland landscapes and logistical challenges of undertaking measurements (Lawson et al., 2015; Upton et al., 2018; Coronado et al., 2021).

Carbon is lost from peatlands as a result of land use conversion and

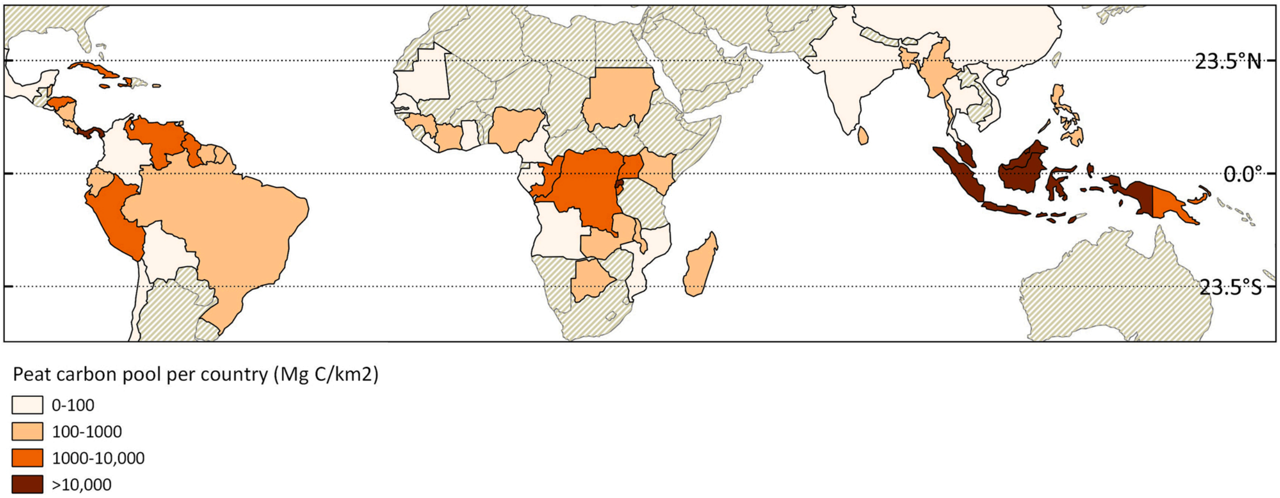

Fig. 3. Estimated density of the peat carbon pool (Mg C km^{-2}) for countries between the tropical latitudes for which data are available (grey dashing represents a lack of data at present). (Data from Dargie et al., 2017 and Page et al., 2011).

other peatland management practices (Fig. 4). Estimated carbon losses over the first 25 years after the establishment of an oil palm plantation on a tropical peatland are substantial. The conversion of one hectare of forest on peat releases over 1300 Mg carbon dioxide equivalents during the first 25-year cycle of oil palm growth (Germer & Sauerborn, 2015), with similar results reported for the clearance of mangroves over peatland in Belize (Lovelock et al., 2011). Net losses over 25 years have been estimated at up to 405 Mg C ha^{-1} in a peatland converted to *Acacia* plantation in Sumatra, Indonesia (Jauhiainen et al., 2012; Murdiyarso et al., 2010). Estimates for the time required for oil palm plantations to offset the net carbon balance extend up to 600 years for plantations developed on peat (Danielsen et al., 2009; Gibbs et al., 2008). However, the financial and agricultural viability of oil palm on peat is substantially shorter than this, as yields halve within 100 years due to impacts of subsidence, which increases the vulnerability of peatlands to flooding and fire events (Sumarga et al., 2016).

Climate change has strong feedbacks on peat carbon stocks, carbon accumulation, and nutrient cycling within peatland ecosystems, particularly through the high temperature sensitivity of greenhouse gas emissions from tropical peat. For example, the temperature sensitivity (Q_{10}) for CH_4 fluxes from a tropical peatland under anaerobic conditions was 6.8, implying a 10 °C rise in temperature would drive a near sevenfold increase in emissions, although values for CO_2 were lower ($Q_{10} <$ 2), with the overall response in emissions to increasing temperature demonstrating an exponential relationship (Sjögersten et al., 2018). Reduced temperature responses have been reported, though in converted sites in Malaysia, with differences likely driven by the large-scale depletion of labile carbon as a result of peatland drainage to enable agriculture (Cooper et al., 2019; Girkin et al., 2020a). Accelerated emissions of greenhouse gases, particularly those with high global

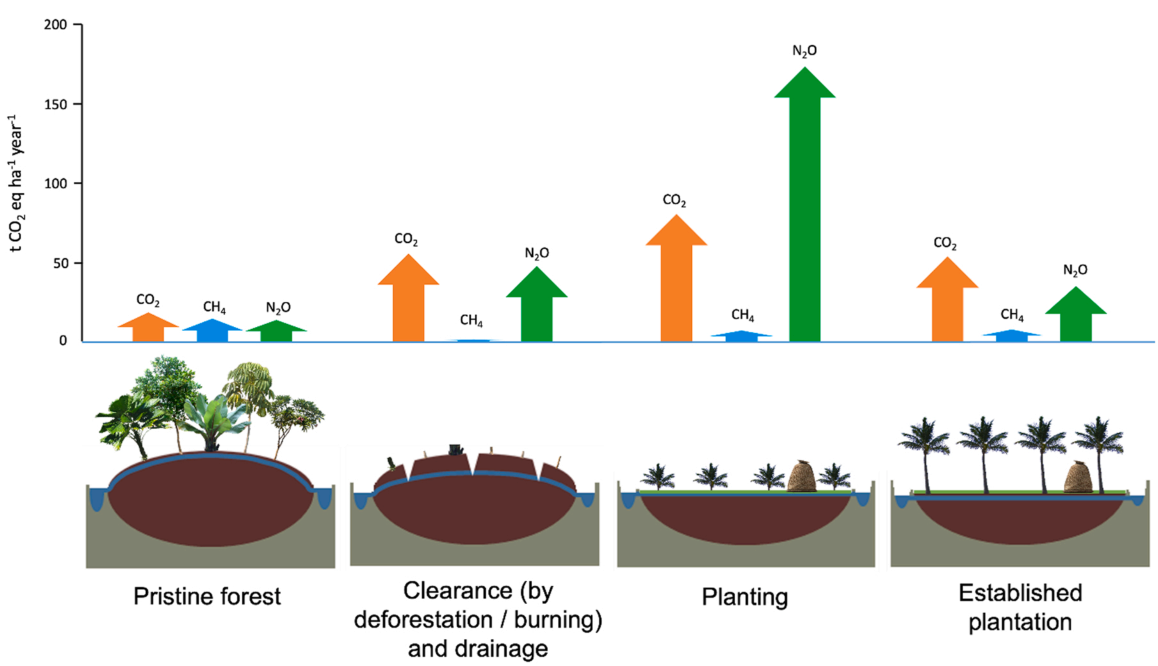

Fig. 4. Four stages of conversion of pristine tropical peatlands to agriculture, with changes in emissions factors, water table, peat depth, peat dome shape, nutrient inputs (fertiliser application) and subsidence and peat compaction. Arrow heights representing emissions factors are proportional to data from Cooper et al. (2020) which focussed on a land use and management gradient in Malaysian peatlands. Emissions factors are CO_2 equivalents (accounting for global warming potential of each gas). CO_2 measurements represent heterotrophic respiration only.

warming potentials (i.e., CH_4 and N_2O), may provide an additional positive feedback mechanism, exacerbating ongoing climate change (Turetsky et al., 2015).

Significant carbon losses from peat oxidation are not, however, only associated with large-scale plantations but also from smaller-scale agricultural activity. While IPCC emissions factors for oxidative peat carbon losses are 55 Mg CO_2 ha^{-1} yr^{-1} for commercial plantations, losses from smallholder plantations are only marginally lower (51 Mg CO_2 ha^{-1} yr^{-1}) (IPCC 2014). Emissions factors for disturbed secondary forests on peat are 80% lower (Khasanah and van Noordwijk, 2018). Smaller-scale agricultural activities are estimated to have contributed 34% of cumulative carbon emissions, compared with 44% from large-scale commercial plantations and 22% from disturbed forests, and are amongst the main contributors to peatland fires (Miettinen et al., 2017). Following conversion, losses through the leaching of labile carbon from the substrate increase by up to 200% (Rixen et al., 2016). While direct losses from conversion can be substantial, changes in the dominant vegetation type can impact future carbon fluxes and storage through additional mechanisms, for example, through the alteration of organic matter properties (Tonks et al., 2017). Shifts in plant community composition can also indirectly affect carbon fluxes through altering the species-specific inputs of root oxygen (Girkin et al., 2020b), and exudates that can accelerate peat decomposition (Girkin et al., 2018a, 2018b; Girkin et al., 2020a; b). Collectively, these changes in peatland condition have resulted in Southeast Asian tropical peatlands alone contributing to 3% of global anthropogenic greenhouse gas emissions (Ballhorn et al., 2009; Posa et al., 2011; van der Werf et al., 2009). This underlies the importance of preserving the remaining hydrologically-intact peatlands in Southeast Asia and other regions.

4.3. Subsidence

Subsidence of converted tropical peatlands is closely associated with carbon losses, saltwater intrusion, and a decrease in the freshwater reservoir, and is an irreversible anthropogenic impact on tropical peatlands (Saputra, 2019). Rates are linearly associated with mean annual water table height (Couwenberg et al., 2010; Hooijer et al., 2010), therefore rapid drainage for construction, agricultural or logging purposes instigates rapid initial subsidence. Couwenberg et al. (2010) report a rate of 0.9 cm yr^1 per 10 cm of drainage, higher than rates for temperate regions (Wösten et al., 1997). Rates are greater closer to drainage ditches and canals (Sinclair et al., 2020). Whilst the type of land use has no significant impact, mean water table depth is the best predictor of subsidence rates across all land uses (Evans et al., 2019). The collapse of large pore structures is a result of an initial rapid consolidation component of subsidence, where saturated peat is compressed by surface peat that is no longer buoyant above the water table. This is followed by a gradual transition to a slow oxidation component, where decomposition of organic matter results in CO_2 release, and a slow shrinkage component where bulk density further increases, leading to irreversible water storage potential loss (Wösten et al., 1997). The slower and long-term processes of oxidation and shrinkage result in a reduction in subsidence rates over time after a drainage event and with a shallower peat depth; five years after drainage, rates have been reported as high as 150 cm, which reduce to 3 – 5 cm in subsequent years (Hooijer et al., 2012). Despite a reduction in subsidence rates over time, it is an irreversible process that can only be halted by rewetting the peat, i.e., returning it to a waterlogged state. Physical structure can only be restored after decades of accumulating native woody fibrous biomass deposits, resulting in a legacy of compromised hydrological functioning. Without the raising of water tables, continued subsidence will lead to the loss of plantation areas and peatlands altogether (Hooijer et al., 2012).

4.4. Smog

Peatland fires have compound consequences, resulting in extreme toxic smog events, chronic health infringements, economic losses, and significant greenhouse gas emissions (Page and Hooijer, 2016). Peat predominantly burns through smouldering fires, emitting GHGs alongside toxic and carcinogenic compounds, including formaldehyde, hydrogen cyanide, ammonia, benzene, polycyclic aromatic hydrocarbons (PAHs) and high levels of small particulate soot particles (PM2.5 – particulate matter with a diameter of less than 2.5 μm) (Rein et al., 2008). In combination, emissions from these fires forms a dense smoke that poses a significant health risk. This includes negative impacts on cardiac and pulmonary health, as has also been documented in temperate peatland regions including the UK and US (Graham et al., 2020; Rappold et al., 2011), further illustrating that this is a global issue. The low temperatures associated with smouldering combustion also result in weakly buoyant smoke plumes that can be blown at low levels into local communities, lowering air quality (Santoso et al., 2019). In 2015, the smog from the peatland fires in Indonesia spread across Indonesia, Malaysia, Singapore, Brunei and southern Thailand, Vietnam and the Philippines; illustrating how peatland fires and their resulting smog are a transboundary and international issue. The impacts of smoke are severe and far-reaching: Crippa et al. (2016) found that high particulate matter concentrations from the 2015 fires in Indonesia exposed 69 million people to unhealthy air quality conditions, with short-term exposure to this pollution potentially causing almost 12,000 excess mortalities (Crippa et al., 2016). Koplitz et al. (2016) estimated that the 2015 fires resulted in over 100,000 premature deaths across the affected regions (Koplitz et al., 2016). Peatland fires and atmospheric pollution now recur in Indonesia on a near-annual basis (Herawati and Santoso, 2011) with potentially severe long-term health consequences: for example, smog exposure during the prenatal period has been linked to decreased adult height attainment (Tan-Soo and Pattanayak, 2019). There is also evidence to suggest that people exposed to this smog are more susceptible to respiratory diseases, such as COVID-19, one of the many reasons why tropical peatland conservation is seen as a positive intervention for mitigation against current and future pandemics (Harrison et al., 2020a, 2020b).

Multiple studies demonstrate the negative impacts of smog on wildlife. Just as humans suffer respiratory symptoms and ailments, so will primates, with negative impacts on both fitness and mortality (Cheyne, 2008). Tan et al. (2018) also found that smog has significant negative impacts on butterfly (specifically *Bicyclus anynana*) development and survival. Insects are important members of food webs, and these impacts are likely to reverberate through the ecosystem. Smog affects ecological processes, for example driving a dramatic increase in forest litter-fall (Harrison et al., 2016). Periods of elevated atmospheric pollution have been shown to reduce photosynthesis (Davies and Unam, 1999), and also affect plant investment in reproduction (i.e. reduced availability of fruit and flower foods for forest animals (Harrison et al., 2016).

Smog also impacts livelihoods. For example, in Central Kalimantan, as is common across Southeast Asia, the way in which houses are commonly constructed makes it impossible to keep the toxic smoke from entering inside the home (Thornton, 2017). Severe atmospheric pollution forces villagers to stay at home, making them unable to work or attend school, impacting incomes and livelihoods (Thornton, 2017). As smog reduces solar irradiance, it in turn reduces the yields of agricultural crops (Tacconi, 2016). Carmenta et al. (2017) found that smallholder farmers face a loss of income from fire, but also unfair blame, as they are often accused of starting them.

4.5. Water resources and management

Intact peat absorbs water rapidly during wet periods and releases it slowly during dry periods. Thus, intact peatlands play an important role in flood mitigation, and in preventing loss of life and damage to infrastructure. Peatland drainage, as has been occurring in Southeast Asian peatlands on a large scale, increases flood risk in wet periods as dried out and compressed peat is increasingly hydrophobic and so cannot take up the excess water (van Beukering et al., 2008). Drainage also leads to peat decomposition and land surface subsidence, which can increase the risk of semi-permanent or permanent flooding (Page et al., 2009; Giesen, 2015). Progressive subsidence means that production cannot be maintained in the medium and long term, even with currently promoted best management practices (Sumarga et al., 2016). For example, peatlands in Central Kalimantan, Indonesia, are only a few meters above high tide sea and river water levels; severe flood risks in the lowest parts of drained peatlands in the region could occur within decades (Wosten et al., 2008), with most current agricultural crops unsuitable for growing in waterlogged conditions (Sumarga et al., 2016). Increased flooding has significant negative impacts on local livelihoods. For example, in parts of the Mega Rice Project area, in Central Kalimantan, communities are facing decreased fish catches and incomes (due to habitat degradation, water pollution, overfishing etc.), but are unable to switch to farming as a reliable source of income due to the now regular, yet historically unprecedented, flooding that occurs in the area (Thornton, 2017).

4.6. Biodiversity

While much of the focus on tropical peatlands is on their importance as a carbon store, they are also significant reservoirs of biodiversity. They have much higher floral diversity than northern peatlands, which tend to be dominated by mosses, grasses and sedges. Individual peatland sites can display significant heterogeneity in plant species composition. Intact tropical peat domes, commonly found in low-lying coastal areas of insular Southeast Asia, tend to display concentric rings of distinct, 'phasic' vegetation, with the composition of each floral community related to the thickness of the peat layer, availability of nutrients and other physical conditions associated with the developmental stage of the dome (Anderson, 1964). The Changuinola peat deposit in Panama features seven distinct plant phasic communities arrayed along a nutrient gradient to the centre of the peat dome, ranging from mangroves through to sawgrass species (Phillips et al., 1997). Similar gradients are reported in peatlands in Brunei Darussalam, Borneo, and Central Kalimantan (Gandois et al., 2013; Poesie et al., 2010).

An assessment of flora in a Southeast Asian peatland indicated that 534 identified species had a potential use to local communities, with 222 known to be used for medicinal purposes, 165 for food, and 165 for other uses, including as a fuel or a dye, with many species having multiple potential applications (Giesen, 2015). An assessment of biodiversity in a peat-forming mangrove in Botum Sakor National Park, Cambodia identified 26 plant species (Lo et al., 2018).

Various uses of peatland flora by indigenous communities in Peruvian Amazonia have also been documented (Schulz et al., 2019a; b). This demonstrates that peatland biodiversity can be crucial for sustaining local livelihoods. The extent and inaccessibility of the peatlands of Central Africa has limited the number of assessments of floral diversity, although some limited classifications are made in Lebrun and Gilbert (1954) and Evrard (1968). Characteristic peat swamp species include *Symphonia gobulifera* L., *Entandrophragma palustre* Staner, the canopy palm *Raphia laurentii* de Wild., and various rattan species (Biddulph et al., 2021).

Limited data is available on fauna in tropical peatlands but there are several indications that peat swamp forests can act as important refuges for species threatened in other forested ecosystems. For example, a faunal biodiversity assessment for peatlands in Southeast Asia determined that 45% of mammals and 33% of birds found in them were either near threatened, vulnerable, or endangered (Posa et al., 2011). Tropical peatlands in Indonesia and Malaysia are particularly known as important habitats for orangutans (*Pongo pygmaeus* Linnaeus), which are now listed as Critically Endangered (Harrison and Rieley, 2018). Approximately 11% of species identified within tropical peatlands are endemic to these ecosystems, with the majority comprising trees, epiphytes and climbing vegetation (Posa et al., 2011). Species that indicate 'healthy' tropical peatlands in Southeast Asia include ants (Schreven et al., 2018), and various mammals: leopard cat (*Prionailurus bengalensis*), pig-tailed macaques (*Macaca nemestrina*); sambar deer (*Rusa unicolour*) and clouded leopards (*Neofelis diardi*) (Cheyne et al., 2016). Similarly, information on faunal diversity in the Central African peatland complex is limited.

Anthropogenic impacts on the biodiversity of tropical peatlands include degradation through logging and fire, hunting and the wider impacts of climate change. For orangutans, logging in peatlands in West Kalimantan, Indonesia has been shown to reduce population density by up to 21%, even in the absence of other forms of peatland exploitation, most likely due to changes in forage availability and increased canopy gaps (Felton et al., 2003). Hunting of various peatland fauna has increased as forests in Central Kalimantan, have become more accessible and human populations increase, for example causing declines in large flying foxes, *Pteropus vampyrus natunae* (Struebig et al., 2007). As these flying foxes (otherwise known as bats) play an important role in pollination and seed dispersal, declines in their population will likely have wider impacts on forest regeneration, ecosystem structure and function. In Indonesia, at least 300 bird species are traded, and 22% of households own pet birds originally taken from wild populations, although these are not necessarily species that inhabit peatlands (Harris et al., 2015). While no studies have directly tested whether changes in the species composition of tropical peatlands are occurring as a result of climate change, there is evidence of changes in the floristic and functional composition of intact lowland forests in Amazonia over the last 30 years, which have implicit consequences for this ecosystem, with overall increases in dry-affiliated genera and increased mortality of wet-affiliated genera (Esquivel-Muelbert et al., 2018).

5. The future of tropical peatlands

5.1. Trajectories of change

It is clear that humans have had substantial impacts on the structure and function of tropical peatlands. To date, these anthropogenic impacts have disproportionately affected peatlands in Southeast Asia compared with those in South America and Central Africa. Even in Southeast Asia's degraded peatland ecosystems, there remain opportunities to implement policies and practices that could lead to the conservation of existing areas of intact peatland and the restoration of degraded systems, through creating economic opportunities that enhance the protection and restoration of peatlands by local communities, and that can support the sustainable provision of livelihoods from these ecosystems. However, it is clear that many anthropogenic impacts of recent decades (e.g. subsidence) are irreversible (Wösten et al., 1997), and it is unclear how quickly fundamental ecological functions will recover. Peatlands will also continue to be affected by indirect anthropogenic processes, such as elevated temperatures and changes in precipitation patterns brought about by anthropogenic climate change, affecting the length and severity of drought events. Climate change is likely to lead to a three-fold increase in the number of days of extreme fire danger per annum in fire-prone regions of Indonesia (Herawati and Santoso, 2011). Here, we discuss future strategies for conserving and managing tropical peatlands. We propose that the conservation of existing intact tropical peatlands, and the restoration of degraded sites, in particular by maintaining high water levels, must be a priority. However, any interventions must be combined with promoting the continued socio-economic development of local communities.

5.2. Conservation

There are no published global syntheses of the protection status of tropical peatlands, although some regional and site-specific data are available. For example the Pastaza-Marañón Foreland Basin in Peru (3.56 Mha) has 31% coverage by protected areas (Roucoux et al., 2017), but other significant peat deposits have much lower coverage: 11% for the Central Congo Basin (Dargie et al., 2018), and only 6.9% for Southeast Asia (Posa et al., 2011). Moreover, site designation does not necessarily afford legal or practical protection, although does generally require the development of site inventories and specific management plans, and can therefore be a useful step in establishing further protections (Roucoux et al., 2017).

Crucial to protecting peatlands from the damaging effects of anthropogenic activity is maintaining a watershed as an intact hydrological unit. Private reserves, which are generally small and fragmented, are likely to be insufficient for achieving this (Roucoux et al., 2017). Evidence from Amazonia indicates that even within large protected areas, small-scale agriculture, as well as more damaging activities, can be ongoing (Dourojeanni, 2015). The establishment of large protected areas can be effective in reducing levels of anthropogenic disturbance, but is often less effective in reducing poverty due to difficulties of attracting tourists. This remains the case even when sustainable resource extraction is allowed. (Miranda et al., 2016). The Ramsar Convention on Wetlands, listing wetland sites of international importance, represents one pathway to site protection, with several tropical peatland sites listed, including the well-studied Central American peatlands at Changuinola, Panama, and the trans-national site covering the Central African peat deposit (Ramsar 2021). However, globally less than 10% of total peatland area is included, and it remains unclear the extent to which protections can be enforced (Ramsar, 2014). Encouragingly, it has recently been announced that a large trans-national Ramsar site will be created across the Republic of Congo and Democratic Republic of Congo, covering 12.9 Mha of wetland, and including significant areas of the region's peatlands (Dargie et al., 2017; Ramsar, 2017).

Funding for peatland conservation is available through payments for carbon conservation or avoided emissions, from a range of sources including the UN Green Climate Fund, the UN's REDD+ schemes (Reducing Emissions from Deforestation and Forest Degradation) as well as other public, private and non-governmental conservation initiatives (Murdiyarso et al., 2010). An example of this approach comes from Peruvian Amazonia, where a $9.1 million Green Climate Fund project is underway, with the goals of promoting sustainable alternatives to unsustainable forest management practices and improving livelihoods for communities living in the Datem del Maranon province (encompassing part of the Pastaza-Marañón Foreland Basin peatlands) (Roucoux et al., 2017). The effectiveness of these carbon conservation approaches relies on both a thorough assessment of the local socio-economic and political context, and on accurate maps of peatland distribution and their above and belowground carbon stocks (WRI, 2018). There remain significant criticisms regarding the scale, viability, administration and effectiveness of these schemes (Blais-McPherson and Rudiak-Gould, 2017). In the case of REDD+ projects developed to implement forest conservation and sustainable forest management practices, it has been argued that they may not lead to real carbon emission reductions, since reductions through this mechanism are potentially non-permanent. REDD+ is predominantly focussed on technical solutions, and does not always directly address the underlying political and socio-economic conditions that cause the deforestation. As a result, in Indonesia, REDD+ has not, as yet, been shown to generate transformational changes in tropical forest protection and management (Moeliono et al., 2020). Moreover, REDD+ has been suggested as potentially harmful to local and Indigenous populations, primarily because economic value is assigned to protecting nature, and not the welfare and needs of people within these landscapes (Skutsch and Turnhout, 2020). This can ultimately result in local and Indigenous populations losing access rights to their land (Skutsch and Turnhout, 2020). Often only a small fraction of investment reaches the communities who need it most (Thompson et al., 2011). These negative aspects can be mitigated: for example, the UN Green Climate Fund's investment in the Datem del Maranon region explicitly entrusts Indigenous communities with the management of their resources, and is designed to improve their livelihoods, and address gender inequality (Green Climate Fund, 2015; Roucoux et al., 2017).

Other established routes for peatland protection include transferring land titles to indigenous communities. In this process, communities have their formal property rights recognised, replacing customary land tenure arrangements. In Indonesia, for example, local or customary land tenure is often not formally recognised, land tenure claims often overlap and conflicts are an important driver of fire use in peatland (see Harrison et al., 2020a; b). Resolving land tenure conflicts is therefore seen as a key step towards more sustainable peatland management in Indonesia (Harrison et al., 2020a; b) as will be the case for other tropical peatland areas facing similar challenges. In developing countries, up to one-third of forests are estimated to be managed by local communities (Blackman et al., 2017). There are few estimates for tropical peatlands specifically (Davis and Wali, 1994; Thompson et al., 2011), apart from for the Pastaza-Marañón Foreland Basin in the Peruvian Amazon, where titled land account for only 7%, providing significant opportunity for expansion (Roucoux et al., 2017).

5.3. Management and sustainable development

Despite causing substantial negative impacts on ecosystem function, as reviewed here, the conversion of tropical peatlands from a natural to an intensively modified state can provide economic benefits at national and regional levels (Padfield et al., 2014). Extensive literature is available on the management of agriculture on tropical peatlands (Goldstein, 2015), although much is based on the assumption that continued exploitation of peatlands is of net benefit to local communities, businesses and/or governments, despite extensive evidence of substantial damage from such practices, and substantial declines in yield over time (Evers et al., 2017; Sumarga et al., 2016; Wijedasa et al., 2017). In some cases, responsible or "wise" use is promoted as the governing principle through which peatlands should be managed to satisfy human needs, but this is not necessarily associated with sustainable activities (Joosten and Clarke, 2002).

For those peatlands still in a largely intact state, for example in the Amazon and Central Congo basins, the intentions and practices underlying 'management' must be carefully defined to avoid perverse outcomes. Traditional uses of peatland ecosystems and associated local ecological knowledge must be incorporated into any higher-level decision making to avoid the promotion of activities which may undermine current subsistence practices that allow the peatlands to remain in a hydrologically intact state (for example, Schulz et al., 2019). For example, agricultural credit programs in Peru led to alterations in farm distribution and altered the degree of disturbance of associated forest ecosystems over several decades (Arce-Nazario, 2007). In Central Kalimantan, the ban on fire use in agriculture has had negative impacts on the income and livelihoods of communities who previously used controlled burning for planting rice paddies (Antang & Jaya, personal communication, 2019). Without the use of fire, rice crops have suffered from pests and disease, and the community now has to buy more rice for consumption, which adds further expenditure. There are few studies investigating these issues in Central African peatlands. While there are limited data on community resource use inside natural parks containing peatlands, details on the nature of the resources being used are absent (Biddulph et al., 2021). A number of alternative options have been proposed in the literature, including the undertaking of commercial cultivation of species that occur naturally in peatland areas (such as swamp taro and sago palm), and the introduction of exotic species which grow in waterlogged soils (Tan et al., 2021). However, the extent to which these approaches are capable of supporting the livelihoods of peatland communities while reducing the negative environmental impacts associated with commercial scale agriculture (requiring the drainage of peatlands), has not yet been proven.

For disturbed peatlands that have already been deforested, drained and converted to agricultural use, specific management practices such as burning of tree stumps, peat compaction and fertiliser application vary substantially but generally further degrade peatlands and can substantially increase fire risks (Leng et al., 2018). As a result, adapting management and controlling for hazards is deemed the most responsible strategy (Joosten et al., 2012). Much regulation focuses on water table management from the perspective of either preserving peat carbon (by maximising water table height), reducing fire risk or optimal development of plantation agriculture, which requires water table drawdown of up to 80 cm (Andriesse, 1988; Evers et al., 2017). While regional and national support is essential for implementing widespread water table management, local efforts are also required. A successful example is at Raja Musa Forestry Reserve, Peninsular Malaysia, where The Friends of North Selangor Peat Swamp Forest was established in 2011 to teach community methods for fire control and blocking ditches to maintain high water tables. This rehabilitation program (developed by the Selangor State Forestry Department and the Global Environment Centre) attracted funding from a number of companies, including HSBC Bank and Malaysia Berhad, to maintain ditch blocking efforts (Parish, 2014).

5.4. Restoration

The practicalities of restoring anthropogenically disturbed peatlands have been recently reviewed (Dohong et al., 2018; Giesen and Sari, 2018). In brief, restoring peatland ecological function generally requires raising water tables (Fig. 5). Hydrological restoration is commonly achieved through blocking drainage channels, resulting in re-wetting and flooding of the peatlands (Jaenicke et al., 2010), although the success of these interventions is strongly limited by the physical state of the peat substrate. Effective restoration generally includes the gradual accumulation of new above-ground biomass, with roots driving pore formation and reducing bulk density, although these processes can be slow, or non-existent without human intervention and local community support and involvement (Green and Page, 2017).

Peatland forest canopy cover may gradually be restored through seed germination and re-planting as either an assisted or unassisted process (Dohong et al., 2018), although the effectiveness of different approaches requires further investigation (Giesen and Sari, 2018). Similar approaches may also be applied for restoring peat-forming mangroves (Taillardat et al., 2020). Evidence from a study of a 25-year chronosequence of artificial sub-tropical mangrove forests in Florida indicated the accumulation of a 20 cm deep layer of peat (Osland et al., 2020). Such efforts are also likely to bring wider benefits beyond carbon accumulation, for example through reducing rates of coastal erosion, and the support of a wider body of ecosystem services. Modelled restoration of a 25-year old oil palm plot resulted in emission reductions of 440 – 1200 Mg CO_2 ha^{-1}, underlining the significant benefits of peatland restoration as a climate change mitigation strategy (Warren et al., 2017). Challenges for peatland restoration, however, include: managing ecological factors, such as the emergence of invasive shrub

Fig. 5. (a) Secondary forest in the Raja Musa Forest Reserve (November 2018); (b) Oil palm plantation at Sungai Besar, near the Sungai Karang Forest Reserve, Selangor (February 2020); (c) Replanted *Macaranga* sp. saplings planted in 2017 at the North Selangor Peat Swamp Forest Reserve site, last burned in 2016; (d) Results of canal blocking to raise water tables at Sungai Karang Reserve in North Selangor Peat Swamp Forest. Blocking was first installed in 2010 at a site previously heavily drained by a canal network. All sites are in Peninsula Malaysia. Photos taken by M Ledger.

communities which prevent sapling growth (Page et al., 2009); the encouragement of birds into degraded peatlands to disperse tree seeds (Graham and Page, 2012); preventing continued impacts from fire; maintaining support from local communities; and developing site-specific restoration plans (Dohong et al., 2018). Full recovery of peat carbon stocks will take substantially longer than it takes to regenerate a tropical forest (Cole et al., 2014; Jones and Schmitz, 2009) due to the slow rates of peat accumulation (1 – 2 mm year^{-1}), even for ecosystems with high rates of primary productivity (Dargie et al., 2017; Upton et al., 2017).

Effective policies are critical for peatland restoration and for protecting existing habitats. The Global Peatland Initiative, launched in 2016, represents an approach targeted at not only assessing peatland extent and carbon stocks, but also planning pilot projects to support a transition to sustainable peatland use in its partner countries (Indonesia, Peru, and the Republic of Congo) (UNEP, 2016). In 2018, the Brazzaville Declaration was signed by the Democratic Republic of Congo, the Republic of Congo, and Indonesia, and committed these countries to the conservation, restoration and sustainable management of the Cuvette Centrale peatland (Kopansky, 2018). This is particularly important in ensuring that this vast peatland complex does not follow the same trajectory as those in Southeast Asia (Fig. 2).

6. Conclusions

Anthropogenic processes and impacts have and are occurring asynchronously in time and space across the major tropical peatland regions. While humans have been active in and around tropical peatland ecosystems throughout the Holocene, anthropogenic impacts have intensified in the recent past, particularly in the last 200 years, and with the most significant consequences to peatlands in Southeast Asia. In large part, these processes have been driven by a narrative of development, but as is made clear, this has not been achieved sustainably. Tropical peatlands globally are also affected indirectly by anthropogenically-driven climate change, particularly changes in precipitation and temperature. Based on climate models for the region, these processes are likely to intensify. These direct and indirect processes affect ecosystem services that peatlands provide, including their function as a carbon store, as a reservoir of biodiversity, and as a source of the provision of livelihoods to local communities.

Locally appropriate and evidence-based policy, effective management, legal protection and enforcement of environmental regulations remain critical to prevent the loss of peatland ecosystem function caused by unsustainable peatland use. Some cause remains for optimism for the future of tropical peatlands. International initiatives, for example the Brazzaville Declaration, may encourage future decision makers to learn from the unsustainable pathway taken across Southeast Asia and to choose an alternative, sustainable route to development. Restoration of degraded peatlands is possible, but on long timescales that exceed human lifespans. Emphasis must remain on the protection and conservation of tropical peatland ecosystems, as well as research on their sustainability, to maintain their globally important ecological and societal functions.

We recommend several areas of future research to address key outstanding gaps in knowledge we have identified through our review. First, more information is needed on past and contemporary carbon and nutrient dynamics of tropical peatlands, particularly in relatively understudied peatlands outside of Southeast Asia (e.g. in Central Africa and regions of the Peruvian Amazon). By quantifying and understanding fundamental environmental processes of the past and their responses to change, we can make better predictions about future responses to anthropogenic change, including land use and climate change impacts. Secondly, more in depth research is required concerning the way in which current patterns of human interaction with tropical peatlands, and the underpinning drivers of these patterns, impact on the wellbeing of multiple actors including in supporting local and Indigenous communities. This work needs to include accurate and rigorous assessments of how current activities impact on provisioning and cultural ecosystem services, and the distribution of the socio-economic costs and benefits of these impacts. For this reason, interdisciplinary approaches are likely to be of particular value. Thirdly, more information is needed of how different management interventions (e.g. restoration and conservation activities) can enhance climate and livelihood resilience. This should include examples where interventions have failed in order to identify the most effective solutions and adapt interventions to mitigate adverse outcomes. Crucially, the impact of such interventions on the wellbeing of local and Indigenous communities should also be assessed, focussing on how such knowledge can be employed to support equitable approaches of these areas in ways which balance conservation and protection of these important ecosystems with the wellbeing of communities.

Declaration of Competing Interest

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

Data availability

No data was used for the research described in the article.

Acknowledgements

This work was supported by the Natural Environment Research Council (Grant No. NE/R016860/1 and NE/R000751/1), the German Research Foundation (BE 2116/32–1) for funding KAH; the Newton-Paulet Institutional Links Grant (Grant ref. 220–2018) for part-funding KHR and LESC, and to the Leverhulme Trust for part-funding LESC (Research Grant RPG-2018–306).

References

Abram, N.K., Xofis, P., Tzanopoulos, J., MacMillan, D.C., Ancrenaz, M., Chung, R., Peter, L., Ong, R., Lackman, I., Goossens, B., Ambu, L., Knight, A.T., 2014. Synergies for improving oil palm production and forest conservation in floodplain landscapes. PLoS ONE. https://doi.org/10.1371/journal.pone.0095388.

Aiken, S.R., 2004. Runaway fires, smoke-haze pollution, and unnatural disasters in Indonesia. Geogr. Rev. https://doi.org/10.1111/j.1931-0846.2004.tb00158.x.

Alongi, D.M., 2015. The impact of climate change on mangrove forests. Curr. Clim. Change Rep. 1, 30–39.

Anderson, J.A.R., 1964. The structure and development of the peat swamps of Sarawak and Brunei. J. Trop. Geogr. 18, 7–16.

Anderson, J.A.R., Muller, J., 1975. Palynological study of a holocene peat and a miocene coal deposit from NW borneo. Rev. Palaeobot. Palynol. 19, 291–317. https://doi.org/10.1016/0034-6667(75)90049-4.

Andriesse, J.P., 1988, Nature and management of tropical peat soils, FAO Soils Bulletin.

Anshari, G., Peter Kershaw, A., Van Der Kaars, S., 2001. A Late Pleistocene and Holocene pollen and charcoal record from peat swamp forest, Lake Sentarum wildlife reserve, West Kalimantan, Indonesia. Palaeogeogr., Palaeoclimatol., Palaeoecol. https://doi.org/10.1016/S0031-0182(01)00246-2.

Anshari, G.Z., Afifudin, M., Nuriman, M., Gusmayanti, E., Arianie, L., Susana, R., Nusantara, R.W., Sugardjito, J., Rafiastanto, A., 2010. Drainage and land use impacts on changes in selected peat properties and peat degradation in West Kalimantan Province, Indonesia. Biogeosciences 7, 3403–3419. https://doi.org/10.5194/bg-7-3403-2010.

Arce-Nazario, J.A., 2007. Human landscapes have complex trajectories: Reconstructing Peruvian Amazon landscape history from 1948 to 2005. Landsc. Ecol. https://doi.org/10.1007/s10980-007-9123-5.

Aronson, R.B., Hilbun, N.L., Bianchi, T.S., Filley, T.R., McKee, B.A., 2014. Land use, water quality, and the history of coral assemblages at Bocas del Toro, Panamá. Mar. Ecol. Prog. Ser. 504, 159–170. https://doi.org/10.3354/meps10765.

ASEAN (2016) Association of Southeast Asian Nations Peatland Management Strategy (APMS) (https://asean.org/speechandstatement/asean-peatland-management-strategy-apms-2/).

Aucour, A.M., Bonnefille, R., Hillaire-Marcel, C., 1999. Sources and accumulation rates of organic carbon in an equatorial peat bog (Burundi, East Africa) during the Holocene: carbon isotope constraints. Palaeogeogr. Palaeoclimatol. Palaeoecol. 150, 179–189. https://doi.org/10.1016/S0031-0182(98)00222-3.

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. Geophys. Res. Lett. 44 (3), 1333–1339.
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. doi:10.1073/pnas.0906457106.
Barker, G., Barton, H., Bird, M., Daly, P., Datan, I., Dykes, A., Farr, L., Gilbertson, D., Harrisson, B., Hunt, C., Higham, T., Kealhofer, L., Krigbaum, J., Lewis, H., McLaren, S., Paz, V., Pike, A., Piper, P., Pyatt, B., Rabett, R., Reynolds, T., Rose, J., Rushworth, G., Stephens, M., Stringer, C., Thompson, J., Turney, C., 2007. The "human revolution" in lowland tropical Southeast Asia: the antiquity and behavior of anatomically modern humans at Niah Cave (Sarawak, Borneo). J. Hum. Evol. https://doi.org/10.1016/j.jhevol.2006.08.011.
van Beukering, P.J.H., Schaafsma, M., Davies, O., Oskolokaite, I., 2008. The economic value of peatland resources within the Central Kalimantan Peatland Project in Indonesia Perceptions of local communities. Wetlands.
Blackman, A., Corral, L., Lima, E.S., Asner, G.P., 2017, Titling indigenous communities protects forests in the Peruvian Amazon. Proceedings of the National Academy of Sciences. doi:10.1073/pnas.1603290114.
Blais-McPherson, M., Rudiak-Gould, P., 2017. Strengthening inter-disciplinary and inter-ideological collaboration on REDD: A cultural theory approach. Glob. Environ. Change. https://doi.org/10.1016/j.gloenvcha.2016.10.008.
Carmenta, R., Zabala, A., Daeli, W., Phelps, J., 2017. Perceptions across scales of governance and the Indonesian peatland fires. Glob. Environ. Change. https://doi.org/10.1016/j.gloenvcha.2017.08.001.
Cattau, M.E., Harrison, M.E., Shinyo, I., Tungau, S., Uriarte, M., DeFries, R., 2016a. Sources of anthropogenic fire ignitions on the peat-swamp landscape in Kalimantan, Indonesia. Glob. Environ. Change. https://doi.org/10.1016/j.gloenvcha.2016.05.005.
Cheyne, S.M., 2008. Effects of meteorology, astronomical variables, location and human disturbance on the singing apes: Hylobates albibarbis. Am. J. Primatol. https://doi.org/10.1002/ajp.20502.
Cheyne, S.M., Sastramidjaja, W.J., Muhalir, Rayadin, Y., Macdonald, D.W., 2016. Mammalian communities as indicators of disturbance across Indonesian Borneo. Glob. Ecol. Conserv. https://doi.org/10.1016/j.gecco.2016.06.002.
Chokkalingam, U., Kurniawan, I., Ruchiat, Y., 2005. Fire, livelihoods, and environmental change in the Middle Mahakam Peatlands, East Kalimantan. Ecol. Soc. https://doi.org/10.5751/ES-01320-100126.
Cole, L.E., Åkesson, C.M., Hapsari, K.A., Hawthorne, D., Roucoux, K.H., Girkin, N.T., Thornton, S.A., 2022. Tropical peatlands in the anthropocene: Lessons from the past. Anthropocene 37, 100324.
Cole, L.E.S., Bhagwat, S.A., Willis, K.J., 2014. Recovery and resilience of tropical forests after disturbance. Nat. Commun. https://doi.org/10.1038/ncomms4906.
Cole, L.E.S., Bhagwat, S.A., Willis, K.J., 2015. Long-term disturbance dynamics and resilience of tropical peat swamp forests. J. Ecol. https://doi.org/10.1111/1365-2745.12329.
Cole, L.E.S., Bhagwat, S.A., Willis, K.J., 2019. Fire in the Swamp Forest: Palaeoecological Insights Into Natural and Human-Induced Burning in Intact Tropical Peatlands. Front. For. Glob. Change. https://doi.org/10.3389/ffgc.2019.00048.
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.
Cooper, H.V., Evers, S., Aplin, P., Crout, N., Dahalan, M.P., Bin, Sjogersten, S., 2020. Greenhouse gas emissions resulting from conversion of peat swamp forest to oil palm plantation. Nat. Commun. https://doi.org/10.1038/s41467-020-14298-w.
Couwenberg, J., Dommain, R., Joosten, H., 2010. Greenhouse gas fluxes from tropical peatlands in south-east Asia. Glob. Change Biol. https://doi.org/10.1111/j.1365-2486.2009.02016.x.
Crezee, B., Dargie, G.C., Ewango, C.E.N., et al., 2022. Mapping peat thickness and carbon stocks of the central Congo Basin using field data. Nat. Geosci. 15, 639–644.
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. Sci. Rep. https://doi.org/10.1038/srep37074.
Danielsen, F., Beukema, H., Burgess, N.D., Parish, F., Br??hl, C.A., Donald, P.F., Murdiyarso, D., Phalan, B., Reijnders, L., Struebig, M., Fitzherbert, E.B., 2009. Biofuel Plantations on Forested Lands: Double Jeopardy for Biodiversity and Climate. Conserv. Biol. https://doi.org/10.1111/j.1523-1739.2008.01096.x.
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. https://doi.org/10.1038/nature21048.
Dargie, G.C., Lawson, I.T., Rayden, T.J., Miles, L., Mitchard, E.T.A., Page, S.E., Bocko, Y. E., Ifo, S.A., Lewis, S.L., 2018. Congo Basin peatlands: threats and conservation priorities. Mitig. Adapt. Strateg. Glob. Change. https://doi.org/10.1007/s11027-017-9774-8.
Davies, S.J., Unam, L., 1999. Smoke-haze from the 1997 Indonesian forest fires: Effects on pollution levels, local climate, atmospheric CO_2 concentrations, and tree photosynthesis. For. Ecol. Manag. https://doi.org/10.1016/S0378-1127(99)00060-2.
Davis, S.H., Wali, A., 1994. Indigenous Land Tenure and Management in Latin Tropical America. Ambio. https://doi.org/10.2307/4314265.
van der Werf, G.R., Morton, D.C., DeFries, R.S., Olivier, J.G.J., Kasibhatla, P.S., Jackson, R.B., Collatz, G.J., Randerson, J.T., 2009. CO_2 emissions from forest loss. Nat. Geosci. https://doi.org/10.1038/ngeo671.
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. Front. For. Glob. Change. https://doi.org/10.3389/ffgc.2020.00070.
Dohong, A., Abdul Aziz, A., Dargusch, P., 2018. A review of techniques for effective tropical peatland restoration. Wetlands. https://doi.org/10.1007/s13157-018-1017-6.
Dommain, R., Couwenberg, J., Joosten, H., 2010, Hydrological self-regulation of domed peatlands in south-east Asia and consequences for conservation and restoration.
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. Quat. Sci. Rev. 30, 999–1010. https://doi.org/10.1016/j.quascirev.2011.01.018.
Dommain, R., Couwenberg, J., Glaser, P.H., Joosten, H., Suryadiputra, I.N.N., 2014. Carbon storage and release in Indonesian peatlands since the last deglaciation. Quat. Sci. Rev. https://doi.org/10.1016/j.quascirev.2014.05.002.
Dourojeanni, M.J., 2015. Human impact on protected areas of the Peruvian Amazon. In: Wuerthner, G., Crist, E., Butler, T. (Eds.), Protecting the Wild: Parks and Wilderness, the Foundation for Conservation. The Foundation for Conservation. Island Press, Washington D.C, pp. 215–225.
Draper, F.C., Roucoux, K.H., Lawson, I.T., Mitchard, E.T., Coronado, E.N.H., Lähteenoja, O., Baker, T.R., 2014. The distribution and amount of carbon in the largest peatland complex in Amazonia. Environ. Res. Lett. 9 (12), 124017.
Esquivel-Muelbert, A., Baker, T.R., Dexter, K.G., Lewis, S.L., Brienen, R.J.W., Feldpausch, T.R., Lloyd, J., Monteagudo-Mendoza, A., Arroyo, L., Álvarez-Dávila, E., Higuchi, N., Marimon, B.S., Marimon-Junior, B.H., Silveira, M., Vilanova, E., Gloor, E., Malhi, Y., Chave, J., Barlow, J., Bonal, D., Davila Cardozo, N., Erwin, T., Fauset, S., Hérault, B., Laurance, S., Poorter, L., Qie, L., Stahl, C., Sullivan, M.J.P., ter Steege, H., Vos, V.A., Zuidema, P.A., Almeida, E., de Oliveira, E., Andrade, A., Vieira, S.A., Aragão, L., Araujo-Murakami, A., Arets, E., Aymard C, G.A., Camargo, P. B., Barroso, J.G., Bongers, F., Boot, R., Camargo, J.L., Castro, W., Chama Moscoso, V., Comiskey, J., Cornejo Valverde, F., da Costa, A.C., del Aguila Pasquel, J., Di Fiore, T., Fernanda Duque, L., Elias, F., Engel, J., Flores Llampazo, G., Galbraith, D., Herrera Fernández, R., Honorio Coronado, E., Hubau, W., Jimenez-Rojas, E., Lima, A.J.N., Umetsu, R.K., Laurance, W., Lopez-Gonzalez, G., Lovejoy, T., Aurelio Melo Cruz, O., Morandi, P.S., Neill, D., Núñez Vargas, P., Pallqui, N.C., Parada Gutierrez, A., Pardo, G., Peacock, J., Peña-Claros, M., Peñuela-Mora, M.C., Petronelli, P., Pickavance, G.C., Pitman, N., Prieto, A., Quesada, C., Ramírez-Angulo, H., Réjou-Méchain, M., Restrepo Correa, Z., Roopsind, A., Rudas, A., Salomão, R., Silva, N., Silva Espejo, J., Singh, J., Stropp, J., Terborgh, J., Thomas, R., Toledo, M., Torres-Lezama, A., Valenzuela Gamarra, L., van de Meer, P.J., van der Heijden, G., van der Hout, P., Vasquez Martinez, R., Vela, C., Vieira, I.C.G., Phillips, O.L., 2018. Compositional response of Amazon forests to climate change, 0 Glob. Change Biol.. https://doi.org/10.1111/gcb.14413.
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. Glob. Change Biol. https://doi.org/10.1111/gcb.13422.
Felton, A.M., Engström, L.M., Felton, A., Knott, C.D., 2003. Orangutan population density, forest structure and fruit availability in hand-logged and unlogged peat swamp forests in West Kalimantan, Indonesia. Biol. Conserv. https://doi.org/10.1016/S0006-3207(03)00013-2.
Gandois, L., Cobb, A.R., Hei, I.C., Lim, L.B.L., Abu Salim, K., Harvey, C.F., 2013. Impact of deforestation on solid and dissolved organic matter characteristics of tropical peat forests: implications for carbon release. Biogeochemistry 114, 183–199. https://doi.org/10.1007/s10533-012-9799-8.
Garcin, Y., Schefuß, E., Dargie, G.C., et al., 2022. Hydroclimatic vulnerability of peat carbon in the central Congo Basin. Nature. https://doi.org/10.1038/s41586-022-05389-3.
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., 2014. Major atmospheric emissions from peat fires in Southeast Asia during non-drought years: Evidence from the 2013 Sumatran fires. Sci. Rep. https://doi.org/10.1038/srep06112.
Gibbs, H.K., Johnston, M., Foley, J.A., Holloway, T., Monfreda, C., Ramankutty, N., Zaks, D., 2008. Carbon payback times for crop-based biofuel expansion in the tropics: The effects of changing yield and technology. Environ. Res. Lett. https://doi.org/10.1088/1748-9326/3/3/034001.
Giesen, W., 2015. Utilising non-timber forest products to conserve Indonesia's peat swamp forests and reduce carbon emissions. J. Indones. Nat. Hist. 3, 10–19.
Giesen, W., Sari, E.N.N., 2018, Tropical Peatland Restoration Report: the Indonesian case. Jakarta.
Girkin, N.T., Turner, B.L., Ostle, N., Craigon, J., Sjögersten, S., 2018a. Root exudate analogues accelerate CO_2 and CH4 production in tropical peat. Soil Biol. Biochem. 117, 48–55. https://doi.org/10.1016/j.soilbio.2017.11.008.
Girkin, N.T., Turner, B.L., Ostle, N., Sjögersten, S., 2018b. Composition and concentration of root exudate analogues regulate greenhouse gas fluxes from tropical peat. Soil Biol. Biochem. 127, 280–285 doi:S0038071718303444.
Girkin, N.T., Vane, C.H., Cooper, H.V., Moss-Hayes, V., Craigon, J., Turner, B.L., Sjögersten, S., 2019. Spatial variability of organic matter properties determines methane fluxes in a tropical forested peatland. Biogeochemistry 142 (2), 231–245.
Girkin, N.T., Dhandapani, S., Evers, S., Ostle, N., Turner, B.L., Sjögersten, S., 2020a. Interactions between labile carbon, temperature and land use regulate carbon dioxide and methane production in tropical peat. Biogeochemistry 147, 87–97.
Girkin, N.T., Vane, C.H., Turner, B.L., Ostle, N.J., Sjögersten, S., 2020b. Root oxygen mitigates methane fluxes in tropical peatlands. Environ. Res. Lett. 11. https://doi.org/10.1088/1748-9326/ab8495.

Goldstein, J.E., 2015. Knowing the subterranean: Land grabbing, oil palm, and divergent expertise in Indonesia's peat soil. Environ. Plan. A. https://doi.org/10.1177/0308518X15599787.

GPI (2021) Global Peatlands Initiative. Available at: https://www.globalpeatlands.org/. Accessed 23rd March 2022.

Graham, A.M., Pope, R.J., Pringle, K.P., Arnold, S., Chipperfield, M.P., Conibear, L.A., Butt, E.W., Kiely, L., McQuaid, J.B., 2020. Impact on air quality and health due to the Saddleworth Moor fire in northern England. Environ. Res. Lett. 15 (7), 074018.

Graham, L.L.B., Page, S.E., 2012. Artificial Bird Perches for the Regeneration of Degraded Tropical Peat Swamp Forest: A Restoration Tool with Limited Potential. Restor. Ecol. https://doi.org/10.1111/j.1526-100X.2011.00805.x.

Green, S.M., Page, S., 2017. Tropical peatlands: current plight and the need for responsible management. Geol. Today. https://doi.org/10.1111/gto.12197.

Green Climate Fund, 2015, FP001: Building the Resilience of Wetlands in the Province of Datem del Marañón, Peru. Available at: ⟨https://www.greenclimate.fund/sites/default/files/document/funding-proposal-fp001-profonanpe-peru.pdf⟩. Accessed 28th October 2022.

Gutiérrez-Vélez, V.H., DeFries, R., Pinedo-Vásquez, M., Uriarte, M., Padoch, C., Baethgen, W., Fernandes, K., Lim, Y., 2011. High-yield oil palm expansion spares land at the expense of forests in the Peruvian Amazon. Environ. Res. Lett. https://doi.org/10.1088/1748-9326/6/4/044029.

Harris, J.B.C., Green, J.M.H., Prawiradilaga, D.M., Giam, X., Giyanto, Hikmatullah, D., Putra, C.A., Wilcove, D.S., 2015. Using market data and expert opinion to identify overexploited species in the wild bird trade. Biol. Conserv. https://doi.org/10.1016/j.biocon.2015.04.009.

Harrison, M., Wijedasa, L., Cole, L., Cheyne, S., Choiruzzad, S., Chua, L., Dargie, G., Ewango, C.E.N., HonorioCoronado, E., Ifo, S., Imron, M., Kopansky, D., Lestarisa, T., O'Reilly, P., Van Offelen, J., Refisch, J., Roucoux, K., Sugardjito, J., Thornton, S., Upton, C., Page, S., 2020a. Tropical peatlands and their conservation are important in the context of COVID-19 and potential future (zoonotic) disease pandemics. PeerJ doi:8:e10283 DOI 10.7717/peerj.10283.

Harrison, M.E., Rieley, J.O., 2018. Tropical peatland biodiversity and conservation in Southeast Asia. Foreword Mires Peat 22, 1–7.

Harrison, M.E., Capilla, B., Thornton, S., Cattau, M., Page, S., 2016, Impacts of the 2015 fire season on peat-swamp forest biodiversity in Indonesian Borneo, in: Proceedings of the 15th International Peat Congress, Kuching, Malaysia. pp. 713–717.

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., Page, S.E., Priyanto, R., Ripoll Capilla, B., Rowland, L., Santos, E.M., Schreer, V., Sudyana, I.N., Taman, S.B.B., Thornton, S.A., Upton, C., Wich, S.A., Veen, F.J.F., 2020b. Tropical forest and peatland conservation in Indonesia: Challenges and directions. People Nat. https://doi.org/10.1002/pan3.10060.

Henderson, J., Osborne, D.J., 2000. The oil palm in all our lives: How this came about. Endeavour. https://doi.org/10.1016/S0160-9327(00)01293-X.

Herawati, H., Santoso, H., 2011. Tropical forest susceptibility to and risk of fire under changing climate: A review of fire nature, policy and institutions in Indonesia. For. Policy Econ. https://doi.org/10.1016/j.forpol.2011.02.006.

Hergoualc'H, K., Verchot, L.V., 2011. Stocks and fluxes of carbon associated with land use change in Southeast Asian tropical peatlands: A review. Glob. Biogeochem. Cycles. https://doi.org/10.1029/2009GB003718.

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. Glob. Change Biol. 18, 3410–3422. https://doi.org/10.1111/j.1365-2486.2012.02793.x.

Hooijer, A., Page, S., Canadell, J.G., Silvius, M., Kwadijk, J., Wosten, H., Jauhiainen, J., 2010. Current and future CO_2 emissions from drained peatlands in Southeast Asia. Biogeosciences 7, 1505–1514. https://doi.org/10.5194/bg-7-1505-2010.

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.

Hope, G., Chokkalingam, U., Anwar, S., 2005. The stratigraphy and fire history of the Kutai Peatlands, Kalimantan, Indonesia. Quat. Res. https://doi.org/10.1016/j.yqres.2005.08.009.

Hoyos-Santillan, J., Lomax, B.H., Large, D., Turner, B.L., Boom, A., Lopez, O.R., Sjögersten, S., 2015. Getting to the root of the problem: litter decomposition and peat formation in lowland Neotropical peatlands. Biogeochemistry 126, 115–129. https://doi.org/10.1007/s10533-015-0147-7.

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. Sci. Rep. https://doi.org/10.1038/srep26886.

Hunt, C.O., Premathilake, R., 2012. Early Holocene vegetation, human activity and climate from Sarawak, Malaysian Borneo. Quat. Int. https://doi.org/10.1016/j.quaint.2011.04.027.

IPCC, 2013, Climate Change 2013: The Physical Science Basis, Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. doi:10.1017/CBO9781107415324.

Jaenicke, J., Wösten, H., Budiman, A., Siegert, F., 2010. Planning hydrological restoration of peatlands in Indonesia to mitigate carbon dioxide emissions. Mitig. Adapt. Strateg. Glob. Change. https://doi.org/10.1007/s11027-010-9214-5.

Janovec, J., Householder, J.E., Tobler, M., 2013, An assessment of current impacts and threats on aguajal wetlands and oxbow lakes of Madre de Dios, Peru. Lima, Peru.

Jauhiainen, J., Hooijer, A., Page, S., 2012. Carbon dioxide emissions from an Acacia plantation on peatland in Sumatra, Indonesia. Biogeosciences 9, 617–630. https://doi.org/10.5194/bg-9-617-2012.

Jefferson, U., Carmenta, R., Daeli, W., Phelps, J., 2020. Characterising policy responses to complex socio-ecological problems: 60 fire management interventions in Indonesian peatlands. Glob. Environ. Change. https://doi.org/10.1016/j.gloenvcha.2019.102027.

Jones, H.P., Schmitz, O.J., 2009. Rapid recovery of damaged ecosystems. PLoS ONE. https://doi.org/10.1371/journal.pone.0005653.

Joosten, H., Clarke, D., 2002. Wise Use of Mires and Peatlands. International Mire Conservation Group and International Peat Society,, Helsinki.

Joosten, H., Tapio-Biström, M.-L., Tol, S., 2012. Peatlands - guidance for climate change mitigation through conservation, rehabilitation and sustainable use. Mitig. Clim. Change Agric. (MICCA) Program. Ser. 5.

Kelly, T.J., Lawson, I.T., Roucoux, K.H., Baker, T.R., Honorio-Coronado, E.N., Jones, T. D., Rivas Panduro, S., 2018. Continuous human presence without extensive reductions in forest cover over the past 2500 years in an aseasonal Amazonian rainforest. J. Quat. Sci. 33, 369–379. https://doi.org/10.1002/jqs.3019.

Khasanah, N., van Noordwijk, M., 2018. Subsidence and carbon dioxide emissions in a smallholder peatland mosaic in Sumatra, Indonesia. Mitig. Adapt. Strateg. Glob. Change. https://doi.org/10.1007/s11027-018-9803-2.

Koh, L.P., Miettinen, J., Liew, S.C., Ghazoul, J., 2011, Remotely sensed evidence of tropical peatland conversion to oil palm. Proceedings of the National Academy of Sciences. doi:10.1073/pnas.1018776108.

Kondo, M., Ichii, K., Patra, P.K., et al., 2018. Land use change and El Niño-Southern Oscillation drive decadal carbon balance shifts in Southeast Asia. Nat. Commun. 9, 1154. https://doi.org/10.1038/s41467-018-03374-x.

Kopansky, D., 2018, Keeping the Cuvette Centrale Peatlands wet [WWW Document]. UN-REDD Programme. URL ⟨http://www.un-redd.org/single-post/2018/05/23/Keeping-the-Cuvette-Centrale-Peatlands-Wet⟩ (accessed 10.20.18).

Koplitz, S.N., Mickley, L.J., Marlier, M.E., Buonocore, J.J., Kim, P.S., Liu, T., Sulprizio, M. P., DeFries, R.S., Jacob, D.J., Schwartz, J., Pongsiri, M., Myers, S.S., 2016. Public health impacts of the severe haze in Equatorial Asia in September-October 2015: Demonstration of a new framework for informing fire management strategies to reduce downwind smoke exposure. Environ. Res. Lett. https://doi.org/10.1088/1748-9326/11/9/094023.

Laurance, W.F., Goosem, M., Laurance, S.G., 2009. Impacts of roads and linear clearings on tropical forests. Trends Ecol. Evol. 24 (12), 659–669.

Lawson, I.T., Kelly, T.J., Aplin, P., Boom, A., Dargie, G., Draper, F.C.H., Wheeler, J., 2015. Improving estimates of tropical peatland area, carbon storage, and greenhouse gas fluxes. Wetl. Ecol. Manag. 23 (3), 327–346.

Lawson, I.T., Honorio Coronado, E.N., Andueza, L., Cole, L., Dargie, G.C., Davies, A.L., Laurie, N., Okafor-Yarwood, I., Roucoux, K.H., Simpson, M., 2022. The vulnerability of tropical peatlands to oil and gas exploration and extraction. Prog. Environ. Geogr. 0 (0) https://doi.org/10.1177/27539687221124046.

Leifeld, J., Menichetti, L., 2018. The underappreciated potential of peatlands in global climate change mitigation strategies. Nat. Commun. https://doi.org/10.1038/s41467-018-03406-6.

Leng, L.Y., Ahmed, O.H., Jalloh, M.B., 2018. Brief review on climate change and tropical peatlands. Geosci. Front. 1–8.

Lewis, S., Lloyd, J., Sitch, S., Mitchard, E., Laurance, W., 2009. Changing ecology of tropical forests: evidence and drivers. Annu. Rev. Ecol., Evol., Syst. 40, 529–549.

Li, W., Dickinson, R.E., Fu, R., Niu, G.Y., Yang, Z.L., Canadell, J.G., 2007. Future precipitation changes and their implications for tropical peatlands. Geophys. Res. Lett. https://doi.org/10.1029/2006GL028364.

Lo, J., Quoi, L.P., Visal, S., 2018. Some preliminary observations on peat-forming mangroves in Botum Sakor. Cambodia Mires Peat 22.

Loisel, J., Gallego-Sala, A., Amesbury, M., Magnan, G., Anshari, G., Beilman, D., Benavides, J., Blewett, J., Camill, P., Charman, D., Chawchai, S., Hedgpeth, A., Kleinen, T., Korhola, A., Large, D., Mansilla, C., Müller, J., van Bellen, S., West, J., Yu, Z., Bubier, J., Garneau, M., Moore, T., Sannel, A., Page, S., Väliranta, M., Bechtold, M., Brovkin, V., Cole, L., Chanton, J., Christensen, T., Davies, M., De Vleeschouwer, F., Finkelstein, S., Frolking, S., Galka, M., Gandois, L., Girkin, N., Harris, L., Heinemeyer, A., Hoyt, A., Jones, M., Joos, F., Juutinen, S., Kaiser, K., Lacourse, T., Lamentowicz, M., Larmola, T., Leifeld, J., Lohila, A., Milner, A., Minkkinen, K., Moss, P., Naafs, B., Nichols, J., O'Donnell, J., Payne, R., Philben, M., Piilo, S., Quillet, A., Ratnayake, A., Roland, T., Sjögersten, S., Sonnentag, O., Swindles, G., Swinnen, W., Talbot, J., Treat, C., Valach, A., Wu, J., 2020, Expert assessment of future vulnerability of the global peatland carbon sink. doi:10.1038/s41558–020-00944–0.

Lovelock, C.E., Ruess, R.W., Feller, I.C., 2011. CO_2 efflux from cleared mangrove peat. PloS One 6 (6), e21279.

MacDonald, G.M., Beilman, D.W., Kremenetski, K.V., Sheng, Y., Smith, L.C., Velichko, A. A., 2006. Rapid early development of circumarctic peatlands and atmospheric CH4and CO2variations. Science. https://doi.org/10.1126/science.1131722.

Malhi, Y., 2010. The carbon balance of tropical forest regions, 1990–2005. Curr. Opin. Environ. Sustain. 2, 237–244.

Malhi, Y., 2012. The productivity, metabolism and carbon cycle of tropical forest vegetation. J. Ecol. 100, 65–75.

Malhi, Y., Gardner, T.A., Goldsmith, G.R., Silman, M.R., Zelazowski, P., 2014. Tropical Forests in the Anthropocene. SSRN. https://doi.org/10.1146/annurev-environ-030713-155141.

Manzo, K., Padfield, R., Varkkey, H., 2020. Envisioning tropical environments: Representations of peatlands in Malaysian media. Environ. Plan. E: Nat. Space. https://doi.org/10.1177/2514848619880895.

Marshall, C., Large, D.J., Athab, A., Evers, S.L., Sowter, A., Marsh, S., Sjögersten, S., 2018. Monitoring tropical peat related settlement using ISBAS InSAR, Kuala Lumpur International Airport (KLIA. Eng. Geol. https://doi.org/10.1016/j.enggeo.2018.07.015.

Miettinen, J., Liew, S.C., 2010. Degradation and development of peatlands in Peninsular Malaysia and in the islands of Sumatra and Borneo since 1990. Land Degrad. Dev. https://doi.org/10.1002/ldr.976.
Miettinen, J., Shi, C., Liew, S.C., 2011a. Influence of peatland and land cover distribution on fire regimes in insular Southeast Asia. Reg. Environ. Change. https://doi.org/10.1007/s10113-010-0131-7.
Miettinen, J., Shi, C., Liew, S.C., 2011b. Deforestation rates in insular Southeast Asia between 2000 and 2010. Glob. Change Biol. https://doi.org/10.1111/j.1365-2486.2011.02398.x.
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. 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. Environ. Res. Lett. https://doi.org/10.1088/1748-9326/aa5b6f.
Miranda, J.J., Corral, L., Blackman, A., Asner, G., Lima, E., 2016. Effects of Protected Areas on Forest Cover Change and Local Communities: Evidence from the Peruvian Amazon. World Dev. https://doi.org/10.1016/j.worlddev.2015.10.026.
Moeliono, M., Brockhaus, M., Gallemore, C., Dwisatrio, B., Maharani, C.D., Muharrom, E., Pham, T.T., 2020. REDD+ in Indonesia: A new mode of governance or just another project. For. Policy Econ. https://doi.org/10.1016/j.forpol.2020.102316.
Moore, P.D., 1989. The Ecology of Peat-Forming Processes - a Review. Int. J. Coal Geol. 12, 89–103. https://doi.org/10.1016/0166-5162(89)90048-7.
Murdiyarso, D., Donato, D., Kauffman, J.B., Kurnianto, S., Stidham, M., & Kanninen, M., 2009, Carbon storage in mangrove and peatland ecosystems: A preliminary account from plots in Indonesia. Working paper 48. Bogor Banat, Indonesia: Center for International Forestry Research. 35 p., 1–35.
Murdiyarso, D., Hergoualc'h, K., Verchot, L.V., 2010, Opportunities for reducing greenhouse gas emissions in tropical peatlands. Proceedings of the National Academy of Sciences of the United States of America. doi:10.1073/pnas.0911966107.
Murphy, D.J., Goggin, K., Paterson, R.R.M., 2021. Oil palm in the 2020s and beyond: challenges and solutions. CABI Agric. Biosci. 2, 39. https://doi.org/10.1186/s43170-021-00058-3.
Oktarita, S., Hergoualc'H, K., Anwar, S., Verchot, L.V., 2017. Substantial N2O emissions from peat decomposition and N fertilization in an oil palm plantation exacerbated by hotspots. Environ. Res. Lett. https://doi.org/10.1088/1748-9326/aa80f1.
Osland, M.J., Feher, L.C., Spivak, A.C., Nestlerode, J.A., Almario, A.E., Cormier, N., From, A.S., Krauss, K.W., Russell, M.J., Alvarez, F., Dantin, D.D., 2020. Rapid peat development beneath created, maturing mangrove forests: ecosystem changes across a 25–yr chronosequence. Ecol. Appl. 30 (4), e02085.
Padfield, R., Waldron, S., Drew, S., Papargyrpoulou, E., Kumaran, S., Page, S., Gilvear, D., Armstrong, A., Evers, S., Williams, P., Zakaria, Z., Chin, S.Y., Balle Hansen, S., Campos-Arceiz, A., Latif, M.T., Sayok, A., Tham, M.H., 2014. Research Agendas for the Sustainable Management of Tropical Peatland in Malaysia. Environ. Conserv. https://doi.org/10.1017/S0376892914000034.
Page, S., Hosciło, 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. https://doi.org/10.1007/s10021-008-9216-2.
Page, S.E., Hooijer, A., 2016. In the line of fire: The peatlands of Southeast Asia. Philos. Trans. R. Soc. B: Biol. Sci. https://doi.org/10.1098/rstb.2015.0176.
Page, S.E., Siegert, F., Rieley, J.O., Boehm, H.D.V., 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.
Page, S.E., Wust, R.A.J., Weiss, D., Rieley, J.O., Shotyk, W., Limin, S.H., 2004a. A record of Late Pleistocene and Holocene carbon accumulation and climate change from an equatorial peat bog (Kalimantan, Indonesia): implications for past, present and future carbon dynamics. J. Quat. Sci. 19, 625–635. https://doi.org/10.1002/Jqs.884.
Page, S.E., Wűst, R.A.J., Weiss, D., Rieley, J.O., Shotyk, W., Limin, S.H., 2004b. A record of Late Pleistocene and Holocene carbon accumulation and climate change from an equatorial peat bog (Kalimantan, Indonesia): implications for past, present and future carbon dynamics. J. Quat. Sci. https://doi.org/10.1002/jqs.884.
Page, S.E., Rieley, J.O., Wüst, R., 2006. Chapter 7 Lowland tropical peatlands of Southeast Asia. Dev. Earth Surf. Process. https://doi.org/10.1016/S0928-2025(06)09007-9.
Page, S.E., Rieley, J.O., Banks, C.J., 2011. Global and regional importance of the tropical peatland carbon pool. Glob. Change Biol. https://doi.org/10.1111/j.1365-2486.2010.02279.x.
Phillips, S., Rouse, G.E., Bustin, R.M., 1997. Vegetation zones and diagnostic pollen profiles of a coastal peat swamp, Bocas del Toro, Panama. Palaeogeogr., Palaeoclimatol., Palaeoecol. 128, 301–338.
Poesie, E.S., Shimamura, T., Page, S.E., Ninomiya, I., Limin, S.H., 2010, Species composition and phylogenetic diversity in a tropical peat swamp forest, Central Kalimantan, Indonesia. Tropics.
Posa, M.R.C., Wijedasa, L.S., Corlett, R.T., 2011. Biodiversity and Conservation of Tropical Peat Swamp Forests. BioScience. https://doi.org/10.1525/bio.2011.61.1.10.
Potter, L., 2015. Managing oil palm landscapes: A seven-country survey of the modern palm oil industry in Southeast Asia, Latin America and West Africa. Manag. oil Palm. Landsc.: A Seven-.-. -Ctry. Surv. Mod. Palm. oil Ind. Southeast Asia, Lat. Am. West Afr. https://doi.org/10.17528/cifor/005612.
Purnomo, H., Shantiko, B., Sitorus, S., Gunawan, H., Achdiawan, R., Kartodihardjo, H., Dewayani, A.A., 2017. Fire economy and actor network of forest and land fires in Indonesia. For. Policy Econ. https://doi.org/10.1016/j.forpol.2017.01.001.
Ramsar, 2014, The List of Wetlands of International Importance.
Ramsar, 2017, Largest Transboundary Ramsar Site in in the world established in the Congo River Basin [WWW Document]. Ramsar. URL https://www.ramsar.org/news/largest-transboundary-ramsar-site-in-in-the-world-established-in-the-congo-river-basin (accessed 10.14.18).
Rappold, A.G., Stone, S.L., Cascio, W.E., Neas, L.M., Kilaru, V.J., Carraway, M.S., Szykman, J.J., Ising, A., Cleve, W.E., Meredith, J.T., Vaughan-Batten, H., Deyneka, L., Devlin, R.B., 2011. Peat Bog Wildfire Smoke Exposure in Rural North Carolina Is Associated with Cardiopulmonary Emergency Department Visits Assessed through Syndromic Surveillance. Environ. Health Perspect. 119 (10), 1415–1420.
Rein, G., Cleaver, N., Ashton, C., Pironi, P., Torero, J.L., 2008. The severity of smouldering peat fires and damage to the forest soil. Catena. https://doi.org/10.1016/j.catena.2008.05.008.
Richards, D.R., Freiss, D.A., 2015. Rates and drivers of mangrove deforestation in Southeast Asia, 2000–2012. PNAS 13 (2), 344–349. https://doi.org/10.1073/pnas.1510272113.
Rixen, T., Baum, A., Wit, F., Samiaji, J., 2016. Carbon Leaching from Tropical Peat Soils and Consequences for Carbon Balances. Front. Earth Sci. https://doi.org/10.3389/feart.2016.00074.
Roberts, A., Bovin, N., Kaplan, J.O., 2018. Finding the anthropocene in tropical forests. Anthropocene 23, 5–16.
Roucoux, K.H., Lawson, I.T., Baker, T.R., Del Castillo Torres, D., Draper, F.C., Lähteenoja, O., Gilmore, M.P., Honorio Coronado, E.N., Kelly, T.J., Mitchard, E.T.A., Vriesendorp, C.F., 2017, Threats to intact tropical peatlands and opportunities for their conservation. Conservation Biology. doi:10.1111/cobi.12925.
Santoso, M.A., Christensen, E.G., Yang, J., Rein, G., 2019. Review of the Transition From Smouldering to Flaming Combustion in Wildfires. Front. Mech. Eng. https://doi.org/10.3389/fmech.2019.00049.
Saputra, E., 2019. Beyond fires and deforestation: Tackling land subsidence in peatland areas, A case study from Riau, Indonesia. Land. https://doi.org/10.3390/land8050076.
Schreven, S.J.J., Perlett, E.D., Jarrett, B.J.M., Marchant, N.C., Harsanto, F.A., Purwanto, A., Sýkora, K.V., Harrison, M.E., 2018. Forest gaps, edge, and interior support different ant communities in a tropical peat-swamp forest in Borneo. Asian Myrmecol. https://doi.org/10.20362/am.010010.
Schulz, C., Martín Brañas, M., Núñez Pérez, C., Del Aguila Villacorta, M., Laurie, N., Lawson, I.T., Roucoux, K.H., 2019. Uses, cultural significance, and management of peatlands in the Peruvian Amazon: Implications for conservation. Biol. Conserv. https://doi.org/10.1016/j.biocon.2019.04.005.
Schulz, C., Martín Brañas, M., Núñez Pérez, C., Del Aguila Villacorta, M., Laurie, N., Lawson, I.T., Roucoux, K.H., 2019a. Peatland and wetland ecosystems in Peruvian Amazonia: indigenous classifications and perspectives. Ecol. Soc. 24. https://doi.org/10.5751/ES-10886-240212.
Schulz, C., Martín Brañas, M., Núñez Pérez, C., Del Aguila Villacorta, M., Laurie, N., Lawson, I.T., Roucoux, K.H., 2019b. Uses, cultural significance, and management of peatlands in the Peruvian Amazon: Implications for conservation. Biol. Conserv. 235, 189–198. https://doi.org/10.1016/j.biocon.2019.04.005.
Sheil, D., Casson, A., Meijaard, E., Noordwijk, M. van, Gaskell, J., Sunderland-Groves, J., Wertz, K., Kanninen, M., 2009. The impacts and opportunities of oil palm in Southeast Asia.
Shlisky, A., Alencar, A., Manta, M., Curran, L.M., 2009, Global fire regime conditions, threats, and opporunities for fire management in the tropics, in: Tropical Fire Ecology. doi:10.1007/978–3-540–77381-8_3.
Simorangkir, D., 2007. Fire use: Is it really the cheaper land preparation method for large-scale plantations. Mitig. Adapt. Strateg. Glob. Change. https://doi.org/10.1007/s11027-006-9049-2.
Sinclair, A.L., Graham, L.L., Putra, E.I., Saharjo, B.H., Applegate, G., Grover, S.P., Cochrane, M.A., 2020. Effects of distance from canal and degradation history on peat bulk density in a degraded tropical peatland. Sci. Total Environ. 699, 134199.
Sjögersten, S., Black, C.R., Evers, S., Hoyos-Santillan, J., Wright, E.L., Turner, B.L., 2014. Tropical wetlands: A missing link in the global carbon cycle? Glob. Biogeochem. Cycles 28, 1371–1386. https://doi.org/10.1002/2014gb004844.
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. https://doi.org/10.1016/j.geoderma.2018.02.029.
Skutsch, M., Turnhout, E., 2020. REDD+: If communities are the solution, what is the problem? World Dev. 130, 104942.
Srisunthon, P., Chawchai, S., 2020. Land-use changes and the effects of oil palm expansion on a peatland in southern Thailand. Front. Earth Sci. 8, 559868.
Struebig, M.J., Harrison, M.E., Cheyne, S.M., Limin, S.H., 2007. Intensive hunting of large flying foxes Pteropus vampyrus natunae in Central Kalimantan, Indonesian Borneo. ORYX. https://doi.org/10.1017/S0030605307000310.
Sumarga, E., Hein, L., Hooijer, A., Vernimmen, R., 2016. Hydrological and economic effects of oil palm cultivation in Indonesian peatlands. Ecol. Soc. https://doi.org/10.5751/ES-08490-210252.
Taillardat, P., Thompson, B.S., Garneau, M., Trottier, K., Friess, D.A., 2020. Climate change mitigation potential of wetlands and the cost-effectiveness of their restoration. Interface Focus 10 (5), 20190129.
Tan, Z.D., Lupascu, M., Wijedasa, L.S., 2021. Paludiculture as a sustainable land use alternative for tropical peatlands: A review. Sci. Total Environ. https://doi.org/10.1016/j.scitotenv.2020.142111.

Tan-Soo, J.S., Pattanayak, S.K., 2019, Seeking natural capital projects: Forest fires, haze, and early-life exposure in Indonesia. Proceedings of the National Academy of Sciences of the United States of America. doi:10.1073/pnas.1802876116.

Thompson, M.C., Baruah, M., Carr, E.R., 2011. Seeing REDD+ as a project of environmental governance. Environ. Sci. Policy. https://doi.org/10.1016/j.envsci.2010.11.006.

Thornton, S.A., 2017. (Un)tangling the Net, Tackling the Scales and Learning to Fish: An Interdisciplinary Study in Indonesian Borneo. University of Leicester,, UK.

Thornton, S.A., Dudin, Page, S.E., Upton, C., Harrison, M.E., 2018. Peatland fish of Sebangau, Borneo: diversity, monitoring and conservation. Mires Peat 22, 1–25.

Tonks, A.J., Aplin, P., Beriro, D.J., Cooper, H., Evers, S., Vane, C.H., Sjogersten, 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.

Turetsky, M.R., Benscoter, B., Page, S., Rein, G., van der Werf, G.R., Watts, A., 2015. Global vulnerability of peatlands to fire and carbon loss. Nat. Geosci. 8, 11–14. https://doi.org/10.1038/Ngeo2325.

UNEP, 2016, The Global Peatland Initiative [WWW Document]. URL https://www.globalpeatlands.org/ (accessed 10.20.18).

Upton, A., Vane, C.H., Girkin, N., Turner, B.L., Sjögersten, S., 2018. Does litter input determine carbon storage and peat organic chemistry in tropical peatlands? Geoderma 326, 76–87. https://doi.org/10.1016/j.geoderma.2018.03.030.

Van Lent, J., Hergoualc'H, K., Verchot, L.V., 2015. Reviews and syntheses: Soil N2O and NO emissions from land use and land-use change in the tropics and subtropics: A meta-analysis. Biogeosciences. https://doi.org/10.5194/bg-12-7299-2015.

Varkkey, H., O'Reilly, P., 2019. Sociopolitical Responses Toward Transboundary Haze. Southeast Asia Environ. Sustain. Context 65.

Wang, S., et al., 2018. Potential shift from a carbon sink to a source in Amazonian peatlands under a changing climate. Proc. Natl. Acad. Sci. 115 (49), 12407–12412. https://doi.org/10.1073/pnas.1801317115.

Warren, M., Frolking, S., Dai, Z., Kurnianto, S., 2017. Impacts of land use, restoration, and climate change on tropical peat carbon stocks in the twenty-first century: implications for climate mitigation. Mitig. Adapt. Strateg. Glob. Change. https://doi.org/10.1007/s11027-016-9712-1.

Wijedasa, L.S., Jauhiainen, J., Könönen, M., Lampela, M., Vasander, H., Leblanc, M.C., Evers, S., Smith, T.E.L., Yule, C.M., Varkkey, H., Lupascu, M., Parish, F., Singleton, I., Clements, G.R., Aziz, S.A., Harrison, M.E., Cheyne, S., Anshari, G.Z., Meijaard, E., Goldstein, J.E., Waldron, S., Hergoualc'h, K., Dommain, R., Frolking, S., Evans, C.D., Posa, M.R.C., Glaser, P.H., Suryadiputra, N., Lubis, R., Santika, T., Padfield, R., Kurnianto, S., Hadisiswoyo, P., Lim, T.W., Page, S.E., Gauci, V., Van Der Meer, P.J., Buckland, H., Garnier, F., Samuel, M.K., Choo, L.N.L.K., O'Reilly, P., Warren, M., Suksuwan, S., Sumarga, E., Jain, A., Laurance, W.F., Couwenberg, J., Joosten, H., Vernimmen, R., Hooijer, A., Malins, C., Cochrane, M.A., Perumal, B., Siegert, F., Peh, K.S.H., Comeau, L.P., Verchot, L., Harvey, C.F., Cobb, A., Jaafar, Z., Wï¿½sten, H., Manuri, S., Mï¿½ller, M., Giesen, W., Phelps, J., Yong, D.L., Silvius, M., Wedeux, B.M.M., Hoyt, A., Osaki, M., Hirano, T., Takahashi, H., Kohyama, T.S., Haraguchi, A., Nugroho, N.P., Coomes, D.A., Quoi, L.P., Dohong, A., Gunawan, H., Gaveau, D.L.A., Langner, A., Lim, F.K.S., Edwards, D.P., Giam, X., Van Der Werf, G., Carmenta, R., Verwer, C.C., Gibson, L., Gandois, L., Graham, L.L.B., Regalino, J., Wich, S.A., Rieley, J., Kettridge, N., Brown, C., Pirard, R., Moore, S., Capilla, B.R., Ballhorn, U., Ho, H.C., Hoscilo, A., Lohberger, S., Evans, T.A., Yulianti, N., Blackham, G., Onrizal, Husson, S., Murdiyarso, D., Pangala, S., Cole, L.E.S., Tacconi, L., Segah, H., Tonoto, P., Lee, J.S.H., Schmilewski, G., Wulffraat, S., Putra, E.I., Cattau, M.E., Clymo, R.S., Morrison, R., Mujahid, A., Miettinen, J., Liew, S.C., Valpola, S., Wilson, D., D'Arcy, L., Gerding, M., Sundari, S., Thornton, S. A., Kalisz, B., Chapman, S.J., Su, A.S.M., Basuki, I., Itoh, M., Traeholt, C., Sloan, S., Sayok, A.K., Andersen, R., 2017. Denial of long-term issues with agriculture on tropical peatlands will have devastating consequences. Global Change Biology. https://doi.org/10.1111/gcb.13516.

Wosten, J.H.M., Clymans, E., Page, S., Rieley, J., Limin, S., 2008. Peat-water interrelationships in a tropical ecosystem in Southeast Asia. Catena 73, 212–224. https://doi.org/10.1016/j.catena.2007.07.010.

Wösten, J.H.M., Ismail, A.B., Van Wijk, A.L.M., 1997. Peat subsidence and its practical implications: A case study in Malaysia. Geoderma. https://doi.org/10.1016/S0016-7061(97)00013-X.

WRI, 2018, Indonesian Peat Prize Announces Winner of $1 million [WWW Document]. Webpage. URL ⟨https://www.wri.org/news/2018/02/release-indonesian-peat-prize-announces-winner-1-million-international-peat-mapping⟩ (accessed 10.30.20).

Xu, J., Morris, P.J., Liu, J., Holden, J., 2018. PEATMAP: Refining estimates of global peatland distribution based on a meta-analysis. Catena. https://doi.org/10.1016/j.catena.2017.09.010.

Yu, Z.C., Loisel, J., Brosseau, D.P., Beilman, D.W., Hunt, S.J., 2010. Global peatland dynamics since the Last Glacial Maximum. Geophys. Res. Lett. 37 doi:Artn L13402Doi 10.1029/2010gl043584.

Yule, C.M., Gomez, L.N., 2009. Leaf litter decomposition in a tropical peat swamp forest in Peninsular Malaysia. Wetl. Ecol. Manag. 17, 231–241. https://doi.org/10.1007/s11273-008-9103-9.