The impact of peatland restoration practice on hydrological functions in tropical environments

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Thesis by alternative format rationale

This PhD thesis follows the University of Leeds Faculty of Environment protocol for the format and presentation of an alternative style of doctoral thesis including published materials. The research questions of the PhD project were investigated using several approaches, resulting in three individual research manuscripts presented as chapters that form the main body of this thesis. Two of the manuscripts have now been published and the final one is under consideration for publication. The main body of this thesis is accompanied by an introduction and a synthesis chapter. The introduction chapter provides background information, reviews of relevant literature, objectives of this PhD research, information about the study site, overviews of the methods used, and an outline of the thesis. The synthesis chapter links the three individual research manuscripts together into a coherent body of work, discussing them in the context of the research questions, and indicating directions for future work.

Declaration of thesis originality

The candidate confirms that the work submitted is his own, except where work which has formed part of jointly-authored publications has been included. The contribution of the candidate and the other authors to this work has been explicitly indicated below. The candidate confirms that appropriate credit has been given within the thesis where reference has been made to the work of others.

The work presented in Chapter II of this thesis has appeared in the following publication:

Putra, S.S., Holden, J. and Baird, A.J. 2021. The effects of ditch dams on water-level dynamics in tropical peatlands. *Hydrological Processes*. **35**(5), article no.: e14174.

In this publication, I was responsible for designing the research, conducting field work, collecting and analysing data, and also composing the manuscript. Joseph Holden and Andy J. Baird were my PhD supervisors, providing advice and critical comments during the research, as well as improving the manuscript.

The work presented in Chapter III of this thesis is under consideration for publication:

Putra, S.S., Holden, J. and Baird, A.J. 2022. Water-table responses to storms in Sebangau tropical peatland, Kalimantan, Indonesia.

In this publication, I was responsible for designing the research, conducting field work, collecting and analysing data, and also composing the manuscript. Joseph Holden and Andy J. Baird were my PhD supervisors, providing advice and critical comments during the research, as well as improving the manuscript.

The work presented in Chapter IV of this thesis has appeared in the following publication:

Putra, S. S., Baird, A. J., & Holden, J. 2022. Modelling the performance of bunds and ditch dams in the hydrological restoration of tropical peatlands. *Hydrological Processes*, **36**(1), article no.: e14470.

In this publication, I was responsible for designing the research, modelling the scenarios, interpreting the results, and also composing the manuscript. Joseph Holden and Andy J. Baird were my PhD supervisors, providing advice and critical comments during the research, as well as improving the manuscript.

Dedications

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Abstract

Tropical peatlands are understudied systems serving as global carbon stores with high biodiversity. Many tropical peatlands have been drained for agriculture leading to their degradation. Restoration of these peatlands often involves blocking drainage ditches, yet little is known about the effectiveness of those initiatives in the recovery of peatland hydrological functions. This study aims to assess the spatial and temporal water-table dynamics in forested, drained, and ditch-dammed tropical peatlands in Sebangau, Kalimantan, Indonesia, investigating variables that contribute to hydrodynamic variability, and modelling possible solutions to improve water-table restoration strategies. Dry-season water tables at all sites were deeper than 40 cm from the surface and ditches had no standing water. In the wet season, the percentage of time during which water tables at wells were deeper than 40 cm from surface was between 16% and 87% at the forested site, from 0% to 38% at the drained site, but 0% at the blocked site. When compared to the forested system, water-table responses to storms were very different at the blocked site, suggesting that it did not function as a natural system. Nevertheless, ditch dams accelerated the water-table rise during the transition from the dry to wet seasons and minimised the hydraulic gradients in the peatland during the wet season. Hydrodynamic modelling of a typical drained peat plot in the study area showed that in the El Niño year, bunds reduce the number of days with water table deeper than 40 cm from the surface by 50% to 73%. A combination of ditch dams and shallow surface reservoirs formed via bunding in restoration plots might help store extra water, reducing the impacts of losses due to evapotranspiration and seepage during the dry season.

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BMKG	Badan Meteorologi, Klimatologi, dan Geofisika di Indonesia, which is translated to Indonesian Meteorology, Climatology, and Geophysics Agency
Bund-d-0cm	results from the Bunded scenario with bund depths of 0 cm
Bund-d-50cm	results from the Bunded scenario with bund depths of 50 cm
Bunded	a typical drained peatland restored with bunds but without dams
Combined	a typical drained peatland restored with dams and bunds
Control	a typical drained peatland without dams and without bunds
d1	ditch cells adjacent to the main outlet of the typical peat plot
d2	ditch cells between Dam 1 and Dam 2 in the typical peat plot
d3	ditch cells between Dam 2 and Dam 3 in the typical peat plot
d4	ditch cells between Dam 3 and Dam 4 in the typical peat plot
Dammed	a typical drained peatland restored with dams but without bunds
d-Bunded	results from the Bunded scenario
d-Combined	results from the Combined scenario
Early Dry	a modelling period from 1 May to 8 August
Early Wet	a modelling period from 1 November to 1 February of the following year
ENSO	El Niño–Southern Oscillation
ET	evapotranspiration
Extended	a scenario that implements bunds with bund depths of 50 cm
ID	identification number
IPCC	Intergovernmental Panel on Climate Change
Late Dry	a modelling period from 1 August to 8 November
Late Wet	a modelling period from 1 February to 11 May
Mod-Dgr	a typical moderately-degraded peatland in Sebangau, Kalimantan, Indonesia
MRP	the Mega Rice Project in Kalimantan, Indonesia
On-Surface	a scenario that implements bunds with bund depths of 0 cm
Sev-Dgr	a typical severely-degraded peatland in Sebangau, Kalimantan, Indonesia
WL	water level

WMO	World Meteorological Organization
wt-diff	the subtraction of the water-table depths of the Extended bund scenario from the Surface bund scenario

Chapter I

An introduction to tropical peatland hydrology

In this chapter, I give an introduction to the thesis, containing an overview of the nature, degradation, and restoration of tropical peatlands. I also justify the need for further understanding of hydrological processes within tropical peatlands to provide better restoration strategies. This chapter consists of eight sections. The first section gives a general overview on peatlands. The second section explains the formation and environmental functions of tropical peatlands, followed by a description of their unique characteristics. In the third and the fourth sections, I provide background about the degradation of tropical peatlands and efforts to restore them. The fifth section addresses the lack of data on peat properties and hydrological behaviour, a key issue that has hindered the understanding of hydrological processes in tropical peatlands. The sixth section explains the aims of the thesis. The seventh section provides an overview of the methodological approaches implemented in this study, while the eighth section sets out the structure of the thesis.

1.1 Introduction to peatlands

Peatlands are ecosystems that develop from the remains of organic material and sustain a waterlogged condition. They are found across a range of latitudes from the sub-Arctic and sub-Antarctic, across the boreal and equatorial zones, and in the tropics (Van der Putten et al., 2009; Page and Baird, 2016; Voigt et al., 2017). Joosten and Clarke (2002) suggested that a landscape can be categorized as a peatland if the soil in that landscape has at least 30% organic materials from its total dry weight and if the organic deposit is at least 30 cm thick. Peat is categorized as a histosol by UN bodies in the World Reference Base for Soil Resources (FAO, 2014).

Peatlands are important for water regulation, influencing the temporal quantity of water supply to downstream rivers and water bodies (Clarke and Rieley, 2019; Langan et al., 2019; Gandois et al., 2020). Peatlands are also important ecosystems for their functions as habitats for various animal and plant species (Troxler et al., 2012; Wijedasa et al., 2020; Harrison, Wijedasa, et al., 2020) but also in storing carbon (Hodgkins et al., 2018; Loisel et al., 2021; Deshmukh et al., 2021). Peatland ecosystems (including peat and vegetation) hold as much as one third of the surface global terrestrial organic carbon store (Scharlemann et al., 2014; Joosten et al., 2016; Harenda et al., 2018). The current best estimate of the global peatland coverage is 4.23 x 10^6 km^2, in which the mid and high latitude northern peatlands cover 3.20 x 10^6 km^2 and tropical peatlands 0.44 x 10^6 km^2 (Page et al., 2011; Xu et al., 2018; Nichols and Peteet, 2019). Present global peatland carbon storage has been estimated as 612 Pg of carbon (Yu et al., 2010). Northern peatlands hold 268 to 562 Pg of carbon, with most commonly accepted estimates around 500 Pg of carbon (Turetsky et al., 2015; Hugelius et al., 2020; Ratcliffe et al., 2021). Tropical peatlands store an estimated additional 100 Pg of carbon, although mapping of previously unidentified tropical peat deposits continues, especially on floodplain, coastal, and montane systems (Taylor, 1990; Donato et al., 2012; Salvador et al., 2014; Hope, 2015; Rieley and Page, 2016; Dargie et al., 2017).

Considering the source of water, peatland can be categorized into two main types, which are bogs and fens. Firstly, bogs (ombrotrophic peatlands) receive most of their water from rainfall. Raised bogs have a mound (convex) shape, usually bounded by water bodies (Clymo, 2017; Campos et al., 2017). The peat thickness at the centre of a raised bog is typically about 4-10 metres higher than at the edge, and the width of the bog can sometimes reach 10000 m or more, particularly in tropical settings (Ritzema et al., 2014; Dargie et al., 2017; Cobb et al., 2020). The water table tends to follow the peatland surface and is higher at the centre of the bog than at the edge. Tropical peat domes are typically much larger than those found in other areas, as raised bogs in the temperate and boreal zones tend to be around 500 to 2000 m in diameter (Humphreys et al., 2014; Hommeltenberg et al., 2014; Pelletier et al., 2015; Strachan et al., 2016; Pellerin et al., 2021). Bogs can also occur on slopes, sometimes as steep as 10 to 25 degrees (Li et al., 2017; Campos et al., 2017), and these are called blanket bogs. Secondly, fens (minerotrophic peatlands) receive water not only from rainfall but also from other sources that contain minerals, such as ground water and river water (Rydin and Jeglum, 2013; Lampela et al., 2014, 2016). The spatial geometry of fens can be very variable, but many are found in depressions in the landscape and on floodplains. In some inner areas of mature pristine domes, floodplains and coastal areas, the landforms can be periodically inundated, resulting in some fen and swamp systems (Posa et al., 2011; Lampela et al., 2016; Kumaran et al., 2016; Clymo, 2017). Over time, a peatland may also go through a transition most commonly from a fen to a bog, given change to the structure of the peatland or external drivers (Page and Baird, 2016; Väliranta et al., 2017). While the simple classification system often suffices, in reality there is a spectrum of peatlands from a pure ombrotrophic and nutrient-poor bog system to minerotrophic rich fens. Some peatlands may contain a mosaic of both fens and bogs (Charman, 2002).

Water-table dynamics in peatlands are closely related to peat growth and crucial to peatland sustainability. Biagioni et al. (2015) and Kelly et al. (2017) noted that water-table behaviour was an important control on the composition and growth rates of the vegetation in tropical peatlands. Page and Baird (2016) highlighted that the water table regulates greenhouse gas emissions (e.g., CO_2 and NH_4) and the chemical composition of peat pore waters (e.g., pH and nutrient availability). Researchers have found that shallow (near-surface) water tables reduce peat decomposition and net greenhouse gas emissions in peatlands (Hirano et al., 2012; Lampela et al., 2014; Ishikura et al., 2019; Huang et al., 2021). The relationship between rainfall and water-table dynamics is relatively well understood in northern peatlands (Evans et al., 1999; Daniels et al., 2008; Haapalehto et al., 2011; Holden et al., 2011; Taminskas et al., 2018). However, there are still few studies that provide information on the water-table dynamics in tropical peatlands.

The hydrological functioning of tropical peatlands may vary from year to year depending on climatic variability. In Southeast Asia, Susilo et al. (2013) reported that El Niño–Southern Oscillation (ENSO) phenomena commonly govern the rainfall characteristics in the tropical peatland areas. However, there is a lack of understanding on how different rainfall characteristics affect the water-table dynamics of tropical peatlands.

1.2 Nature of tropical peatlands

Some tropical peatlands initiated before the Holocene, but it is thought that development and growth was more rapid during the Holocene (Lähteenoja et al., 2012; Morris et al., 2018; Treat et al., 2019; Ruwaimana et al., 2020). Tropical bogs are more common than fens (Joosten, 2016), but there are few reliable data enabling us to establish the proportion of each (Minasny et al., 2019). In Southeast Asia, the initiation of tropical peatlands has been related to the geomorphology of the area and past climatic conditions (Dommain et al., 2011, 2014). The inland tropical peatland in Central

Kalimantan, Indonesia, formed progressively at the onset of the Holocene (Dommain et al., 2014), triggered by post-glacial sea-level rise. The estuary-based peatlands in East Kalimantan, Indonesia, are younger than ~8300 cal BP and their initiation coincides with the accretion of Mahakam River (Dommain et al., 2011). Rapid coastal peatland development across the lowlands of Peninsular Malaysia, Sumatra and Borneo, were induced by Late Holocene sea-level regression and greater total regional rainfalls (Dommain et al., 2011). In Pastaza-Marañón Basin, Amazonia, tropical peatlands started to accumulate between 1975 and 8870 cal BP on a subsiding foreland basin (on broad floodplains of many rivers, adjacent to a mountain belt) (Lähteenoja and Page, 2011; Lähteenoja et al., 2012). In the Congo basin, some of the peatlands along the Congo River rapidly grew between 10554 and 7137 cal BP, but their initiation was much earlier than 11000 BP (Dargie et al., 2017).

The natural development of tropical peatlands takes millennia, as accumulation rates are around 1 mm per year (Table 1.1). The data in Table 1.1 also show that accumulation rates for some Southeast Asian and South American peatlands were higher than in Equatorial African peatlands; however, there is a low number of accumulation rates reported in Equatorial Africa.

Table 1.1 Examples of reported tropical peat accumulation rates. The "ni" means no information.

Location	Current land cover	Rate [mm year^{-1}]	Note
Inland tropical peatland, Central Kalimantan, Indonesia	ni	0.54	The time weighted mean rate within the Holocene (Dommain et al., 2011)
Estuary tropical peatland, East Kalimantan, Indonesia	ni	1.89	
Coastal tropical peatland, across the lowlands of Peninsular Malaysia, Sumatra and Borneo	ni	1.77	
CIMTROP LAHG, Sebangau, Central Kalimantan, Indonesia	Mixed forest	0.15 – 2.55	Long term accumulation to present (Page et al., 2004)
San San Pond Sak tropical peatland, Panama	Mangrove forest	1.20 – 2.60	Long term accumulation to present (Upton et al., 2018)
	Mixed forest	2.00 – 2.90	
	Campnosperma area	2.30	
	Cyperus sawgrass area	1.70	
The Cuvette Centrale swamps, Congo River Basin	Hardwood swamp forest	0.16 – 0.21	Long term accumulation to present (Dargie et al., 2017)
	Palm-dominated swamp forest	0.16 – 0.29	

The rainfall and evapotranspiration rates are key hydrological variables that govern water-table depths in pristine ombrotrophic tropical peatlands. Table 1.2 shows that rainfall in tropical peatlands with humid climate tends to be in excess of 2000 mm per year, whereas continental tropical peatlands such as the Cuvette Centrale peatland may receive much less rainfall. Evapotranspiration rates for some tropical peatlands shown in Table 1.2 are comparable to each other, typically being substantially in excess of 1000 mm per year. Hirano et al. (2015) indicated that the evapotranspiration rates in tropical peatlands were dependent on water table and peatland cover conditions, although more research is required on how land cover influences evapotranspiration from tropical peatlands. Nevertheless, hydrological processes in tropical peatlands are strongly influenced by local climatic variability, such as the El Niño–Southern Oscillation around the Pacific (Rossita et al., 2018), and local geomorphological setting, such as peatlands in the rain shadow area in the eastern region of Minas Gerais, Brazil (Bispo et al., 2016; Silva et al., 2020).

Table 1.2 The variation of rainfall and evapotranspiration rates in several tropical peatlands.

Location	Mean Evapotranspiration (mm $year^{-1}$)	Mean Precipitation (mm $year^{-1}$)	Monitoring Period
Central Kalimantan (Hirano et al., 2015).	1636 ± 53 (pristine forest) 1553 ± 117 (drained peatlands) 1374 ± 75 (burned peatlands)	2384 – 2506	2002 – 2009
The region in the vicinity of Iquitos, around Peruvian Amazonia peatlands (Bruijnzeel, 1990; Marengo, 1998, 2013; Kelly et al., 2014).	> 1500	> 3000	1970 – 2012
The town of Bocas del Toro, 14-22 km distant from the San San Pond Sak peat dome, Panama (Baird et al., 2017; Paton, 2022).	1460	3175	2014
Congo-Brazzaville area (Samba and Nganga, 2012; Burnett et al., 2020)	1172	1723	January 1932 – December 2007 for the rainfall and April 2002 – November 2016 for the evapotranspiration.

Tropical peatlands appear to have high values of peat hydraulic conductivity, compared to peatlands found elsewhere, although there are few tropical datasets enabling full comparisons. Baird et al. (2017) provided hydraulic conductivity values taken around the central area of an undisturbed tropical peatland, Changuinola swamp, in northwest Panama, which had an interquartile range from 27.82 to 69.72 m day^{-1} at 0.6 m depth, and from 5.53 to 8.55 m day^{-1} at 2.5 m depth. Kelly et al. (2014) collected hydraulic conductivity data from three sites in Peruvian Amazonia, with values that ranged from 0.28 to 95.04 m day^{-1} at 0.5 m depth and from 0.23 to 49.25 m day^{-1} at 0.9 m depth. These hydraulic conductivity values were higher than for temperate peatlands at comparable depths. As an example, Kopp et al. (2013) provided hydraulic conductivity values at the Mer Bleue Bog, Ontario, Canada, which were from 0.001 to 10 m day^{-1} (n = 5, SD = 7.17 m day^{-1}) for depths less than 0.55 m, and from 10^{-4} to 10^{-1} m day^{-1} (n = 14, SD = 0.14 m day^{-1}) for depths between 0.55 m and 2 m.

Tropical peatland bulk density values vary with the degree of peat degradation. A study by Könönen et al. (2015) found that in the upper layer of an undrained tropical peatland (at 0.1 m depth) in Sebangau, Kalimantan, Indonesia, the mean bulk density values were 0.13 g cm^{-3} (SD = 0.01 g cm^{-3}), smaller than at a drained forested peatland (mean = 0.17 g cm^{-3}, SD = 0.03 g cm^{-3}) and at a drained less vegetated peatland (mean = 0.20 g cm^{-3}, SD = 0.03 g cm^{-3}). Dargie et al. (2017) measured bulk density values on 372 samples taken at pristine peatlands from the Cuvette Centrale, Central Congo Basin, reporting a mean value of 0.19 g cm^{-3} (SD = 0.06 g cm^{-3}). These reported bulk density values for tropical peatlands are comparable to those in temperate peatlands. The mean peat bulk density for boreal forested peatlands was 0.23 g cm^{-3} (SD = 0.20 g cm^{-3}) in western Siberia (Schulze et al., 2015) and 0.05 g cm^{-3} (SD = 0.01 g cm^{-3}) in southern Finland (Zhang et al., 2020).

1.3 Degradation of tropical peatlands

In the past, based on assessments of the palaeo-record contained in peat cores, tropical peatlands were naturally degraded due to variations in water availability caused by climate change and geomorphological dynamics (Page et al., 2004, 2006; Dommain et al., 2014, 2015; Hapsari et al., 2021). As an example, the growth rates for the inland Central Kalimantan peatland decreased as the sea level declined after 8500 cal BP (before 1 January 1950) and the dry climatic conditions in the Late Holocene (Dommain et al., 2011). In between 7240 and 8000 cal BP, the occurrence of natural fires also damaged tropical peat swamp forests, allowing changes from *Campnosperma sp.* – *Cratoxylum sp.* vegetated peatlands to *Palaquium sp.* – *Cratoxylum sp.* vegetated peatlands (Yulianto and Hirakawa, 2006).

In the present time (~ after 1 January 1950), tropical peatlands have been degraded, mainly due to unsustainable human-induced interference. In Southeast Asia, for example, tropical peatland degradation has resulted from excessive logging, massive expansion of agriculture, uncontrolled drainage (to lower water tables), and wildfires, which have been amplified by socioeconomic conditions, policies, and climate change (Dohong, Aziz, et al., 2017). Logging activities became intense during the 1970s and 1980s in Indonesia, marked by more than 60 million hectares of forest leased to about 579 forest management businesses by the early 1990s, primarily aimed at supplying international demands of high-quality forest products (Brockhaus et al., 2012). There have been similar patterns in neighbouring Southeast Asian countries (Bryan et al., 2013). A distinct example of agricultural expansion in tropical peatlands was the Mega Rice Project which involved creating one million hectares of rice fields in peatlands in Central Kalimantan, Indonesia between 1995 and 1999. The project involved peatland drainage and land-use conversion, led to peat fires and ecological degradation, and failed to support the initial aim of national food security (Mawardi, 2007; Limin et al., 2007; Hoscilo et al., 2011; Ritzema et al., 2014; Blackham et al., 2014; Horton et al., 2021). The peatland degradation in Southeast Asia has been a lesson learned for many stakeholders, such as land-use planning agencies, public health agencies, and logging companies (Rosa et al., 2016; Roucoux et al., 2017; Murdiyarso et al., 2019; Lilleskov et al., 2019), and it is hoped that we can avoid the same catastrophic environmental disaster happening in other tropical peatlands. The approach of reducing tropical peatland degradation while supporting human livelihoods and the economy is still being explored and discussed by local, national, and international stakeholders, such as the Indonesian government, Global Peatlands Initiative, and WWF International (Thornton et al., 2020; Ward et al., 2020, 2021; UNEP, 2021).

Tropical peatlands are very sensitive to artificial drainage (Evans et al., 2014; Baird et al., 2017, Cobb et al., 2020; Urzainki et al., 2020), given drainage introduces a hydraulic head difference between the canal or ditch and the peatland that resulting in deeper water tables in the peatland. The density of ditch and canal drainage schemes varies according to the degree of water-table control required. The width of primary canals in Kalimantan, for example, is up to 25 m and the depth is about 3 to 4.5 m (Page et al., 2009; Sinclair et al., 2020). Primary canals usually stretch for tens of km, draining the surrounding peatlands and collecting drainage water from smaller drainage ditch networks (Vernimmen et al., 2020; Dadap et al., 2021). Secondary canals are smaller, shorter, and shallower than the primary ones, and were created to extend the coverage of drainage in the peatlands (Mawdsley et al., 2009; Blackham et al., 2014). Ditches are from 2 to 4 m wide and 1.5 to 3 m deep, constructed in a grid pattern to drain small peat plots of 0.15 to 0.25 km^2. The depth of canals and ditches tend to decrease over time as a result of subsidence (Evans et al., 2019; Hoyt et al., 2020). In the ex-Mega Rice Project area, about 4400 km of artificial drainage canals and ditches are estimated to be present (Houterman and Ritzema, 2009; Ritzema et al., 2014; Dohong, Aziz, et al., 2017; Ismail et al., 2018). Dadap et al. (2021) estimated that at least 65% of 93,000 km^2 of tropical peatlands investigated in Southeast Asia had drainage networks in them, consisting of canals ≥ 5 m width. Few reports mention artificial drainage networks in African and South American tropical peatlands, yet researchers have warned about the risk of drainage in these areas in the near future if local governments in these areas and international organisations (e.g., United Nations Environment Programme) do not act to prevent it (Gumbricht et al., 2017; Dargie et al., 2019; Lilleskov et al., 2019).

Our understanding of the possible impact of drainage in tropical peatlands is limited, especially in relation to the response of water tables to rainfall. While studies on the interaction between rainfall and water tables in tropical peatlands are available (Wösten et al., 2006; Mezbahuddin et al., 2015; Cobb et al., 2017; Marwanto et al., 2018; Sutikno et al., 2019; Cobb and Harvey, 2019; Deshmukh et al., 2021), none have quantified the contribution of different storm variables (e.g., duration and intensity) to the variation of water tables. The storm response information is important to assess the results of hydrological restoration efforts, to improve models, and to develop climate change adaptation strategies in tropical peatlands, which will be further discussed in Chapter III.

Drainage and lowering of the water table in tropical peatlands lead to very high CO_2 emissions (from peat decay) and peat subsidence (from peat decay and compression).

Couwenberg et al. (2010) reviewed subsidence rates of peat from 12 locations in Southeast Asia to estimate CO_2 emissions, deducing that a 10-cm of additional drainage depth caused increased CO_2 losses of 900 g m^{-2} $year^{-1}$. In an *Acacia* plantation in Sumatra, Indonesia, Hooijer et al. (2012) reported that drainage caused peatland subsidence of 1.42 m (with a 1.78×10^4 g m^{-2} $year^{-1}$ of CO_2 equivalent emissions) in the first 5 years (2001 to 2006), for which half of it occurred during the first year, while more data from the same location suggested much lower rates of subsidence (but still considerable), of 0.043 to 0.044 m per year (Evans et al., 2019). Deshmukh et al. (2021) found for Kampar forested peatland, Sumatra, Indonesia that a mean water table of 0.27 m below the surface (June 2017 until May 2020) was associated with 2×10^3 g m^{-2} $year^{-1}$ of CO_2 equivalent emissions and subsidence at an 'average' rate of 0.033 m $year^{-1}$ (unspecified whether pooled mean or median). They reported that a drained peatland with an average water table 0.66 m below the surface (October 2016 to September 2020) emitted greenhouse gases of 4.38×10^3 g m^{-2} $year^{-1}$ of CO_2 equivalent and subsided at an average rate of 0.042 m per year.

In a review of 91 tropical peatland studies (published from 1997 to 2018), Prananto et al. (2020) calculated that each 10 cm drop of water table in drained agricultural peatlands triggered an extra 370 to 510 g m^{-2} $year^{-1}$ of CO_2 equivalent emissions compared to the condition before the water table dropped. They reported that soil respiration was generally greater in agricultural (median = 5.23×10^3 g m^{-2} $year^{-1}$ of CO_2 equivalent emission) than in less disturbed (median = 3.59×10^3 g m^{-2} $year^{-1}$ of CO_2 equivalent emission) tropical peatlands. In the long term, a study from the North Selangor peat swamp forest, Malaysia, by Cooper et al. (2020) indicated that the estimated soil respiration (~ greenhouse gas emission) rate in 30 years (around 1980 to 2010) was also greater in agricultural (5.44×10^3 g m^{-2} $year^{-1}$ of CO_2 equivalent emission) than in less disturbed (1.87×10^3 g m^{-2} $year^{-1}$ of CO_2 equivalent emission) tropical peatlands. In a study from the Central Kalimantan peat swamp forest, Indonesia, the mean soil respiration rate (during 2002 to 2011) was correspondingly greater in agricultural (2.92×10^3 g m^{-2} $year^{-1}$ of C emission) than in less disturbed (1.11×10^3 g m^{-2} $year^{-1}$ of C emission) tropical peatlands (Ishikura et al., 2017).

Drainage reduces the resilience of tropical peatlands against severe climatic conditions. In drained tropical peatlands in Southeast Asia, there are records of peat fires associated with dry El Niño conditions, such as those in 1997 (~ 1.90×10^{-1} Pg to 2.30×10^{-1} Pg of carbon emissions, see Page et al. (2002)) and in 2015 (~ 2.27×10^{-1} Pg of carbon emissions during September to October 2015, see Huijnen et al. (2016)). Fires eradicate large quantities of near surface peat and modify the topography of the peatland. Burn scars were found to reach an average depth of 0.33 m (SD = 0.18 m) as a result of the 2006 peat fire disaster in Central Kalimantan (Ballhorn et al., 2009; Konecny et al., 2016). The susceptibility to fire of drained peatlands may increase in the future with changes in climate (Andreoli et al., 2017; Fer et al., 2017); fires can also be easily initiated by accidental and deliberate human actions (Herawati and Santoso, 2011; Konecny et al., 2016). On the other hand, as the elevation of drained peatlands becomes lower due to subsidence, drained peatlands are also prone to inundation in the wet season (Sumarga et al., 2016). Changes in regional rainfall patterns (higher in intensity and shorter in duration than current conditions) may also increase the inundation of drained peatlands during the wet season (Feng et al., 2013; Marzuki et al., 2016; Chadwick et al., 2016; Putnam and Broecker, 2017; Mandapaka et al., 2017; Ma et al., 2018).

1.4 Restoration of tropical peatlands

Many stakeholders, including government and environmental NGOs (e.g., Global Peatlands Initiative), have started to discuss peatland conservation and join forces in restoring tropical peatlands. The ‘Brazzaville Declaration’, 21–23 March 2018, was a significant example of multi government cooperation towards tropical peatland conservation, attended by representatives of Democratic Republic of the Congo, Republic of the Congo, Republic of Peru and Republic of Indonesia. In the declaration, the represented governments pledged to develop and implement strategies in combatting peatland losses more effectively (Desai, 2017; International Climate Initiative, 2021). In Indonesia, the government aimed to restore 2.49×10^4 km^2 of degraded peatlands and maintain the water table in tropical peatlands to be shallower than 40 cm below the peat surface, a level incorporated in national regulations and their derivatives (Republic of Indonesia Government, 2016; President of Indonesia, 2020). Many studies have indicated that by maintaining water table near to the peatland surface, the risk of peatland fire can be minimized (Wösten et al., 2006; Page et al., 2009; Evers et al., 2017; Ismail et al., 2021).

In Indonesia, as an example, the common restoration techniques for tropical peatlands have been canal and ditch blocking (with non-permanent dams), canal backfilling, and revegetation (Dohong, Cassiophea, et al., 2017; Dohong et al., 2018). The canal and ditch dams have usually been constructed from mature peat available in the surrounding area, which is then covered by layers of wood and plastics (Ritzema et al., 2014). Permanent dams created by concrete are not feasible for the area, considering the complexity of materials and their transportation to the location, and also the weight of the dams on compressible peat. However, there is a lack of studies on the hydrological impacts of ditch dams in tropical peatlands. Canal back filling, which uses mature peat (sapric) (Houterman and Ritzema, 2009), has been implemented in some canal segments in Indonesia where there are no navigation activities by local people. Revegetation, using native species, has been found to be effective in the areas that were reasonably moist in the dry season and were not extensively inundated in the wet season (Dohong et al., 2018; Harrison, Ottay, et al., 2020; Yuwati et al., 2021). In some areas, based on local ranger observations, the distribution of seedlings from nurseries to the fields for revegetation and agriculture occurred through the canals, meaning there was further reluctance to block canals in those areas.

Tropical peatland restoration is costly, but it is worthwhile compared to the loss due to further degradation. In the case of Indonesian peatlands, Hansson and Dargusch (2018) suggested that around US $4.6 billion of investment should be allocated to restore a 2×10^4 km^2 of drained peatlands between 2015 and 2020, but only US $200 million was reserved in 2017. Sari et al. (2021) noted that the restoration funds reached US $1.70 billion in 2020, following the increased restoration target of 2.49×10^4 km^2. On the other hand, Kiely et al. (2021) calculated that the economic loss due to the 2015 fires in Indonesia reached US$28 billion, whilst the loss due to the six largest peatland fire events from 2004 to 2015 was US$93.9 billion. If the conversion of peatland and peat swamp forests into agricultural land was suspended, the economic loss in annual revenue to local governments (based on 2015 data) was estimated at about US$184 million in Riau and US $100 million in Central Kalimantan (Glauber et al., 2016; Yusuf et al., 2018). Apart from the expensive cost of restoration, it is not clear over what timescale restoration of tropical peatland is achievable and what techniques might be required to enhance water-table restoration once peatlands become degraded. In temperate peatlands, restoration may not recover the hydrological functioning of peatlands in a short term period (Holden et al., 2011; Williamson et al., 2017; Kreyling et al., 2021). Many studies, especially from temperate peatlands, have suggested that full restoration of degraded peatlands might not be possible (Page and Baird, 2016; Price et al., 2016; Alderson et al., 2019).

Aside from the restoration efforts, governments also started to restrict new canal and ditch construction, promoting more sustainable ways of managing tropical peatlands (Padfield et al., 2015; Parish, Lew, et al., 2021; Ward et al., 2021; Applegate et al., 2022). Several studies strongly urge that national and international investors that support agricultural activities in tropical peatlands should move to invest in more sustainable agricultural practices (Ramdani and Lounela, 2020; Parish, Afham, et al., 2021; Astuti, 2021). Moreover, the Indonesian government has banned the use of fire for land clearance.

1.5 Gaps in tropical peatland hydrology studies

Up to this moment, studies on spatial and temporal water-table dynamics as a response to drainage in tropical peatland settings are limited. Many tropical peatland water-table studies have focussed on regional monitoring (Wösten et al., 2006, 2008), while there have been few investigations of processes across individual sites. The main limitations of previous water-table studies are: i) the distance between wells and the distribution of monitoring wells were not designed to expose the effect of individual ditches or ditch dams on water tables (see Ishii et al. (2016)), ii) the time interval of the monitoring data was low resolution (i.e., monthly, see Ritzema et al. (2014)), and iii) the spatial variation of water tables with reference to the position of canals or ditches was not investigated (see Taufik et al. (2020)). People cannot simply apply results from detailed water-table studies in temperate and high-latitude peatlands (Holden et al., 2017; Goodbrand et al., 2019; Harris et al., 2020) to tropical peatland settings because tropical peat properties are somewhat different. Despite the large investment on tropical peatland restoration (Hansson and Dargusch, 2018; Kiely et al., 2021), there are still large knowledge gaps on how the restoration measures may affect water-table dynamics. We have limited understanding of whether ditch dams are effective at restoring hydrological functions of tropical peatlands, the spatial patterns of water tables with respect to ditches and ditch dams. We also have limited understanding of the detailed water-table dynamics in intact (forested) tropical peatlands. There is limited understanding of whether tropical peatland restoration measures, specifically ditch damming, maintain water tables close to the peatland surface at all times of the year (shallower than the legislated Indonesian value of 40 cm).

There are no studies of how water tables respond to individual rainfall events in tropical peatlands and so we cannot evaluate whether restoration practice affects the hydrological functioning of peatlands in response to these events. Improving understanding of the response of water tables to rainfall in tropical peatlands may enhance the performance of hydrology and carbon dynamics models (Mezbahuddin et al., 2014; Farmer et al., 2014; Hoyt et al., 2019; Young et al., 2019).

Equally, the performance of restoration measures may also vary with short-term fluctuations in climate, caused, for example, by the El Niño–Southern Oscillation (ENSO). During dry years there is increased fire risk (Wösten et al., 2006; Page et al., 2009; Evers et al., 2017; Ismail et al., 2021) and so restoration practice may need to be modified to ensure peatlands are resilient across the ENSO cycle. To retain extra water, supplementing ditch dams with surface bunds has been adopted in some initial studies in temperate peatlands (Shantz and Price, 2006; Land and Brock, 2017; Payne et al., 2018; Glenk et al., 2020). Bunds are impermeable or very low permeability barriers that allow rainfall to be stored on the peatland surface. The performance of bunds in tropical settings is still unknown, because no studies have investigated their use.

1.6 Research aims

The aim of the research is to improve understanding of the effects of different types of restoration measure on peatland hydrological functioning. This research focusses on the relationship between rainfall, evapotranspiration, and water tables in typical small-scale tropical peat plots (around 0.2 km^2). This research also assesses the effectiveness of ditch dams and the potential use of bunds as restoration measures for drained tropical peatlands.

The specific research questions are:

1. How do hydrological processes differ across tropical peatlands with different restoration conditions?

2. How do drainage and variations in peat physical properties can affect peatland water-table responses to storm events?
3. How well does the use of bunds in combination with ditch dams maintain the water table near to the peatland surface in different climate conditions?

1.7 Methodological overview

To investigate the water-table dynamics in peatlands with different restoration conditions (Questions 1 and 2), I chose three typical nearby sites for detailed comparative investigation. These were located in an area that was intensively drained during the Ex-Mega Rice Project. Here, a lot of investment is now being put towards restoration which makes it an ideal location for study. The proximity of the sites meant that field visits between sites were feasible for monitoring and sample collection. The three sites consisted of a forested peatland (Forested), a drained peatland with ditch dams (Blocked), and a drained peatland without ditch dams (Drained). The study sites were selected to allow me having comparable spatial settings, some existing meteorological data, and good available information about site history. The sites were designed to have a comparable size, each ~ 0.2 km^2, so that the hydrological dynamics could be studied in enough detail, within resource constraints, to understand spatial patterns associated with ditches and ditch dams.

In the three study sites, the water levels in ditches and water tables in the peat mass were monitored using automatic loggers, to provide high resolution temporal data. Some manual monitoring wells were also operated in Drained/Blocked to give more detailed spatial water-table profiles, but access to Forested was very challenging due to dense vegetation and so spatial data collection was more restricted there. Wells were installed spatially with reference to ditch locations to understand how ditches and ditch damming influenced spatial patterns of water table and their response to rainfall or dry periods. The field monitoring period covered six-months during a typical ENSO Neutral year (WMO, 2019; Becker, 2020). The period included the late dry season of 2019 and the early wet season of 2019, so that the water-table dynamics in the dry season, the transition from the dry season to the wet season and the wet season conditions could be captured efficiently. In order to provide background data about the peat at the three study sites, bulk density and hydraulic conductivity measurements were undertaken in line with the distribution of wells at the sites.

To answer Question 3 on the performance of bunds in combination with ditch dams in maintaining the water table near to the peatland surface, hydrodynamics modelling was implemented. This was considered to be more cost effective than a field trial, and would enable initial feasibility designs to be tested in advance of future application. The model was used for simulating flows and water-table fluctuations in a synthetic but typically arranged drained peat plot that was common in the Ex-Mega Rice Project Area. The implemented scenarios included variations in the degree of peatland degradation, the presence or absence of ditch dams and bunds, the bund depth, and climate. The climate scenarios captured rainfall patterns from three different years, which were El Niño, Neutral and La Niña years.

The hydrodynamic software used in this study is DigiBog_Hydro (Baird et al., 2012; Morris et al., 2012), because DigiBog_Hydro demands modest inputs (i.e., drainable porosity, hydraulic conductivity, net rainfall, and ditch water-level condition). Specifically, DigiBog_Hydro allowed the simulation of bunds and ditch dams in the modelled domain. DigiBog_Hydro can represent spatial variations of peat properties and rapid fluctuations of near-surface water table in peatlands, which would be difficult using other hydrodynamics models, for example Modflow (Reeve et al., 2006; Painter et al., 2008). The boundary conditions in DigiBog_Hydro can be selected and arranged by considering the ditch water-level data collected from the study sites. Furthermore, DigiBog_Hydro was freely available, providing practicality for use in this research project (Baird et al., 2020) and future research within the study region.

1.8 Thesis structure

This thesis is in the University of Leeds's alternative format which means 'by papers'. The three central chapters are formed of journal articles while the fifth chapter contains a synthesis of the findings and a discussion of what my PhD work has achieved and future challenges. Below is a brief overview of subsequent chapters.

Chapter II — The effects of ditch dams on water-level dynamics in tropical peatlands

This chapter deals with the first research question. Using field data, I present and discuss differences in water-level dynamics between forested peatland, drained peatland with dams, and drained peatland without dams. I consider the influence of ditches and ditch dams on the spatial and seasonal patterns of water tables in the study sites. Consideration is also given to how well the three study sites meet with the Indonesian regulatory requirement of a 40 cm water-table depth limit. I make recommendations on how to improve current restoration policies toward a more effective water-table management in tropical peatlands.

Chapter III — Water-table responses to storms in Sebangau tropical peatland, Kalimantan, Indonesia

This chapter addresses the second research question. In particular, it considers how the response of peatland water tables to storm events varies between peatland type, position in the peatland landscape (e.g., with respect to ditches), and peat properties. I investigate which storm variables are strongly correlated to the increase of the water table in the study sites. I also examine the pattern of the water-table drawdown after storm occurrence. I provide some insights on how the water-table responses to storms may change if the storm parameters are different, such as in other ENSO conditions.

Chapter IV — Modelling the performance of bunds and ditch dams in the hydrological restoration of tropical peatlands

Here, I explain how I used a hydrodynamic physically-based model to explore the possible use of bunds to support ditch dams in raising water tables in damaged peatlands. Firstly, I highlight the finding of Chapter II, that the ditch dams alone are not enough to maintain peatland water tables near to the ground surface in the dry season. I present a range of restoration scenarios that were simulated with the model. These scenarios are typical of those in the Sebangau peatland, Kalimantan, Indonesia. The results show that different ditch dam and bunding arrangements influence seasonal and spatial water-table dynamics in tropical peatlands. Bunds are found to be beneficial in maintaining high and stable water tables. I make recommendations on how bunds can be used in future restoration work.

Chapter V — Synthesis and Conclusion

This fifth chapter brings together the findings from the previous three chapters. I summarise connections between the restoration condition of a tropical peatland and its hydrodynamic behaviour. I propose directions in which the restoration measures can be implemented more effectively and provide insights on peatland degradation mitigation strategies given that the future climate is uncertain. I discuss how hydrodynamic modelling can be used in understanding the hydrological responses to restoration measures in peatlands, and to predict the water-table dynamics under different management conditions. The limitations of this research are identified, and new avenues of enquiry are suggested. At the end of the chapter, I highlight some central conclusions of this thesis, putting those in a practical context of tropical peatland restoration planning and assessment.

1.9 References

Alderson, D. M., Evans, M. G., Shuttleworth, E. L., Pilkington, M., Spencer, T., Walker, J. and Allott, T. E. 2019. Trajectories of ecosystem change in restored blanket peatlands. *Science of the Total Environment.* **665**, pp. 785-796.

Andreoli, R.V., de Oliveira, S.S., Kayano, M.T., Viegas, J., de Souza, R.A.F. and Candido, L.A. 2017. The influence of different El Niño types on the South American rainfall. *International Journal of Climatology*. **37**(3), pp.1374–1390.

Applegate, G., Freeman, B., Tular, B., Sitadevi, L. and Jessup, T.C. 2022. Application of agroforestry business models to tropical peatland restoration. *Ambio*. **51**(4), pp.863–874.

Astuti, R. 2021. Governing the ungovernable: The politics of disciplining pulpwood and palm oil plantations in Indonesia's tropical peatland. *Geoforum*. **124**, pp.381–391.

Baird, A.J., Gill, P. and Young, D. 2020. *DigiBog Hydro User Manual*. [Online]. Leeds, UK. Available from: https://water.leeds.ac.uk/wp-content/uploads/sites/36/2020/12/DigiBog_Hydro_user_manual_v1_FINAL.pdf.

Baird, A.J., Low, R., Young, D., Swindles, G.T., Lopez, O.R. and Page, S.E. 2017. High permeability explains the vulnerability of the carbon store in drained tropical peatlands. *Geophysical Research Letters*. **44**(3), pp.1333–1339.

Baird, A.J., Morris, P.J. and Belyea, L.R. 2012. The DigiBog peatland development model 1: rationale, conceptual model, and hydrological basis. *Ecohydrology*. **5**(3), pp.242–255.

Ballhorn, U., Siegert, F., Mason, M. and 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*. **106**(50), pp.21213–21218.

Becker, E. 2020. February 2020 ENSO update: mind reading. *NOAA ENSO Blog*.

[Online], p.1. [Accessed 1 January 2022]. Available from: https://www.climate.gov/news-features/blogs/enso/february-2020-enso-update-mind-reading.

Biagioni, S., Krashevska, V., Achnopha, Y., Saad, A., Sabiham, S. and Behling, H. 2015. 8000 years of vegetation dynamics and environmental changes of a unique inland peat ecosystem of the Jambi Province in Central Sumatra, Indonesia. *Palaeogeography, Palaeoclimatology, Palaeoecology*. **440**, pp.813–829.

Bispo, D.F.A., Silva, A.C., Christofaro, C., Silva, M.L.N., Barbosa, M.S., Silva, B.P.C., Barral, U.M. and Fabris, J.D. 2016. Hydrology and carbon dynamics of tropical peatlands from Southeast Brazil. *CATENA*. **143**, pp.18–25.

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

Brockhaus, M., Obidzinski, K., Dermawan, A., Laumonier, Y. and Luttrell, C. 2012. An overview of forest and land allocation policies in Indonesia: Is the current framework sufficient to meet the needs of REDD+? *Forest Policy and Economics*. **18**, pp.30–37.

Bruijnzeel, L.A. 1990. *Hydrology of moist tropical forests and effects of conversion: a state of knowledge review.* [Online] (L. S. Hamilton, ed.). Amsterdam: UNESCO International Hydrological Programme. Available from: http://unesdoc.unesco.org/images/0009/000974/097405eo.pdf.

Bryan, J.E., Shearman, P.L., Asner, G.P., Knapp, D.E., Aoro, G. and Lokes, B. 2013. Extreme Differences in Forest Degradation in Borneo: Comparing Practices in Sarawak. *PLoS ONE*. **8**(7), article no.: e69679.

Burnett, M.W., Quetin, G.R. and Konings, A.G. 2020. Data-driven estimates of evapotranspiration and its controls in the Congo Basin. *Hydrology and Earth System Sciences*. **24**(8), pp.4189–4211.

Campos, J.R. da R., Christófaro Silva, A., Nanni, M.R., dos Santos, M. and Vidal-Torrado, P. 2017. Influence of the structural framework on peat bog distribution in the tropical highlands of Minas Gerais, Brazil. *CATENA*. **156**, pp.228–236.

Chadwick, R., Good, P., Martin, G. and Rowell, D.P. 2016. Large rainfall changes consistently projected over substantial areas of tropical land. *Nature Climate Change*. **6**(2), pp.177–181.

Charman, D. 2002. *Peatlands and environmental change.* 1st ed. Chichester, UK: John

Wiley & Sons Ltd.

Clarke, D. and Rieley, J.O. 2019. *Strategy for Responsible Peatland Management*. 6th ed. Jyväskylä, Finland: International Peatland Society.

Clymo, R.S.D. 2017. A video simulating the growth of a raised bog. *Mires and Peat*. **19**, pp.1–7.

Cobb, A.R., Dommain, R., Tan, F., Hwee En Heng, N. and Harvey, C.F. 2020. Carbon storage capacity of tropical peatlands in natural and artificial drainage networks. *Environmental Research Letters*. **15**(11), article no.: 114009.

Cobb, A.R. and Harvey, C.F. 2019. Scalar Simulation and Parameterization of Water Table Dynamics in Tropical Peatlands. *Water Resources Research*. **55**(11), pp.9351–9377.

Cobb, A.R., Hoyt, A.M., Gandois, L., Eri, J., Dommain, R., Abu Salim, K., Kai, F.M., Haji Su'ut, N.S. and Harvey, C.F. 2017. How temporal patterns in rainfall determine the geomorphology and carbon fluxes of tropical peatlands. *Proceedings of the National Academy of Sciences*. **114**(26), pp.E5187–E5196.

Cooper, H. V., Evers, S., Aplin, P., Crout, N., Dahalan, M.P. Bin and Sjogersten, S. 2020. Greenhouse gas emissions resulting from conversion of peat swamp forest to oil palm plantation. *Nature Communications*. **11**, article no.: 407.

Couwenberg, J., Dommain, R. and Joosten, H. 2010. Greenhouse gas fluxes from tropical peatlands in south-east Asia. *Global Change Biology*. **16**(6), pp.1715–1732.

Dadap, N.C., Hoyt, A.M., Cobb, A.R., Oner, D., Kozinski, M., Fua, P. V., Rao, K., Harvey, C.F. and Konings, A.G. 2021. Drainage Canals in Southeast Asian Peatlands Increase Carbon Emissions. *AGU Advances*. **2**(1), article no.: e2020AV000321.

Daniels, S. M., Agnew, C. T., Allott, T. E. H. and Evans, M. G. 2008. Water table variability and runoff generation in an eroded peatland, South Pennines, UK. *Journal of Hydrology*. **361**(1-2), pp.214-226.

Dargie, G.C., Lawson, I.T., Rayden, T.J., Miles, L., Mitchard, E.T.A., Page, S.E., Bocko, Y.E., Ifo, S.A. and Lewis, S.L. 2019. Congo Basin peatlands: threats and conservation priorities. *Mitigation and Adaptation Strategies for Global Change*. **24**(4), pp.669–686.

Dargie, G.C., Lewis, S.L., Lawson, I.T., Mitchard, E.T.A., Page, S.E., Bocko, Y.E. and Ifo, S.A. 2017. Age, extent and carbon storage of the central Congo Basin peatland complex. *Nature*. **542**(7639), pp.86–90.

Desai, B.H. 2017. United Nations Environment Program (UNEP). *Yearbook of International Environmental Law*. **28**(14), pp.498–505.

Deshmukh, C.S., Julius, D., Desai, A.R., Asyhari, A., Page, S.E., Nardi, N., Susanto, A.P., Nurholis, N., Hendrizal, M., Kurnianto, S., Suardiwerianto, Y., Salam, Y.W., Agus, F., Astiani, D., Sabiham, S., Gauci, V. and Evans, C.D. 2021. Conservation slows down emission increase from a tropical peatland in Indonesia. *Nature Geoscience*. **14**(7), pp.484–490.

Dohong, A., Aziz, A.A. and Dargusch, P. 2018. A review of techniques for effective tropical peatland restoration. *Wetlands*. **38**(2), pp.275–292.

Dohong, A., Aziz, A.A. and Dargusch, P. 2017. A review of the drivers of tropical peatland degradation in South-East Asia. *Land Use Policy*. **69**, pp.349–360.

Dohong, A., Cassiophea, L., Sutikno, S., Triadi, B.L., Wirada, F., Rengganis, P. and Sigalingging, L. 2017. *Modul pelatihan pembangunan infrastruktur pembasahan gambut sekat kanal berbasis masyarakat (Training Module for community based peatland rewetting infrastructure development)*. [Online]. Jakarta, Indonesia. Available from: https://brg.go.id/files/Publikasi/Modul_Sekat_Kanal_Berbasis_Masyarakat_Final_30082017.pdf.

Dommain, R., Cobb, A.R., Joosten, H., Glaser, P.H., Chua, A.F.L., Gandois, L., Kai, F.-M., Noren, A., Salim, K.A., Su'ut, N.S.H. and Harvey, C.F. 2015. Forest dynamics and tip-up pools drive pulses of high carbon accumulation rates in a tropical peat dome in Borneo (Southeast Asia). *Journal of Geophysical Research: Biogeosciences*. **120**(4), pp.617–640.

Dommain, R., Couwenberg, J., Glaser, P.H., Joosten, H. and Suryadiputra, I.N.N. 2014. Carbon storage and release in Indonesian peatlands since the last deglaciation. *Quaternary Science Reviews*. **97**, pp.1–32.

Dommain, R., Couwenberg, J. and Joosten, H. 2011. Development and carbon sequestration of tropical peat domes in south-east Asia: links to post-glacial sea-level changes and Holocene climate variability. *Quaternary Science Reviews*. **30**(7–8), pp.999–1010.

Donato, D.C., Kauffman, J.B., Mackenzie, R.A., Ainsworth, A. and Pfleeger, A.Z. 2012. Whole-island carbon stocks in the tropical Pacific: Implications for mangrove conservation and upland restoration. *Journal of Environmental*

Management. **97**, pp.89–96.

Evans, M. G., Burt, T. P., Holden, J. and Adamson, J. K. 1999. Runoff generation and water table fluctuations in blanket peat: evidence from UK data spanning the dry summer of 1995. *Journal of Hydrology*. **221**(3-4), pp.141-160.

Evans, C. D., Page, S. E., Jones, T., Moore, S., Gauci, V., Laiho, R., Hruška, J., Allott, T. E. H., Billett, M. F., Tipping, E., Freeman, C. and Garnett, M. H. 2014. Contrasting vulnerability of drained tropical and high- latitude peatlands to fluvial loss of stored carbon. *Global Biogeochemical Cycles*. **28**(11), pp.1215-1234.

Evans, C.D., Williamson, J.M., Kacaribu, F., Irawan, D., Suardiwerianto, Y., Hidayat, M.F., Laurén, A. and Page, S.E. 2019. Rates and spatial variability of peat subsidence in Acacia plantation and forest landscapes in Sumatra, Indonesia. *Geoderma*. **338**, pp.410–421.

Evers, S., Yule, C.M., Padfield, R., O'Reilly, P. and Varkkey, H. 2017. Keep wetlands wet: the myth of sustainable development of tropical peatlands - implications for policies and management. *Global Change Biology*. **23**(2), pp.534–549.

FAO 2014. *World Reference Base for Soil Resources (WRB)*. [Online] 2015 ed. Rome: Food and Agriculture Organization of the United Nations. Available from: http://www.fao.org/soils-portal/soil-survey/soil-classification/world-reference-base/en/.

Farmer, J., Matthews, R., Smith, P. and Smith, J.U. 2014. The Tropical Peatland Plantation-Carbon Assessment Tool: estimating CO_2 emissions from tropical peat soils under plantations. *Mitigation and Adaptation Strategies for Global Change*. **19**(6), pp.863–885.

Feng, X., Porporato, A. and Rodriguez-Iturbe, I. 2013. Changes in rainfall seasonality in the tropics. *Nature Climate Change*. **3**(9), pp.811–815.

Fer, I., Tietjen, B., Jeltsch, F. and Wolff, C. 2017. The influence of El Niño–Southern Oscillation regimes on eastern African vegetation and its future implications under the RCP8.5 warming scenario. *Biogeosciences*. **14**(18), pp.4355–4374.

Gandois, L., Hoyt, A. M., Mounier, S., Le Roux, G., Harvey, C. F., Claustres, A., Nuriman, M. and Anshari, G. 2020. From canals to the coast: dissolved organic matter and trace metal composition in rivers draining degraded tropical peatlands in Indonesia. *Biogeosciences.* **17**(7), pp.1897-1909.

Glauber, A.J., Moyer, S., Adriani, M. and Gunawan, I. 2016. *The Cost of Fire: An Economic Analysis of Indonesia's 2015 Fire Crisis*. [Online]. Jakarta, Indonesia:

World Bank, Jakarta. Available from: https://olc.worldbank.org/content/cost-fire-economic-analysis-indonesia’s-2015-fire-crisis.

Glenk, K., Novo, P., Roberts, M., Sposato, M., Martin-Ortega, J., Shirkhorshidi, M. and Potts, J. 2020. *The costs of peatland restoration: Analysis of an evolving database based on the Peatland Action Programme in Scotland.* [Online]. Edinburgh. Available from: https://research-scotland.ac.uk/handle/20.500.12594/9745.

Goodbrand, A., Westbrook, C.J. and Kamp, G. 2019. Hydrological functions of a peatland in a Boreal Plains catchment. *Hydrological Processes*. **33**(4), pp.562–574.

Gumbricht, T., Roman-Cuesta, R.M., Verchot, L., Herold, M., Wittmann, F., Householder, E., Herold, N. and Murdiyarso, D. 2017. An expert system model for mapping tropical wetlands and peatlands reveals South America as the largest contributor. *Global Change Biology*. **23**(9), pp.3581–3599.

Hansson, A. and Dargusch, P. 2018. An Estimate of the Financial Cost of Peatland Restoration in Indonesia. *Case Studies in the Environment*. **2**(1), pp.1–8.

Haapalehto, T. O., Vasander, H., Jauhiainen, S., Tahvanainen, T. and Kotiaho, J. S. 2011. The effects of peatland restoration on water-table depth, elemental concentrations, and vegetation: 10 years of changes. *Restoration Ecology*. **19**(5), 587-598.

Hapsari, K.A., Biagioni, S., Jennerjahn, T.C., Saad, A., Sabiham, S., Corre, M.D., Veldkamp, E. and Behling, H. 2021. Late Holocene ENSO-related fire impact on vegetation, nutrient status and carbon accumulation of peatlands in Jambi, Sumatra, Indonesia. *Review of Palaeobotany and Palynology*. **293**, article no.: 104482.

Harenda, K.M., Lamentowicz, M., Samson, M. and Chojnicki, B.H. 2018. The Role of Peatlands and Their Carbon Storage Function in the Context of Climate Change *In*: T. Zielinski, I. Sagan and W. Surosz, eds. *Interdisciplinary Approaches for Sustainable Development Goals. GeoPlanet: Earth and Planetary Sciences*. Cham: Springer, pp.169–187.

Harris, L.I., Roulet, N.T. and Moore, T.R. 2020. Drainage reduces the resilience of a boreal peatland. *Environmental Research Communications*. **2**(6), article no.: 065001.

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. and Veen, F.J.F. 2020. Tropical forest and peatland conservation in Indonesia: Challenges and directions. *People and Nature*. **2**(1), pp.4–28.

Harrison, M.E., Wijedasa, L.S., Cole, L.E.S., Cheyne, S.M., Choiruzzad, S.A.B., Chua, L., Dargie, G.C., Ewango, C.E.N., Honorio Coronado, E.N., Ifo, S.A., Imron, M.A., Kopansky, D., Lestarisa, T., O'Reilly, P.J., Van Offelen, J., Refisch, J., Roucoux, K., Sugardjito, J., Thornton, S.A., Upton, C. and Page, S.E. 2020. Tropical peatlands and their conservation are important in the context of COVID-19 and potential future (zoonotic) disease pandemics. *PeerJ*. **8**, article no.: e10283.

Herawati, H. and Santoso, H. 2011. Tropical forest susceptibility to and risk of fire under changing climate: A review of fire nature, policy and institutions in Indonesia. *Forest Policy and Economics*. **13**(4), pp.227–233.

Hirano, T., Kusin, K., Limin, S. and Osaki, M. 2015. Evapotranspiration of tropical peat swamp forests. *Global Change Biology*. **21**(5), pp.1914–1927.

Hirano, T., Segah, H., Kusin, K., Limin, S., Takahashi, H. and Osaki, M. 2012. Effects of disturbances on the carbon balance of tropical peat swamp forests. *Global Change Biology*. **18**(11), pp.3410–3422.

Hodgkins, S.B., Richardson, C.J., Dommain, R., Wang, H., Glaser, P.H., Verbeke, B., Winkler, B.R., Cobb, A.R., Rich, V.I., Missilmani, M., Flanagan, N., Ho, M., Hoyt, A.M., Harvey, C.F., Vining, S.R., Hough, M.A., Moore, T.R., Richard, P.J.H., De La Cruz, F.B., Toufaily, J., Hamdan, R., Cooper, W.T. and Chanton, J.P. 2018. Tropical peatland carbon storage linked to global latitudinal trends in peat recalcitrance. *Nature Communications*. **9**, article no.: 3640.

Holden, J., Green, S.M., Baird, A.J., Grayson, R.P., Dooling, G.P., Chapman, P.J., Evans, C.D., Peacock, M. and Swindles, G. 2017. The impact of ditch blocking on the hydrological functioning of blanket peatlands. *Hydrological Processes*. **31**(3), pp.525–539.

Holden, J., Wallage, Z.E., Lane, S.N. and McDonald, A.T. 2011. Water table dynamics in undisturbed, drained and restored blanket peat. *Journal of Hydrology*. **402**(1–2), pp.103–114.

Hommeltenberg, J., Schmid, H.P., Drösler, M. and Werle, P. 2014. Can a bog drained for forestry be a stronger carbon sink than a natural bog forest? *Biogeosciences*.

11(13), pp.3477–3493.

Hooijer, A., Page, S.E., Jauhiainen, J., Lee, W.A., Lu, X.X., Idris, A. and Anshari, G. 2012. Subsidence and carbon loss in drained tropical peatlands. *Biogeosciences*. **9**(3), pp.1053–1071.

Hope, G.S. 2015. Peat in the mountains of New Guinea. *Mires and Peat*. **15**(13), pp.1–21.

Horton, A.J., Virkki, V., Lounela, A., Miettinen, J., Alibakhshi, S. and Kummu, M. 2021. Identifying Key Drivers of Peatland Fires Across Kalimantan's Ex-Mega Rice Project Using Machine Learning. *Earth and Space Science*. **8**(12), pp.1–17.

Hoscilo, A., Page, S.E., Tansey, K.J. and Rieley, J.O. 2011. Effect of repeated fires on land-cover change on peatland in southern Central Kalimantan, Indonesia, from 1973 to 2005. *International Journal of Wildland Fire*. **20**(4), pp.578–588.

Houterman, J. and Ritzema, H. 2009. *Land and water management in the Ex-Mega Rice Project Area in Central Kalimantan. Technical Report; No. 4*. [Online]. Jakarta. Available from: https://www.researchgate.net/publication/272510959_Land_and_water_management_in_the_Ex-Mega_Rice_Project_Area_in_Central_Kalimantan.

Hoyt, A.M., Chaussard, E., Seppalainen, S.S. and Harvey, C.F. 2020. Widespread subsidence and carbon emissions across Southeast Asian peatlands. *Nature Geoscience*. **13**(6), pp.435–440.

Hoyt, A.M., Gandois, L., Eri, J., Kai, F.M., Harvey, C.F. and Cobb, A.R. 2019. CO_2 emissions from an undrained tropical peatland: Interacting influences of temperature, shading and water table depth. *Global Change Biology*. **25**(9), pp.2885–2899.

Huang, Y., Ciais, P., Luo, Y., Zhu, D., Wang, Y., Qiu, C., Goll, D.S., Guenet, B., Makowski, D., De Graaf, I., Leifeld, J., Kwon, M.J., Hu, J. and Qu, L. 2021. Tradeoff of CO_2 and CH_4 emissions from global peatlands under water-table drawdown. *Nature Climate Change*. **11**(7), pp.618–622.

Hugelius, G., Loisel, J., Chadburn, S., Jackson, R.B., Jones, M., MacDonald, G., Marushchak, M., Olefeldt, D., Packalen, M., Siewert, M.B., Treat, C., Turetsky, M., Voigt, C. and Yu, Z. 2020. Large stocks of peatland carbon and nitrogen are vulnerable to permafrost thaw. *Proceedings of the National Academy of Sciences*. **117**(34), pp.20438–20446.

Huijnen, V., Wooster, M.J., Kaiser, J.W., Gaveau, D.L.A., Flemming, J., Parrington, M., Inness, A., Murdiyarso, D., Main, B. and van Weele, M. 2016. Fire carbon emissions over maritime southeast Asia in 2015 largest since 1997. *Scientific Reports*. **6**, article no.: 26886.

Humphreys, E.R., Charron, C., Brown, M. and Jones, R. 2014. Two Bogs in the Canadian Hudson Bay Lowlands and a Temperate Bog Reveal Similar Annual Net Ecosystem Exchange of CO_2. *Arctic, Antarctic, and Alpine Research*. **46**(1), pp.103–113.

International Climate Initiative 2021. The Global Peatlands Initiative: Assessing, Measuring and Preserving Peat Carbon. *Food and Agriculture Organization of the United Nations*. [Online]. [Accessed 1 February 2021]. Available from: https://www.international-climate-initiative.com/en/details/project/the-global-peatlands-initiative-assessing-measuring-and-preserving-peat-carbon-18_III_096-3039.

Ishii, Y., Koizumi, K., Fukami, H., Yamamoto, K., Takahashi, H., Limin, S.H., Kusin, K., Usup, A. and Susilo, G.E. 2016. Groundwater in Peatland *In*: M. Osaki and N. Tsuji, eds. *Tropical Peatland Ecosystems*. Tokyo: Springer Japan, pp.265–279.

Ishikura, K., Hirata, R., Hirano, T., Okimoto, Y., Wong, G.X., Melling, L., Aeries, E.B., Kiew, F., Lo, K.S., Musin, K.K., Waili, J.W. and Ishii, Y. 2019. Carbon Dioxide and Methane Emissions from Peat Soil in an Undrained Tropical Peat Swamp Forest. *Ecosystems*. **22**(8), pp.1852–1868.

Ishikura, K., Yamada, H., Toma, Y., Takakai, F., Morishita, T., Darung, U., Limin, A., Limin, S.H. and Hatano, R. 2017. Effect of groundwater level fluctuation on soil respiration rate of tropical peatland in Central Kalimantan, Indonesia. *Soil Science and Plant Nutrition*. **63**(1), pp.1–13.

Ismail, I., Haghighi, A.T., Marttila, H., Kurniawan, U., Karyanto, O. and Kløve, B. 2021. Water table variations on different land use units in a drained tropical peatland island of Indonesia. *Hydrology Research*. **52**(6), pp.1372–1388.

Ismail, Karyanto, O. and Aditya, T. 2018. Multi-Sensors Remote Sensing for Mapping and Monitoring Canals on Tropical Peatland *In*: T. R. Nuringtyas, ed. *2018 4th International Conference on Science and Technology (ICST)*. Yogyakarta, Indonesia: IEEE, pp.1–5.

Joosten, H. 2016. Changing Paradigms in the History of Tropical Peatland Research *In*: M. Osaki and N. Tsuji, eds. *Tropical Peatland Ecosystems*. Tokyo: Springer Japan, pp.33–48.

Joosten, H. and Clarke, D. 2002. *Wise use of mires and peatlands*. Devon, UK: International Mire Conservation Group and International Peat Society.

Joosten, H., Sirin, A., Couwenberg, J., Laine, J. and Smith, P. 2016. The role of peatlands in climate regulation *In*: A. Bonn, H. Joosten, M. Evans, R. Stoneman and T. Allott, eds. *Peatland Restoration and Ecosystem Services: Science, Policy and Practice*. Ecological Reviews. Cambridge: Cambridge University Press, pp.63–76.

Kelly, T.J., Baird, A.J., Roucoux, K.H., Baker, T.R., Honorio Coronado, E.N., Ríos, M. and Lawson, I.T. 2014. The high hydraulic conductivity of three wooded tropical peat swamps in northeast Peru: measurements and implications for hydrological function. *Hydrological Processes*. **28**(9), pp.3373–3387.

Kelly, T.J., Lawson, I.T., Roucoux, K.H., Baker, T.R., Jones, T.D. and Sanderson, N.K. 2017. The vegetation history of an Amazonian domed peatland. *Palaeogeography, Palaeoclimatology, Palaeoecology*. **468**, pp.129–141.

Kiely, L., Spracklen, D. V., Arnold, S.R., Papargyropoulou, E., Conibear, L., Wiedinmyer, C., Knote, C. and Adrianto, H.A. 2021. Assessing costs of Indonesian fires and the benefits of restoring peatland. *Nature Communications*. **12**, article no.: 7044.

Konecny, K., Ballhorn, U., Navratil, P., Jubanski, J., Page, S.E., Tansey, K., Hooijer, A., Vernimmen, R. and Siegert, F. 2016. Variable carbon losses from recurrent fires in drained tropical peatlands. *Global Change Biology*. **22**(4), pp.1469–1480.

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

Kopp, B.J., Fleckenstein, J.H., Roulet, N.T., Humphreys, E., Talbot, J. and Blodau, C. 2013. Impact of long-term drainage on summer groundwater flow patterns in the Mer Bleue peatland, Ontario, Canada. *Hydrology and Earth System Sciences*. **17**(9), pp.3485–3498.

Kreyling, J., Tanneberger, F., Jansen, F., van der Linden, S., Aggenbach, C., Blüml, V., Couwenberg, J., Emsens, W.-J., Joosten, H., Klimkowska, A., Kotowski, W., Kozub, L., Lennartz, B., Liczner, Y., Liu, H., Michaelis, D., Oehmke, C., Parakenings, K., Pleyl, E., Poyda, A., Raabe, S., Röhl, M., Rücker, K., Schneider, A., Schrautzer, J., Schröder, C., Schug, F., Seeber, E., Thiel, F., Thiele, S., Tiemeyer, B., Timmermann, T., Urich, T., van Diggelen, R., Vegelin, K., Verbruggen, E., Wilmking, M., Wrage-Mönnig, N., Wołejko, L., Zak, D. and Jurasinski, G. 2021. Rewetting does not return drained fen peatlands to their old selves. *Nature Communications*. **12**, article no.: 5693.

Kumaran, N.K.P., Padmalal, D., Limaye, R.B., S., V.M., Jennerjahn, T. and Gamre, P.G. 2016. Tropical Peat and Peatland Development in the Floodplains of the Greater Pamba Basin, South-Western India during the Holocene C. Li, ed. *PLOS ONE*. **11**(5), article no.: e0154297.

Lähteenoja, O. and Page, S.E. 2011. High diversity of tropical peatland ecosystem types in the Pastaza-Marañón basin, Peruvian Amazonia. *Journal of Geophysical Research*. **116**(G2), article no.: G02025.

Lähteenoja, O., Reátegui, Y.R., Räsänen, M., Torres, D.D.C., Oinonen, M. and Page, S.E. 2012. The large Amazonian peatland carbon sink in the subsiding Pastaza-Marañón foreland basin, Peru. *Global Change Biology*. **18**(1), pp.164–178.

Langan, C., Farmer, J., Rivington, M., Novo, P. and Smith, J. U. 2019. A wetland ecosystem service assessment tool; Development and application in a tropical peatland in Uganda. *Ecological Indicators*. **103**, pp.434-445.

Lampela, M., Jauhiainen, J., Kämäri, I., Koskinen, M., Tanhuanpää, T., Valkeapää, A. and Vasander, H. 2016. Ground surface microtopography and vegetation patterns in a tropical peat swamp forest. *CATENA*. **139**, pp.127–136.

Lampela, M., Jauhiainen, J. and Vasander, H. 2014. Surface peat structure and chemistry in a tropical peat swamp forest. *Plant and Soil*. **382**(1–2), pp.329–347.

Land, D. and Brock, A. 2017. *Peatland Restoration Advice for the Norwegian Environment Agency*. [Online]. London, UK. Available from: https://www.statsforvalteren.no/contentassets/c836cde50e1e448ca6a1897ff76531fe/peatland-restoration-advice-norwegian-ea-17oct17-djl.pdf.

Li, P., Holden, J., Irvine, B. and Mu, X. 2017. Erosion of Northern Hemisphere blanket peatlands under 21st-century climate change. *Geophysical Research Letters*. **44**(8), pp.3615–3623.

Lilleskov, E., McCullough, K., Hergoualc'h, K., del Castillo Torres, D., Chimner, R., Murdiyarso, D., Kolka, R., Bourgeau-Chavez, L., Hribljan, J., del Aguila Pasquel, J. and Wayson, C. 2019. Is Indonesian peatland loss a cautionary tale for Peru? A two-country comparison of the magnitude and causes of tropical peatland degradation. *Mitigation and Adaptation Strategies for Global Change*. **24**(4), pp.591–623.

Limin, S.H., Jentha and Ermiasi, Y. 2007. History of the Development of Tropical Peatland in Central Kalimantan, Indonesia. *Tropics*. **16**(3), pp.291–301.

Loisel, J., Gallego-Sala, A. V., Amesbury, M.J., Magnan, G., Anshari, G., Beilman, D.W., Benavides, J.C., Blewett, J., Camill, P., Charman, D.J., Chawchai, S., Hedgpeth, A., Kleinen, T., Korhola, A., Large, D., Mansilla, C.A., Müller, J., van Bellen, S., West, J.B., Yu, Z., Bubier, J.L., Garneau, M., Moore, T., Sannel, A.B.K., Page, S.E., Väliranta, M., Bechtold, M., Brovkin, V., Cole, L.E.S., Chanton, J.P., Christensen, T.R., Davies, M.A., De Vleeschouwer, F., Finkelstein, S.A., Frolking, S., Gałka, M., Gandois, L., Girkin, N., Harris, L.I., Heinemeyer, A., Hoyt, A.M., Jones, M.C., Joos, F., Juutinen, S., Kaiser, K., Lacourse, T., Lamentowicz, M., Larmola, T., Leifeld, J., Lohila, A., Milner, A.M., Minkkinen, K., Moss, P., Naafs, B.D.A., Nichols, J., O'Donnell, J., Payne, R., Philben, M., Piilo, S., Quillet, A., Ratnayake, A.S., Roland, T.P., Sjögersten, S., Sonnentag, O., Swindles, G.T., Swinnen, W., Talbot, J., Treat, C., Valach, A.C. and Wu, J. 2021. Expert assessment of future vulnerability of the global peatland carbon sink. *Nature Climate Change*. **11**, pp.70–77.

Ma, J., Chadwick, R., Seo, K.-H., Dong, C., Huang, G., Foltz, G.R. and Jiang, J.H. 2018. Responses of the Tropical Atmospheric Circulation to Climate Change and Connection to the Hydrological Cycle. *Annual Review of Earth and Planetary Sciences*. **46**, pp.549–580.

Mandapaka, P. V., Qin, X. and Lo, E.Y.M. 2017. Analysis of spatial patterns of daily precipitation and wet spell extremes in Southeast Asia. *International Journal of Climatology*. **37**(S1), pp.1161–1179.

Marengo, J.A. 1998. Climatología de la zona de Iquítos, Perú R. Kalliola & S. F. Paitan, eds. *Annales Universitatis Turkuensis*. **2**(3), pp.35–57.

Marengo, J.A., Borma, L.S., Rodriguez, D.A., Pinho, P., Soares, W.R. and Alves, L.M. 2013. Recent Extremes of Drought and Flooding in Amazonia: Vulnerabilities and Human Adaptation. *American Journal of Climate Change*. **2**(2), pp.87–96.

Marwanto, S., Watanabe, T., Iskandar, W., Sabiham, S. and Funakawa, S. 2018. Effects of seasonal rainfall and water table movement on the soil solution composition of tropical peatland. *Soil Science and Plant Nutrition*. **64**(3), pp.386–395.

Marzuki, Hashiguchi, H., Kozu, T., Shimomai, T., Shibagaki, Y. and Takahashi, Y. 2016. Precipitation microstructure in different Madden–Julian Oscillation phases over Sumatra. *Atmospheric Research*. **168**, pp.121–138.

Mawardi, I. 2007. Rehabilitasi dan revitalisasi eks proyek pengembangan lahan gambut di Kalimantan Tengah. *Jurnal Teknologi Lingkungan*. **8**(3), pp.287–297.

Mawdsley, N., Giesen, W., Haag, A., Hooijer, A., van der Vat, M., Ichsan, N., Ibie, B.F. and Hidayat, T. 2009. *Strategic peatland rehabilitation plan for block A (north-west) in the Ex-Mega Rice Project Area, Central Kalimantan: A report prepared for the Kalimantan Forests and Climate Partnership. Euroconsult Mott MacDonald/Deltares| Delft Hydraulics*. Palangkaraya, Indonesia.

Mezbahuddin, M., Grant, R.F. and Hirano, T. 2015. How hydrology determines seasonal and interannual variations in water table depth, surface energy exchange, and water stress in a tropical peatland: Modeling versus measurements. *Journal of Geophysical Research: Biogeosciences*. **120**(11), pp.2132–2157.

Mezbahuddin, M., Grant, R.F. and Hirano, T. 2014. Modelling effects of seasonal variation in water table depth on net ecosystem CO_2 exchange of a tropical peatland. *Biogeosciences*. **11**(3), pp.577–599.

Minasny, B., Berglund, Ö., Connolly, J., Hedley, C., de Vries, F., Gimona, A., Kempen, B., Kidd, D., Lilja, H., Malone, B., McBratney, A., Roudier, P., O'Rourke, S., Rudiyanto, Padarian, J., Poggio, L., ten Caten, A., Thompson, D., Tuve, C. and Widyatmanti, W. 2019. Digital mapping of peatlands – A critical review. *Earth-Science Reviews*. **196**, article no.: 102870.

Morris, P.J., Baird, A.J. and Belyea, L.R. 2012. The DigiBog peatland development model 2: ecohydrological simulations in 2D. *Ecohydrology*. **5**(3), pp.256–268.

Morris, P.J., Swindles, G.T., Valdes, P.J., Ivanovic, R.F., Gregoire, L.J., Smith, M.W., Tarasov, L., Haywood, A.M. and Bacon, K.L. 2018. Global peatland initiation driven by regionally asynchronous warming. *Proceedings of the National Academy of Sciences*. **115**(19), pp.4851–4856.

Murdiyarso, D., Lilleskov, E. and Kolka, R. 2019. Tropical peatlands under siege: the need for evidence-based policies and strategies. *Mitigation and Adaptation Strategies for Global Change*. **24**(4), pp.493–505.

Nichols, J.E. and Peteet, D.M. 2019. Rapid expansion of northern peatlands and doubled estimate of carbon storage. *Nature Geoscience*. **12**(11), pp.917–921.

Padfield, R., Waldron, S., Drew, S., Papargyropoulou, E., Kumaran, S., Page, S.E., Gilvear, D., Armstrong, A., Evers, S., Williams, P., Zakaria, Z., Chin, S.Y., Balle Hansen, S., Campos-Arceiz, A., Latif, M.T., Sayok, A. and Tham, M.H. 2015. Research agendas for the sustainable management of tropical peatland in Malaysia.

Environmental Conservation. **42**(1), pp.73–83.

Page, S.E. and Baird, A.J. 2016. Peatlands and Global Change: Response and Resilience. *Annual Review of Environment and Resources*. **41**, pp.35–57.

Page, S.E., Hosciło, A., Wösten, H., Jauhiainen, J., Silvius, M., Rieley, J., Ritzema, H., Tansey, K., Graham, L., Vasander, H. and Limin, S. 2009. Restoration Ecology of Lowland Tropical Peatlands in Southeast Asia: Current Knowledge and Future Research Directions. *Ecosystems*. **12**(6), pp.888–905.

Page, S.E., Rieley, J. and Banks, C. 2011. Global and regional importance of the tropical peatland carbon pool. *Global Change Biology*. **17**(2), pp.798–818.

Page, S.E., Rieley, J.O. and Wüst, R. 2006. Lowland tropical peatlands of Southeast Asia *In*: I. P. Martini, A. M. Cortizas and W. Chesworth, eds. *Developments in Earth Surface Processes*. New York: Elsevier Science, pp.145–172.

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

Page, S.E., Wűst, R.A.J., Weiss, D., Rieley, J.O., Shotyk, W. and Limin, S.H. 2004. 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. *Journal of Quaternary Science*. **19**(7), pp.625–635.

Painter, S., Başağaoğlu, H. and Liu, A. 2008. Robust Representation of Dry Cells in Single-Layer MODFLOW Models. *Ground Water*. **46**(6), pp.873–881.

Parish, F., Afham, A. and Lew, S.Y. (Serena) 2021. Role of the Roundtable on Sustainable Palm Oil (RSPO) in Tropical Peatland Management *In*: M. Osaki, N. Tsuji, N. Foead and J. O. Rieley, eds. *Tropical Peatland Eco-management*. Springer Singapore, pp.509–533.

Parish, F., Lew, S.Y. (Serena) and Mohd Hassan, A.H. 2021. National Strategies on Responsible Management of Tropical Peatland in Malaysia *In*: M. Osaki, N. Tsuji, N. Foead and J. Rieley, eds. *Tropical Peatland Eco-management*. Springer Singapore, pp.677–723.

Paton, S. 2022. *Meteorological and Oceanographic Summary for the Bocas del Toro Research Station 2021*. Bocas del Toro, Panama: Smithsonian Tropical Research Institute.

Payne, R.J., Anderson, A.R., Sloan, T., Gilbert, P., Newton, A., Ratcliffe, J., Mauquoy, D., Jessop, W. and Andersen, R. 2018. The future of peatland forestry in Scotland:

balancing economics, carbon and biodiversity. *Scottish Forestry*. **72**(1), pp.34–40.

Pellerin, S., Lavoie, M. and Talbot, J. 2021. Rapid broadleave encroachment in a temperate bog induces species richness increase and compositional turnover. *Écoscience*. **28**(3–4), pp.283–300.

Pelletier, L., Strachan, I.B., Roulet, N.T. and Garneau, M. 2015. Can boreal peatlands with pools be net sinks for CO_2? *Environmental Research Letters*. **10**(3), p.035002.

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

Prananto, J.A., Minasny, B., Comeau, L., Rudiyanto, R. and Grace, P. 2020. Drainage increases CO 2 and N 2 O emissions from tropical peat soils. *Global Change Biology*. **26**(8), pp.4583–4600.

President of Indonesia 2020. *President of the Republic of Indonesia Regulation No. 120 Year 2020 about Peatland Restoration and Mangrove Agency (BRGM)*. Ministry of State Secretariat, Indonesia.

Price, J., Evans, C., Evans, M., Allott, T. and Shuttleworth, E. 2016. Peatland restoration and hydrology. In A. Bonn, T. Allott, M. Evans, H. Joosten, and R. Stoneman (Eds.), *Peatland Restoration and Ecosystem Services: Science, Policy and Practice*. Cambridge: Cambridge University Press, Ecological Reviews, pp. 77-94.

Putnam, A.E. and Broecker, W.S. 2017. Human-induced changes in the distribution of rainfall. *Science Advances*. **3**(5), article no.: e1600871.

Van der Putten, N., Verbruggen, C., Ochyra, R., Spassov, S., de Beaulieu, J.-L., De Dapper, M., Hus, J. and Thouveny, N. 2009. Peat bank growth, Holocene palaeoecology and climate history of South Georgia (sub-Antarctica), based on a botanical macrofossil record. *Quaternary Science Reviews*. **28**(1–2), pp.65–79.

Ramdani, R. and Lounela, A.K. 2020. Palm oil expansion in tropical peatland: Distrust between advocacy and service environmental NGOs. *Forest Policy and Economics*. **118**, article no.: 102242.

Ratcliffe, J.L., Peng, H., Nijp, J.J. and Nilsson, M.B. 2021. Lateral expansion of northern peatlands calls into question a 1,055 GtC estimate of carbon storage. *Nature Geoscience*. **14**(7), pp.468–469.

Reeve, A.S., Evensen, R., Glaser, P.H., Siegel, D.I. and Rosenberry, D. 2006. Flow path oscillations in transient ground-water simulations of large peatland systems. *Journal of Hydrology*. **316**(1–4), pp.313–324.

Republic of Indonesia Government 2016. *Indonesian Government Regulation No. 57 Year 2016 about Peatland Ecosystem Protection and Management*. Jakarta, Indonesia: Ministry of State Secretariat, Indonesia.

Rieley, J.O. and Page, S.E. 2016. Tropical Peatland of the World *In*: M. Osaki and N. Tsuji, eds. *Tropical Peatland Ecosystems*. Tokyo: Springer Japan, pp.3–32.

Ritzema, H., Limin, S., Kusin, K., Jauhiainen, J. and Wösten, H. 2014. Canal blocking strategies for hydrological restoration of degraded tropical peatlands in Central Kalimantan, Indonesia. *CATENA*. **114**, pp.11–20.

Rosa, I.M.D., Smith, M.J., Wearn, O.R., Purves, D. and Ewers, R.M. 2016. The Environmental Legacy of Modern Tropical Deforestation. *Current Biology*. **26**(16), pp.2161–2166.

Rossita, A., Witono, A., Darusman, T., Lestari, D.P. and Risdiyanto, I. 2018. Water table depth fluctuations during ENSO phenomenon on different tropical peat swamp forest land covers in Katingan, Indonesia *In*: S. K. Panda, K. Noda, A. Kent, K. Uddin, M. Z. Khan and K. Komariah, eds. *International Conference on Climate Change: Challenges and Opportunity on Environment Degradation Researches 24–26 October 2017*. Surakarta, Indonesia: IOP Conference Series: Earth and Environmental Science, article no.: 012001.

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. and Vriesendorp, C.F. 2017. Threats to intact tropical peatlands and opportunities for their conservation. *Conservation Biology*. **31**(6), pp.1283–1292.

Ruwaimana, M., Anshari, G.Z., Silva, L.C.R. and Gavin, D.G. 2020. The oldest extant tropical peatland in the world: a major carbon reservoir for at least 47 000 years. *Environmental Research Letters*. **15**(11), article no.: 114027.

Rydin, H. and Jeglum, J.K. 2013. *The Biology of Peatlands* 2nd ed. Oxford: Oxford University Press.

Salvador, F., Monerris, J. and Rochefort, L. 2014. Peatlands of the Peruvian Puna ecoregion: types, characteristics and disturbance. *Mires and Peat*. **15**, pp.1–17.

Samba, G. and Nganga, D. 2012. Rainfall variability in Congo-Brazzaville: 1932-2007. *International Journal of Climatology*. **32**(6), pp.854–873.

Sari, A.P., Dohong, A. and Wardhana, B. 2021. Innovative Financing for Peatland Restoration in Indonesia *In*: R. Djalante, J. Jupesta and E. Aldrian, eds. *Climate Change Research, Policy and Actions in Indonesia*. Springer Climate, pp.247–264.

Scharlemann, J.P., Tanner, E.V., Hiederer, R. and Kapos, V. 2014. Global soil carbon: understanding and managing the largest terrestrial carbon pool. *Carbon Management*. **5**(1), pp.81–91.

Schulze, E.D., Lapshina, E., Filippov, I., Kuhlmann, I. and Mollicone, D. 2015. Carbon dynamics in boreal peatlands of the Yenisey region, western Siberia. *Biogeosciences*. **12**(23), pp.7057–7070.

Shantz, M.A. and Price, J.S. 2006. Characterization of surface storage and runoff patterns following peatland restoration, Quebec, Canada. *Hydrological Processes*. **20**(18), pp.3799–3814.

Silva, A.C., Horàk-Terra, I., Barral, U.M., Costa, C.R., Gonçalves, S.T., Pinto, T., Christófaro Silva, B.P., Cunha Fernandes, J.S., Mendonça Filho, C.V. and Vidal-Torrado, P. 2020. Altitude, vegetation, paleoclimate, and radiocarbon age of the basal layer of peatlands of the Serra do Espinhaço Meridional, Brazil. *Journal of South American Earth Sciences*. **103**, article no.: 102728.

Sinclair, A.L., Graham, L.L.B., Putra, E.I., Saharjo, B.H., Applegate, G., Grover, S.P. and Cochrane, M.A. 2020. Effects of distance from canal and degradation history on peat bulk density in a degraded tropical peatland. *Science of The Total Environment*. **699**, article no.: 134199.

Strachan, I.B., Pelletier, L. and Bonneville, M.-C. 2016. Inter-annual variability in water table depth controls net ecosystem carbon dioxide exchange in a boreal bog. *Biogeochemistry*. **127**, pp.99–111.

Sumarga, E., Hein, L., Hooijer, A. and Vernimmen, R. 2016. Hydrological and economic effects of oil palm cultivation in Indonesian peatlands. *Ecology and Society*. **21**(2), article no.: 52.

Susilo, G.E., Yamamoto, K., Imai, T., Ishii, Y., Fukami, H. and Sekine, M. 2013. The effect of ENSO on rainfall characteristics in the tropical peatland areas of Central Kalimantan, Indonesia. *Hydrological Sciences Journal*. **58**(3), pp.539–548.

Sutikno, S., Rinaldi, R., Saputra, E., Kusairi, M., Saharjo, B.H. and Putra, E.I. 2019. Water management for hydrological restoration and fire prevention in tropical peatland *In*: T. B. Aulia, ed. *The 2nd Aceh International Symposium on Civil Engineering (AISCE) ,18-19 September 2019*. Banda Aceh, Indonesia: IOP Conference Series: Materials Science and Engineering, article no.: 012053.

Taminskas, J., Linkevičienė, R., Šimanauskienė, R., Jukna, L., Kibirkštis, G. and Tamkevičiūtė, M. 2018. Climate change and water table fluctuation: implications for raised bog surface variability. *Geomorphology*. **304**, 40-49.

Taufik, M., Minasny, B., McBratney, A.B., Van Dam, J.C., Jones, P.D. and Van Lanen, H.A.J. 2020. Human-induced changes in Indonesian peatlands increase drought severity. *Environmental Research Letters*. **15**(8), article no.: 084013.

Taylor, D.M. 1990. Late quaternary pollen records from two Ugandan mires: evidence for environmental changes in the Rukiga Highlands of southwest Uganda. *Palaeogeography, Palaeoclimatology, Palaeoecology*. **80**(3–4), pp.283–300.

Thornton, S.A., Setiana, E., Yoyo, K., Dudin, Yulintine, Harrison, M.E., Page, S.E. and Upton, C. 2020. Towards biocultural approaches to peatland conservation: The case for fish and livelihoods in Indonesia. *Environmental Science & Policy*. **114**, pp.341–351.

Treat, C.C., Kleinen, T., Broothaerts, N., Dalton, A.S., Dommain, R., Douglas, T.A., Drexler, J.Z., Finkelstein, S.A., Grosse, G., Hope, G., Hutchings, J., Jones, M.C., Kuhry, P., Lacourse, T., Lähteenoja, O., Loisel, J., Notebaert, B., Payne, R.J., Peteet, D.M., Sannel, A.B.K., Stelling, J.M., Strauss, J., Swindles, G.T., Talbot, J., Tarnocai, C., Verstraeten, G., Williams, C.J., Xia, Z., Yu, Z., Väliranta, M., Hättestrand, M., Alexanderson, H. and Brovkin, V. 2019. Widespread global peatland establishment and persistence over the last 130,000 y. *Proceedings of the National Academy of Sciences*. **116**(11), pp.4822–4827.

Troxler, T.G., Ikenaga, M., Scinto, L., Boyer, J.N., Condit, R., Perez, R., Gann, G.D. and Childers, D.L. 2012. Patterns of Soil Bacteria and Canopy Community Structure Related to Tropical Peatland Development. *Wetlands*. **32**(4), pp.769–782.

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

UNEP 2021. *Peatlands in spotlight at COP26*. [Online]. [Accessed 24 June 2022]. Glasgow, UK. Available from: https://www.unep.org/news-and-stories/story/peatlands-spotlight-cop26.

Upton, A., Vane, C.H., Girkin, N., Turner, B.L. and Sjögersten, S. 2018. Does litter input determine carbon storage and peat organic chemistry in tropical peatlands? *Geoderma*. **326**, pp.76–87.

Urzainki, I., Laurén, A., Palviainen, M., Haahti, K., Budiman, A., Basuki, I., Netzer, M. and Hökkä, H. 2020. Canal blocking optimization in restoration of drained peatlands. *Biogeosciences*. **17**(19), pp.4769–4784.

Väliranta, M., Salojärvi, N., Vuorsalo, A., Juutinen, S., Korhola, A., Luoto, M. and Tuittila, E.-S. 2017. Holocene fen–bog transitions, current status in Finland and future perspectives. *The Holocene*. **27**(5), pp.752–764.

Vernimmen, R., Hooijer, A., Mulyadi, D., Setiawan, I., Pronk, M. and Yuherdha, A.T. 2020. A New Method for Rapid Measurement of Canal Water Table Depth Using Airborne LiDAR, with Application to Drained Peatlands in Indonesia. *Water*. **12**(5), article no.: 1486.

Voigt, C., Marushchak, M.E., Lamprecht, R.E., Jackowicz-Korczyński, M., Lindgren, A., Mastepanov, M., Granlund, L., Christensen, T.R., Tahvanainen, T., Martikainen, P.J. and Biasi, C. 2017. Increased nitrous oxide emissions from Arctic peatlands after permafrost thaw. *Proceedings of the National Academy of Sciences*. **114**(24), pp.6238–6243.

Ward, C., Stringer, L.C., Warren-Thomas, E., Agus, F., Crowson, M., Hamer, K., Hariyadi, B., Kartika, W.D., Lucey, J., McClean, C., Nurida, N.L., Petorelli, N., Pratiwi, E., Saad, A., Andriyani, R., Ariani, T., Sriwahyuni, H. and Hill, J.K. 2021. Smallholder perceptions of land restoration activities: rewetting tropical peatland oil palm areas in Sumatra, Indonesia. *Regional Environmental Change*. **21**, article no.: 1.

Ward, C., Stringer, L.C., Warren-Thomas, E., Agus, F., Hamer, K., Pettorelli, N., Hariyadi, B., Hodgson, J., Kartika, W.D., Lucey, J., McClean, C., Nurida, N.L., Saad, A. and Hill, J.K. 2020. Wading through the swamp: what does tropical peatland restoration mean to national-level stakeholders in Indonesia? *Restoration Ecology*. **28**(4), pp.817–827.

Wijedasa, L.S., Vernimmen, R., Page, S.E., Mulyadi, D., Bahri, S., Randi, A., Evans, T.A., Lasmito, Priatna, D., Jensen, R.M. and Hooijer, A. 2020. Distance to forest, mammal and bird dispersal drive natural regeneration on degraded tropical peatland. *Forest Ecology and Management*. **461**, article no.: 117868.

Williamson, J., Rowe, E., Reed, D., Ruffino, L., Jones, P., Dolan, R., Buckingham, H., Norris, D., Astbury, S. and Evans, C.D. 2017. Historical peat loss explains limited short-term response of drained blanket bogs to rewetting. *Journal of Environmental Management*. **188**, pp.278–286.

WMO 2019. *The WMO El Niño/La Niña Update 29 November 2019*. [Online]. Geneva, Switzerland, Switzerland. Available from: https://community.wmo.int/activity-areas/climate/wmo-el-ninola-nina-updates#archive.

Wösten, H., Van Den Berg, J., Van Eijk, P., Gevers, G.J.M., Giesen, W.B.J.T., Hooijer, A., Idris, A., Leenman, P.H., Rais, D.S., Siderius, C., Silvius, M.J., Suryadiputra, N. and Wibisono, I.T. 2006. Interrelationships between Hydrology and Ecology in Fire Degraded Tropical Peat Swamp Forests. *International Journal of Water Resources Development*. **22**(1), pp.157–174.

Wösten, H., Clymans, E., Page, S.E., Rieley, J.O. and Limin, S.H. 2008. Peat–water interrelationships in a tropical peatland ecosystem in Southeast Asia. *CATENA*. **73**(2), pp.212–224.

Xu, J., Morris, P.J., Liu, J. and Holden, J. 2018. PEATMAP: Refining estimates of global peatland distribution based on a meta-analysis. *CATENA*. **160**, pp.134–140.

Young, D.M., Baird, A.J., Charman, D.J., Evans, C.D., Gallego-Sala, A. V., Gill, P.J., Hughes, P.D.M., Morris, P.J. and Swindles, G.T. 2019. Misinterpreting carbon accumulation rates in records from near-surface peat. *Scientific Reports*. **9**, article no.: 17939.

Young, D.M., Baird, A.J., Morris, P.J. and Holden, J. 2017. Simulating the long-term impacts of drainage and restoration on the ecohydrology of peatlands. *Water Resources Research*. **53**(8), pp.6510–6522.

Yu, Z., Loisel, J., Brosseau, D.P., Beilman, D.W. and Hunt, S.J. 2010. Global peatland dynamics since the Last Glacial Maximum. *Geophysical Research Letters*. **37**(13), article no.: L13402.

Yulianto, E. and Hirakawa, K. 2006. Vegetation and environmental change in the early-Middle holocene at a tropical peat swamp forest, Central Kalimantan, Indonesia. *Tropics*. **15**(1), pp.65–73.

Yusuf, A.A., Roos, E.L. and Horridge, J.M. 2018. Indonesia's Moratorium on Palm Oil Expansion from Natural Forests: Economy-Wide Impacts and the Role of International Transfers. *Asian Development Review*. **35**(2), pp.85–112.

Yuwati, T.W., Rachmanadi, D., Qirom, M.A., Santosa, P.B., Kusin, K. and Tata, H.L. 2021. Peatland Restoration in Central Kalimantan by Rewetting and Rehabilitation with Shorea balangeran *In*: M. Osaki, N. Tsuji, N. Foead and J. Rieley, eds. *Tropical Peatland Eco-management*. Springer Singapore, pp.595–611.

Zhang, H., Väliranta, M., Piilo, S., Amesbury, M.J., Aquino-López, M.A., Roland, T.P., Salminen-Paatero, S., Paatero, J., Lohila, A. and Tuittila, E. 2020. Decreased carbon accumulation feedback driven by climate-induced drying of two southern boreal bogs over recent centuries. *Global Change Biology*. **26**(4), pp.2435–2448.

Chapter II

The effects of ditch dams on water-level dynamics in tropical peatlands

Abstract

A significant proportion of tropical peatlands has been drained for agricultural purposes, resulting in severe degradation. Hydrological restoration, which usually involves blocking ditches, is therefore a priority. Nevertheless, the influence of ditch blocking on tropical peatland hydrological functioning is still poorly understood. We studied water-level dynamics using a combination of automated and manual dipwells, and also meteorological data during dry and wet seasons over 6 months at three locations in Sebangau National Park, Kalimantan, Indonesia. The locations were a forested peatland (Forested), a drained peatland with ditch dams (Blocked), and a drained peatland without ditch dams (Drained). In the dry season, water tables at all sites were deeper than the Indonesian regulatory requirement of 40 cm from the peat surface. In the dry season, the ditches were dry, and water did not flow to them. The dry season water-table drawdown rates – solely due to evapotranspiration – were 9.3 mm day^{-1} at Forested, 9.6 mm day^{-1} at Blocked, but 12.7 mm day^{-1} at Drained. In the wet season, the proportion of time during which water tables in the wells were deeper than the 40 cm limit ranged between 16% and 87% at Forested, 0% at Blocked, and between 0% and 38% at Drained. In the wet season, water flowed from the peatland to ditches at Blocked and Drained. The interquartile range of hydraulic gradients between the lowest ditch outlet and the farthest well from ditches at Blocked was 3.7×10^{-4} to 7.8×10^{-4} m m^{-1}, but 1.9×10^{-3} to 2.6×10^{-3} m m^{-1} at Drained. Given the results from Forested, a water-table depth limit policy based on field data may be required, to reflect natural seasonal dynamics in tropical peatlands. Revised spatial designs of dams or bunds are also required, to ensure effective water-table management as part of tropical peatland restoration.

2.1 Introduction

Tropical peatlands cover around 1 million km^2 (Xu et al., 2018) and are unique ecosystems that contribute to global carbon storage and biodiversity (Posa et al., 2011; Page et al., 2011; Warren et al., 2017; Harrison et al., 2020). Tropical peatlands can form in a range of settings where there is poor drainage and organic matter can accumulate over time (Dommain et al., 2011; Kurnianto et al., 2015). Domed peatlands are common, mainly supplied by rainfall, with water being lost through evapotranspiration and lateral flow into streams at the edge of the peat mass (Dommain et al., 2010; Kelly et al., 2014; Hirano et al., 2015; Dargie et al., 2017). Many tropical peatlands have been artificially drained using ditch networks as part of site preparation for plantations or other agricultural activities (Hooijer et al., 2012; Dohong, Aziz, et al., 2017). These peatlands are thought to be susceptible to very rapid degradation following such artificial drainage, due to relatively high hydraulic conductivity (Baird et al., 2017). Drainage causes water-table drawdown in tropical peatlands (Limin et al., 2007; Wösten et al., 2008), which may cause subsidence due to consolidation associated with peat water loss, increased peat bulk density, and additional mass loss due to oxic decay (Evans et al., 2019; Kurnianto et al., 2019; Sinclair et al., 2020; Hoyt et al., 2020). However, there is a lack of studies about the effects of drainage on spatial and temporal water-level dynamics across tropical peatlands. Such knowledge would be useful to support our understanding of tropical peatland degradation and restoration after drainage.

Drainage of tropical peatlands has also been associated with enhanced fire risk (Page et al., 2009; Hoscilo et al., 2011), though land use and land cover are important factors that also contribute to this risk (Cattau et al., 2016; Uda et al., 2017; Dohong et al., 2018a). There are major negative impacts of tropical peatland fires. Higher carbon emissions have been reported from tropical peatlands converted for rice or palm oil production than from natural or secondary forest peatlands (Inubushi et al., 2003; Murdiyarso et al., 2010; Prananto et al., 2020). Aside from very high rates of CO_2 emissions, peat fire also causes biodiversity loss, economic loss, and human respiratory problems (Ballhorn et al., 2009; Glauber et al., 2016; Wooster et al., 2018; Agus et al., 2019; Uda et al., 2019). As a result of negative impacts of peatland drainage, some tropical countries are supporting conservation and restoration efforts on drained peatlands. For example, the Indonesian government has stipulated that peat water tables may not be deeper than 40 cm from the surface (the 40 cm limit), which is thought to be a severe fire risk threshold, through the peatland regulation initiative (President of the Republic of Indonesia Regulation No. 120 Year 2020 about Peatland Restoration and Mangrove Agency [BRGM], 2020; Republic of Indonesia Government Regulation No. 57 Year 2016 about Peatland Ecosystem Protection and Management, 2016).

Figure 2.1 Drain blocking dam in the study site, Sebangau, Indonesia.

The Indonesian Agency of Peatland Restoration and Mangrove (BRGM) has identified three main aspects to restoration, which are rewetting (including ditch blocking at intervals with dams and ditch infilling), revegetation, and revitalisation (including protection of the local economy and fire risk reduction; Dohong et al., 2018; Dohong, Cassiophea, et al., 2017; Harrison et al., 2020); President of the Republic of Indonesia Regulation No. 120 Year 2020 about Peatland Restoration and Mangrove Agency [BRGM], 2020). In Indonesia, ditch dams are mostly built from locally sourced wood and mature peat (Figure 2.1), and are smaller than dams across the main canals into which the ditch networks drain (Ritzema et al., 2014). Ditch dams are most commonly constructed in conservation areas. The dam core is often covered by durable plastic to stop seepage. Ditch infilling, which uses mature peat (Giesen and Sari, 2018), is often implemented in fire prone areas, where the ditches are not used by local people for navigation. Revegetation measures use local species (Lampela et al., 2017; Wijedasa et al., 2020), and tend to be conducted in previously forested areas that have been subjected to fire or have been affected by drainage. Revitalization encourages sustainable local economic growth so that the risk of anthropogenic fire can be minimized (Puspitaloka et al., 2020; Harrison et al., 2020; Sari et al., 2021). Despite the high costs of tropical peatland restoration measures, it is still not known how the different strategies affect water-level dynamics in different tropical peatland settings. There may be limitations to rewetting schemes due to changes in peat physical properties, topography, and vegetation, brought about by peatland degradation (Lampela et al., 2016; Kelly et al., 2017; Roucoux et al., 2017) and ongoing human activity (Wijedasa et al., 2017; Dohong et al., 2018a; Harrison et al., 2020).

Measurements of the effects of ditch dams on tropical peatland water-level dynamics are scarce. While there are detailed water-level dynamics studies in temperate and boreal peatlands (Holden et al., 2017; Goodbrand et al., 2019; Harris et al., 2020), these are lacking in natural, drained and restoration sites in tropical settings. To date, tropical peatland water-level studies have focussed on regional monitoring rather than on processes occurring across individual sites (Wösten et al., 2006, 2008). Taufik et al. (2020) studied six tropical peatlands with different land uses and in different locations. However, only one groundwater-level monitoring point was installed at each site, so distance effects from ditches or other features could not be determined. Ritzema et al. (2014) measured water-level dynamics around a drained tropical peatland, before and after the canals had been blocked, using 11 observation wells up to a distance of 500 m from the canals. Monthly water tables after canal blocking were shallower than before blocking, but more detailed water-table dynamics could not be inferred from the monthly data. Ishii et al. (2016) used data from 32 monthly manual water-level monitoring wells along a 14-km transect, covering part of the Sebangau Peat Dome, but the resolution was too coarse to describe the effect of individual ditches or ditch dams on water tables (Jaenicke et al., 2010; Dohong et al., 2018b). All these studies recommended that more detailed spatial and temporal information on water-table dynamics in drained and in less disturbed sites was needed, not only to better understand the hydrological dynamics and functioning of tropical peatland, but also to provide evidence on how to mitigate drainage impacts and aid effective restoration.

This chapter examines the effects of ditch dams on tropical peatland water-level dynamics, by comparing temporal patterns across fine spatial scales within and between sites. The main research questions are:

1. How do water-level dynamics vary between forested peatland, drained peatland with dams and drained peatland without dams?
2. How do ditches and ditch dams influence spatial patterns of tropical peatland water-levels in different seasons?
3. How do water-table residence times of the studied peatlands conform to the Indonesian regulatory requirement of a 40 cm water-table depth limit?

2.2 Data and methods

2.2.1 Study sites

Three tropical peatland sites were chosen for this study, located in Sebangau National Park (SNP), Kalimantan, Indonesia (Figure 2.2). Those areas were a forested peatland that had a single narrow 60 cm deep (but dammed) trench, historically used by locals to evacuate logs, hereafter referred to as 'Forested' (11.4 hectares), a drained peatland with dams, referred to as 'Blocked' (18.4 hectares), and a drained peatland without dams, referred to as 'Drained' (15.5 hectares). The geomorphology of these three locations was similar, with a broadly flat terrain with microtopographic variations (Dommain et al., 2010; Lampela et al., 2016). The area is underlain by Miocene siliciclastic sedimentary rocks (Page et al., 2004; Witts et al., 2012). The Sebangau tropical bog system is located between the Katingan River and the Kahayan River, and dissected by the Sebangau River. Satellite delineation combined with field measurements suggest that the Sebangau peat dome covers an area of 7,347 km^2 and stores 2.30 ± 0.46 Pg of carbon (Jaenicke et al., 2008). The mean peat depth at the Sebangau peat dome is 5.40 ± 1.08 m, which may reach to 9.8 m depth in some places (Page et al., 2004; Jaenicke et al., 2008). There are two seasons in this area: a dry season (May–October) and a wet season (November–April). Localized rainfall may still occur in May and June. The typical annual rainfall in Sebangau is between 2,700 and 3,300 mm (Page et al., 2004; Itakura et al., 2016). The mean annual temperature ranges from around 26.2 to 28.1°C (Hirano et al., 2014; Rahajoe et al., 2016).

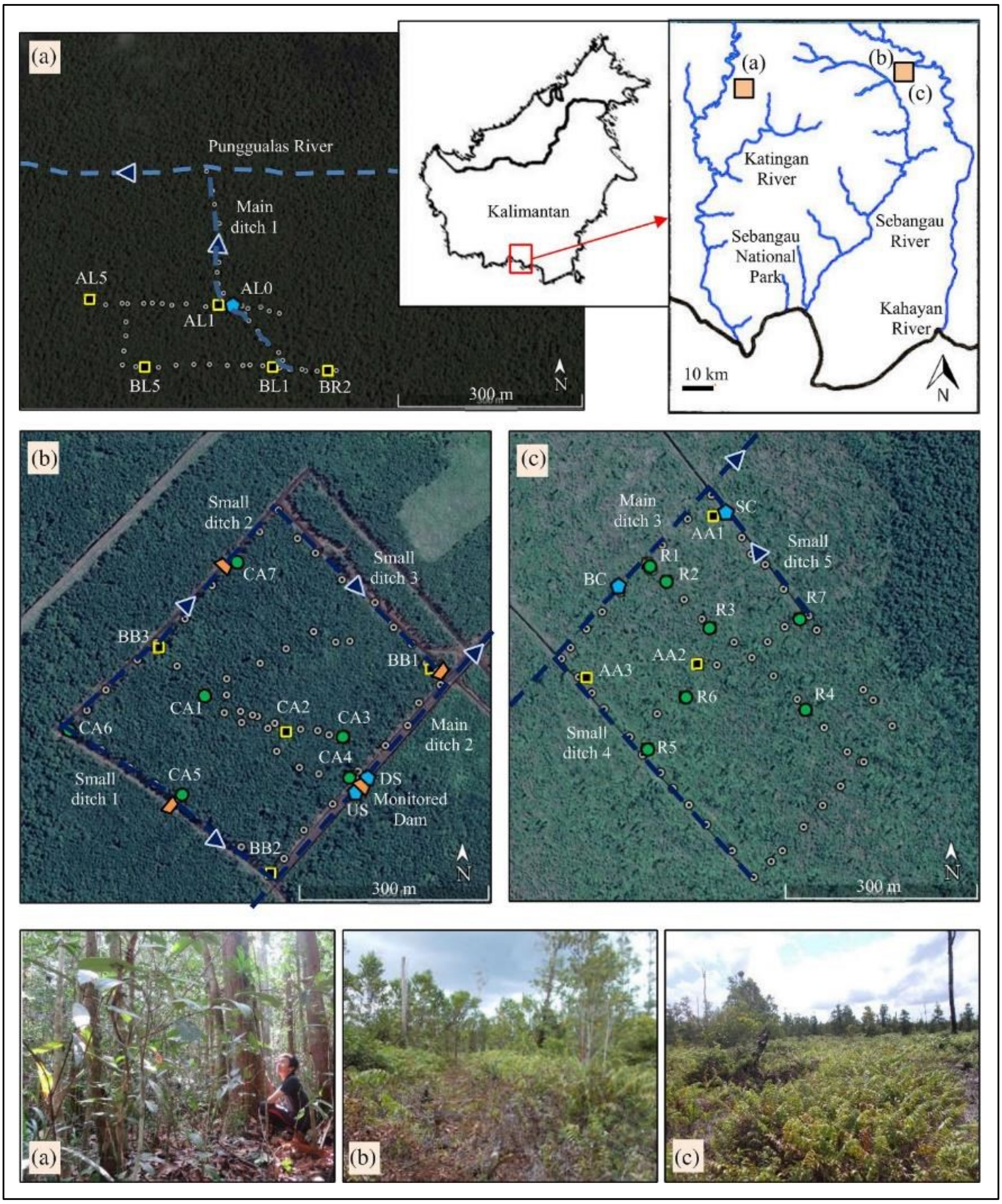

Figure 2.2 Studied sites and instrumentation in Sebangau, Indonesia, showing Forested (a), Blocked (b), and Drained (c). Squares symbolize logger wells, blue pentagons are ditch loggers, green circles are manual wells, smaller circles are levelling points, and trapezoids are dams. The main water flow directions are shown by arrows. The satellite images are from Google Earth, captured on 4 December 2015 (a), 26 July 2014 (b), and 14 August 2014 (c). Drained (b) and Blocked (c) were burned during the 2015 dry season. The bottom row of photographs shows land-cover across the sites.

The Forested site was chosen as an example of a less disturbed peatland and was ecologically similar to the low pole forest category described by Husson et al. (2018). The site was located in the Mawardi plot, Punggualas, Karuing, Katingan, Kalimantan, Indonesia (2.3893°S, 113.4524°E). The Forested site was 4.2 km to the east of Katingan River. There was logging in the area before 1997. The site was strictly protected after 2003. No forest fires have occurred at the site since at least 2003. The trench on the studied plot was formed in the 1990s from dragging trees towards the main river following logging (Figure 2.2). This trench was blocked with a dam in 2017 as a fire risk reduction measure. The Punggualas River next to the site is naturally quite shallow (commonly less than 150 cm deep) and some adjustable dams to aid navigation have been constructed within it by local people.

Drained and Blocked were located in the “RePeat” area, Tumbang Nusa Research Forest Zone, Pulang Pisau, Kalimantan, Indonesia. The International Tropical Peatlands Center (2018) and my communication with the RePeat area manager, confirmed that the Ministry of Environment and Forestry Republic of Indonesia allocated the RePeat area in 2016 as an area for public tree planting and for boosting awareness of peat forest restoration among young people and communities. These sites were in the Sebangau–Kahayan catchment, 70 km from Forested, yet still in the Sebangau Peat Dome area. Drained and Blocked were located 7.1 and 5.7 km, respectively, to the west of Kahayan River. Drained (2.3527°S, 114.0580°E) and Blocked (2.3402°S, 114.0720°E) were within 2.1 km of each other. Both were part of the Mega Rice Project Area before 1997, when many canals and ditches were installed to prepare the land for agriculture. The zone did not fall under the SNP protection that started in 2004 (when SNP was formally established). However, it became a conservation area under the Tumbang Nusa Research Forest protection in 2003, and has been the subject of biodiversity research (Morrogh-Bernard et al., 2003; Limin et al., 2007; Lampela et al., 2017). The vegetation at Drained and Blocked in 2003 was described as mixed swamp forest by Blackham et al. (2014) and Cattau et al. (2015). Despite protection, the area has had repeated fires, with the largest fire occurring in 2015. The government constructed four ditch dams between 2016 and 2018 at Blocked (Figure 2.2). Revegetation actions were also implemented in the areas from 2016 onwards.

At Forested, the peatland's low pole forest was dominated by *Campnosperma sp*. and *Shorea sp*. (Page et al., 1999; Husson et al., 2018). Limin et al. (2007) noted that Sebangau was relatively pristine tropical rainforest peatland before 1970 but the area was developed since the transmigration period (organized people migration from Java to other islands in Indonesia, 1970–2000), causing deforestation and drainage. The peak of the development was during the Mega Rice Project implementation around 1997 (Dohong, Aziz, et al., 2017). Many scattered forest fires occurred, causing vegetation change (Hoscilo et al., 2013). At Drained and Blocked, the vegetation canopy was dominated by *Shorea balangeran*, *Dyera costulata*, and *Combretocarpus rotundatus*, which were partly replanted after fires (Blackham et al., 2014; Cattau et al., 2015; Husson et al., 2018). In early 2020, there were more young trees at Drained than at Blocked. In drier zones that have less water during the dry season, ferns (*Polypodiopsida*) dominate the land cover. In wetter zones, the sedge *Lepironia articulata* is well established.

2.2.2 Data collection

Hydrological monitoring was conducted at the three sites between 22 August 2019 and 17 January 2020. Automatic weather stations (AWS) and water-level loggers were installed in order to capture both late dry season and wet season conditions. A Davis Vantage Pro2 AWS was installed behind Tumbang Nusa Camp (2.3556°S, 114.0896°E), which is 2.7 km from Drained and 3.5 km from Blocked. Another AWS (Qingdao Tlead AW003) was installed in 2017 by the World Wildlife Fund for Nature (WWF) Indonesia, located beside Punggualas Camp (2.3865°S, 113.4453°E), 0.8 km from Forested. Both AWSs recorded rainfall, temperature, wind speed and direction, solar radiation, and relative humidity, allowing the calculation of potential evapotranspiration (PET based on the Penman–Monteith equation [Davis Instruments, 2006; Jensen and Allen, 2016]). The AWS monitoring frequency was 30 min at Tumbang Nusa Camp and 5 min at Punggualas Camp. The AWSs were calibrated, and data were downloaded every month. We obtained a complete series of data from each AWS, except for 2 days at the Punggualas AWS (non-consecutive day gaps, 10 and 13 November 2019) and a day at the Tumbang Nusa AWS (whole day gap, 1 November 2019) when no data were collected. Both AWSs were calibrated to ensure accurate readings.

At Drained and Blocked, wells were installed to capture the spatial variability of water levels with reference to the location of the ditches (Figure 2.2). Both Drained and Blocked comprised rectangular plots. Automatic wells were located at the centre of the plots and at some corners, with manual wells spread across the plots. Drained had three water-table loggers, two ditch level loggers, and seven manual wells. Blocked had four water-level loggers, two ditch level loggers, and seven manual wells (Figure 2.2). The wells at Forested were arranged in two transects (Figure 2.2), to cover a similar area to those studied at Drained and Blocked. All wells were created using a Russian corer and lined with PVC pipe. The PVC pipe had an outer diameter of 6.4 cm, an inner diameter of 5.7 cm, and was perforated at intervals of 20 cm along the pipe. There were four holes distributed evenly for each perforation interval. The hole diameter was 1 cm. Each well was 2 m deep.

During the monitoring period, 42 readings were collected in total from the manual wells at Drained and 44 from the manual wells at Blocked. At Drained and Blocked, water levels in the automatic wells were measured using In-situ Level TROLL 500 vented loggers, recording at a three-hour interval. Ditch water levels were monitored at 30-min intervals using Schlumberger Diver non-vented pressure loggers, positioned within stilling wells. There were no manual monitoring wells at Forested due to access restrictions for routine data collection; therefore, six TROLL 500 vented loggers were used to monitor water tables, recording at a 3-h interval. Three of these loggers operated only between November 2019 and January 2020 (Figure 2.2). There were 12 manual water-table readings collected from the automatic wells at Forested as part of calibration checks.

At Blocked, ditch water level was recorded at points upstream and downstream of Main ditch 2 dam only, though there were three other dams on the site (Figure 2.2). At Drained, the ditch level loggers were installed with one in the larger ditch (Main ditch 3) and one in the smaller ditch (Small ditch 5), which was closed at one end (Figure 2.2). There was also a barometric logger installed, to compensate for the atmospheric pressure recorded by the Diver logger. Prior to November 2019, water levels of zero at the ditch bed were recorded as all ditches were dry.

In order to determine absolute water-level profiles across the study sites, the wells were surveyed using an automatic Leica NA720 level. The levelling point intervals were between 12 and 64 m. At the main entrance of each site, we hammered a wooden post into and through the peat. The surface on which this post was anchored served as the local benchmark (BM). These surface BMs were beside Well AL0 at Forested (2.3894°S, 113.4524°E), beside Well BB3 at Blocked (2.3389°S, 114.0703°E), and beside the larger canal monitoring point at Drained (2.3513°S, 114.0569°E).

2.2.3 Data analyses

Logger data outliers were detected by comparing each data point to temporally adjacent data points and were removed when a value was more than 10 cm different to that 30 min either side. Filtered data then were aggregated to three-hourly and daily intervals for comparison purposes. The absolute water-level data were also used to define characteristic hydrological periods in the study, including a dry period (water level continuously decreasing), an oscillatory period (water level increases after rainfall but returns to its initial low position each time), a transition period (the time from the last day of low oscillatory water level to the first wet season peak water level), a wet period (high water level fluctuating at the end of the transition period), and a ponding period (water-level recession barely detectable over more than 15 days).

We assumed that the AWSs provided data that were representative of the sites they were located near to, and that meteorological conditions did not vary within sites. Therefore, differences in the descriptive statistics for the two AWSs are assumed to represent real differences between Forested on the one hand and Drained and Blocked on the other. We also faced constraints with the hydrological monitoring and were not able to install multiple wells with loggers at all sites (this was only possible at Forested as noted above). Therefore, we did not have datasets that could be compared using inferential statistics. Nevertheless, our manual well data from Drained and Blocked gave confidence that data from the logged wells at these sites were representative of each site as a whole. In other words, we are confident that differences in our descriptive statistics for water tables and water levels reflect real differences in site conditions.

2.3 Results

2.3.1 Meteorological summary

Forested and Drained/Blocked had different meteorological conditions. Between 22 August 2019 and 17 January 2020, total rainfall at Forested was 614 mm, with 62 days on which rainfall occurred and an estimated total potential evapotranspiration (PET) of 376 mm. At Drained and at Blocked, between 11 September 2019 and 13 January 2020, total rainfall was 937 mm with 65 rain days, and 665 mm of PET. The mean temperature from 30 min data at Forested and at Drained/Blocked was the same, which is 26.7°C (standard deviations [SD] using the 30-min data: ±3.1°C Forested; ±4.0°C Drained/Blocked). The diurnal temperature interquartile range was 23.1–30.7°C in the dry season, and 23.9–29.8°C in the wet season for Drained/Blocked. At Forested, the diurnal interquartile range was narrower, which was 24.2–29.3°C in the dry season, and 24.8–28.3°C in the wet season. In 2019, the wet season started in mid-November (Figure 2.3).

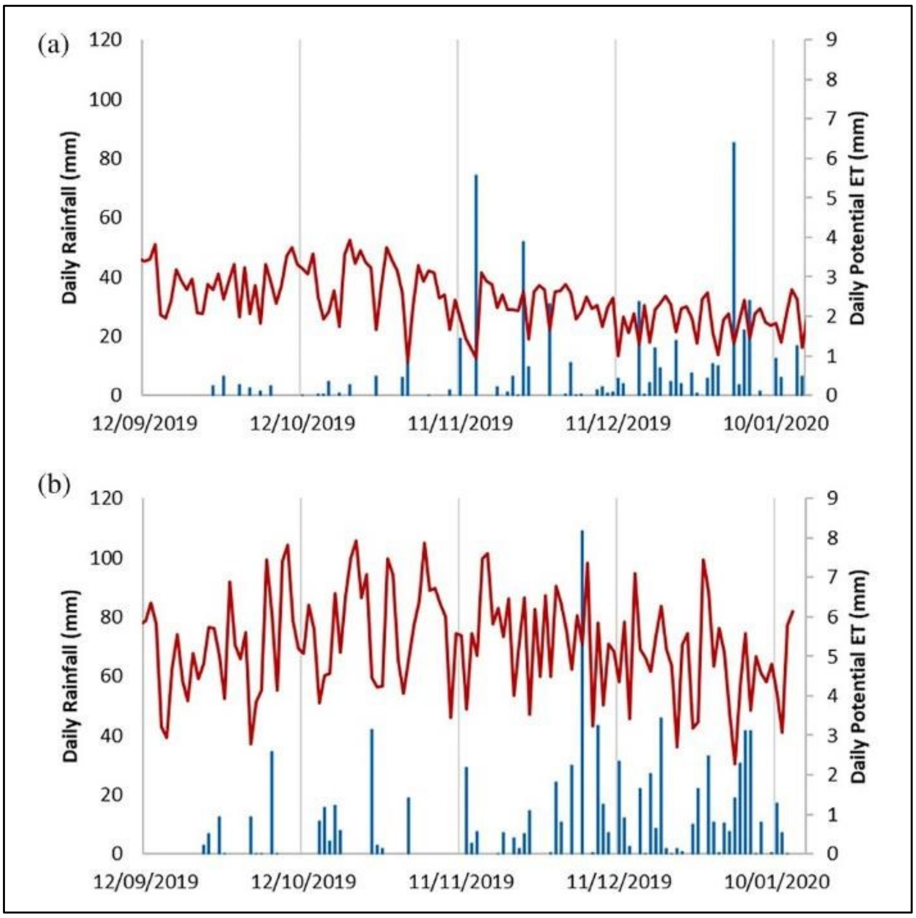

Figure 2.3 Daily rainfall and potential evapotranspiration time series in Sebangau Tropical Peatland: (a) Forested (b) Drained/Blocked. Red lines represent the estimated potential evapotranspiration (PET) and bars represent rainfall.

Although the wet season started simultaneously at both Forested and Drained/Blocked, the first three consecutive days with rain came later at Forested, which were from 21 to 23 November 2019 (Figure 2.3a) compared to 12–14 November 2019 for Drained/Blocked (Figure 2.3b). However, the second period with high intensity rain occurred concurrently at both sites, in early December. Our data showed that daily PET decreased from the dry season to the wet season. The days with the highest PET were in early September at Forested, but in October at Drained/Blocked. At Forested (Figure 2.3a), the mean dry season PET was 2.9 (SD = 0.8) mm day^{-1}, whereas the mean wet season PET was 2.1 (SD = 0.5) mm day^{-1}. At Drained/Blocked (Figure 2.3b), mean PET was 5.5 (SD = 1.3) mm day^{-1} and 5.1 (SD = 1.2) mm day^{-1} for the dry and wet seasons respectively. These large differences in PET between AWSs were mainly caused by differences in wind speeds which were much lower at Forested.

2.3.2 Seasonal water-level dynamics

Figure 2.4 presents time series for relative water levels for the wells and ditches using the local BM at each site. A summary of descriptive statistics for seasonal water levels is presented in Table 2.1. In the dry period, water-level decline occurred at a slower rate at Forested (9.3 mm day^{-1}), compared to Blocked (9.6 mm day^{-1}) and Drained (12.7 mm day^{-1}). The data showed that water-level responses to rainfall were faster at Forested, compared to those at Drained and at Blocked. Forested water levels started to rise in the late dry period (27 September 2019), while water levels at the other two sites were still oscillating from a low level. The first wet period peak water level at Forested occurred on 15 November 2019 resulting in a transition time of 50 days, while the transition period was 12 days at Drained and 5 days at Blocked. The interquartile range of water level in the wet period was 11.3 cm at Forested, 18.1 cm at Blocked, and 18.8 cm at Drained (see Table 2.1). In the wet period, sub-surface water-level fluctuations at all sites were mostly triggered by rainfall.

There were greater differences in absolute water levels between individual wells for Drained than for Blocked. The water-level fluctuation at Forested was much larger than for the other sites, particularly during the transition period and wet period (as presented in Figure 2.4). In the dry period, there were fewer differences in absolute water level between individual wells for all sites. Our data show that, in the oscillatory period, water levels at Blocked rose and fell seven times (by 8 to 27 cm) in response to rainfall events. After any rises, water levels declined gradually back to their initial level (oscillatory period), which was around 85 cm below the local benchmark. At Drained, water-level oscillations occurred only four times, between 5 and 15 cm above the common lowest water level (122 cm below the local BM). At Forested, no distinct oscillatory period occurred as the water levels rose, responding to a series of small rainfall events during the transition period.

Table 2.1 Summary water-level statistics across seasons and spatial locations in the study sites. Water-table data are in cm, with negative values indicating levels below the peat surface. Water-level difference data are in cm, and negative values indicate that ditch water level is higher than peat water level. The Q1, Q2, and Q3 represent the quartiles of the data. Residence time data are in percentage of the monitoring time. The period 'all' indicates the whole monitoring time of the ditch water level (see the Methodology section).

No.	Findings	Mean	Min	Q1	Q2	Q3	Max	Value	Period
1	Seasonal water-table dynamics (cm)								
	• Wells at Forested	-40.2	-55.1	-46.1	-40.6	-34.8	-23	-	Wet
	• Wells at Forested	-13.6	-34.6	-17.8	-13.3	-8.7	-4.7	-	Ponding
	• Wells at Blocked	-1.2	-21.6	-10.2	-3.2	7.9	23.2	-	Wet
	• Wells at Drained	-32.2	-52.1	-41.3	-35.6	-22.5	-4.7	-	Wet
	• Upstream dam	96.7	36.5	49.1	123.1	130.7	156.2	-	All
	• Downstream dam	78.7	-6.5	47.8	97.3	113.3	141.3	-	All
	• Main ditch 3	38.6	-37.4	5.5	52	67.1	108.3	-	All
	• Small ditch 5	39	-39.2	12.4	45.6	61.2	101.7	-	All
2	The differences in absolute water-level of wells to the lowest ditch water-level at each site (cm)								
	• Wells at Blocked	14.9	-3.8	10.9	14.4	18.3	36.4	-	Wet
	• Wells at Drained	22.1	-2.6	11.9	19.3	31	55	-	Wet
	• Wells at Forested	-3.1	-18.1	-7.9	-1.9	1.9	6.4	-	Wet
	• Upstream dam	19.3	9.7	14.6	18.9	23.8	38.1	-	All
	• Small ditch 5	15.6	-0.5	9.4	16	22.6	48.1	-	All
3	Cumulative residence time (%) during which water-tables were below the 40 cm limit								
	• Wells at Forested	-	16	-	-	-	87	-	Wet
	• Wells at Forested	-	0	-	-	-	0	-	Ponding
	• Wells at Blocked	-	0	-	-	-	0	-	Wet
	• Wells at Drained	-	0	-	-	-	38	-	Wet
	• Upstream dam	-	-	-	-	-	-	42	All
	• Downstream dam	-	-	-	-	-	-	46	All
	• Main ditch 3	-	-	-	-	-	-	97	All
	• Small ditch 5	-	-	-	-	-	-	96	All

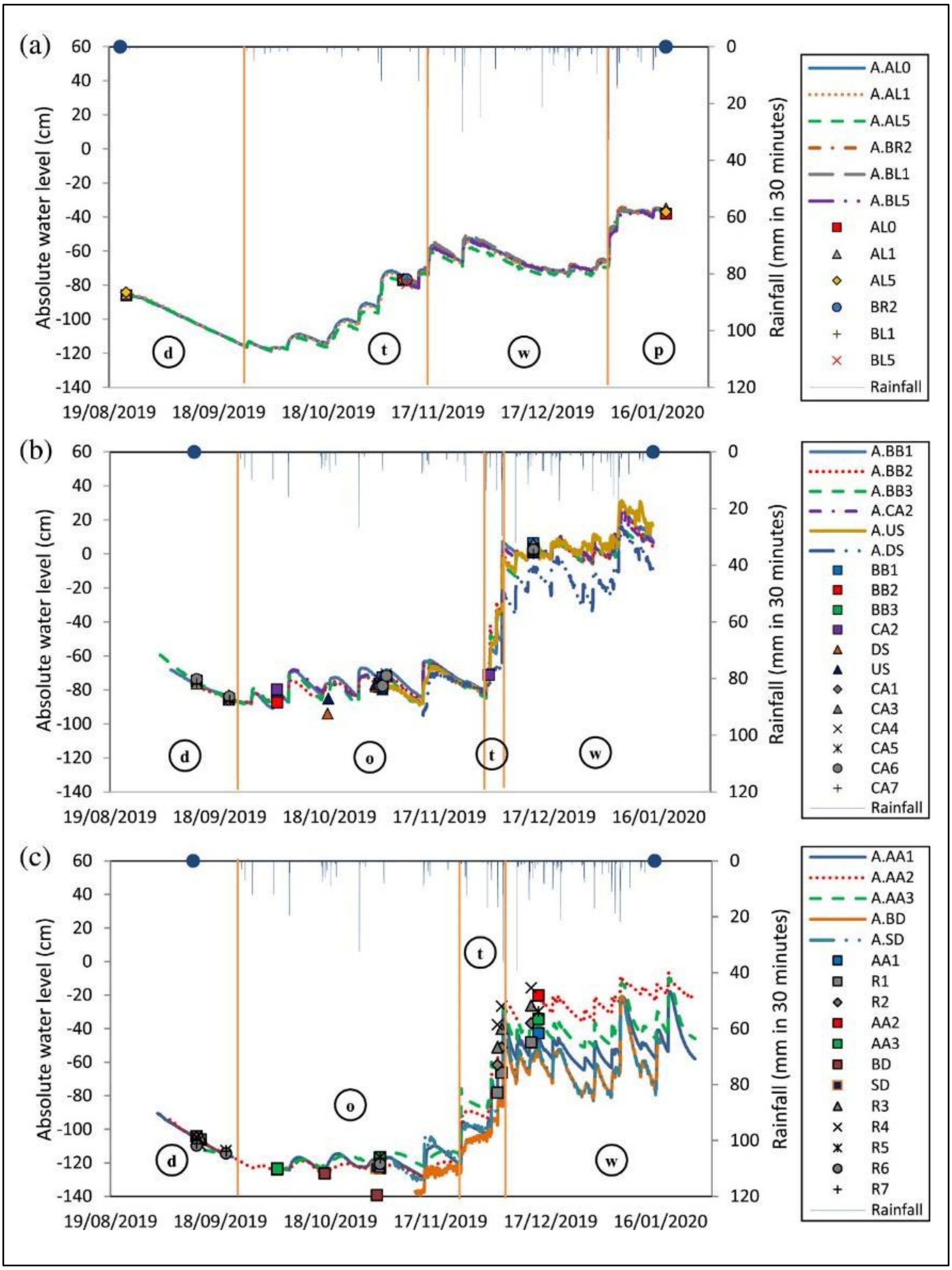

Figure 2.4 Water-level time series for late dry to early wet seasons at Forested (a), at Blocked (b), and at Drained (c) sites. Lines represent automatic logger data. Symbols indicate manual measurements. Bars represent rainfall data. Abbreviation 'A.' in legend indicates automatic logger, 'US' is upstream of the dam, 'DS' is downstream of the dam, 'BD' is Main ditch 3, and 'SD' is Small ditch 5. Different periods are dry (d), oscillatory (o), transition (t), wet (w), and ponding (p).

2.3.3 Spatial water-level variations

Figure 2.5 presents the difference in water level between wells and the lowest ditch level (the well—ditch difference) at Blocked and at Drained (Forested data are not shown because differences were less than 10 cm). The left-side graph in Figure 2.6 presents water-level differences between the upstream and downstream sides of the dam at Blocked. The right-side graph in Figure 2.6 displays the Small ditch 5 and Main ditch 3 water-level differences at Drained. All calculated water-level data from each site are relative to the local site BM (absolute water levels).

Differences in absolute water level for each individual well to the lowest ditch water level at all sites (the well—ditch difference) varied between time periods. The differences were small (<10 cm) in the dry period at all sites. The differences became larger during the transition period, except at Forested. In the wet period, the well—ditch differences at each site were between 11.9 and 31.0 cm at Drained, which were generally larger than those at Blocked (10.9 to 18.3 cm) and at Forested (−7.9 to 1.9 cm). Negative difference values indicate that ditch water level was higher than the peat water level. This condition did not necessarily cause water to spill across the surface of a site because the ditch water level was still lower than the bank of the ditch.

At Drained, the absolute water level in the wells near the ditches was lower than that in more distant wells, especially during the wet season (see Figure 2.4). Specifically, the spatial water-level differences were more affected by the monitoring well's average distance to the nearby side ditches. Figure 2.5b shows that the water level in the well closest to the ditches during the wet period was similar to the ditch water level. In contrast, the water level in the furthest well from the ditches (i.e., well AA2) was higher than the ditch water level. Interestingly, during the transition time, the water level in well AA2 was lower than ditch water levels, showing that ditch water levels rose more quickly than peat water levels at the centre of Drained.

At Blocked, there were similar patterns to Drained, with water levels in the wells close to the main ditch outlet showing more variation in comparison to water levels in the wells that were more distant (see Figure 2.4). The spatial differences were greater during the oscillatory period, compared to those in the other periods (Figure 2.5a). During the transition period, the furthest well (BB3) from the main ditch outlet had a higher water level compared to the well nearest to the ditch outlet. However, wells BB2 and CA2 had higher water levels than BB3 during this same period. BB2 was located near the upstream part of a dammed ditch segment, while CA2 was at the centre of the peatland block, furthest from any ditches.

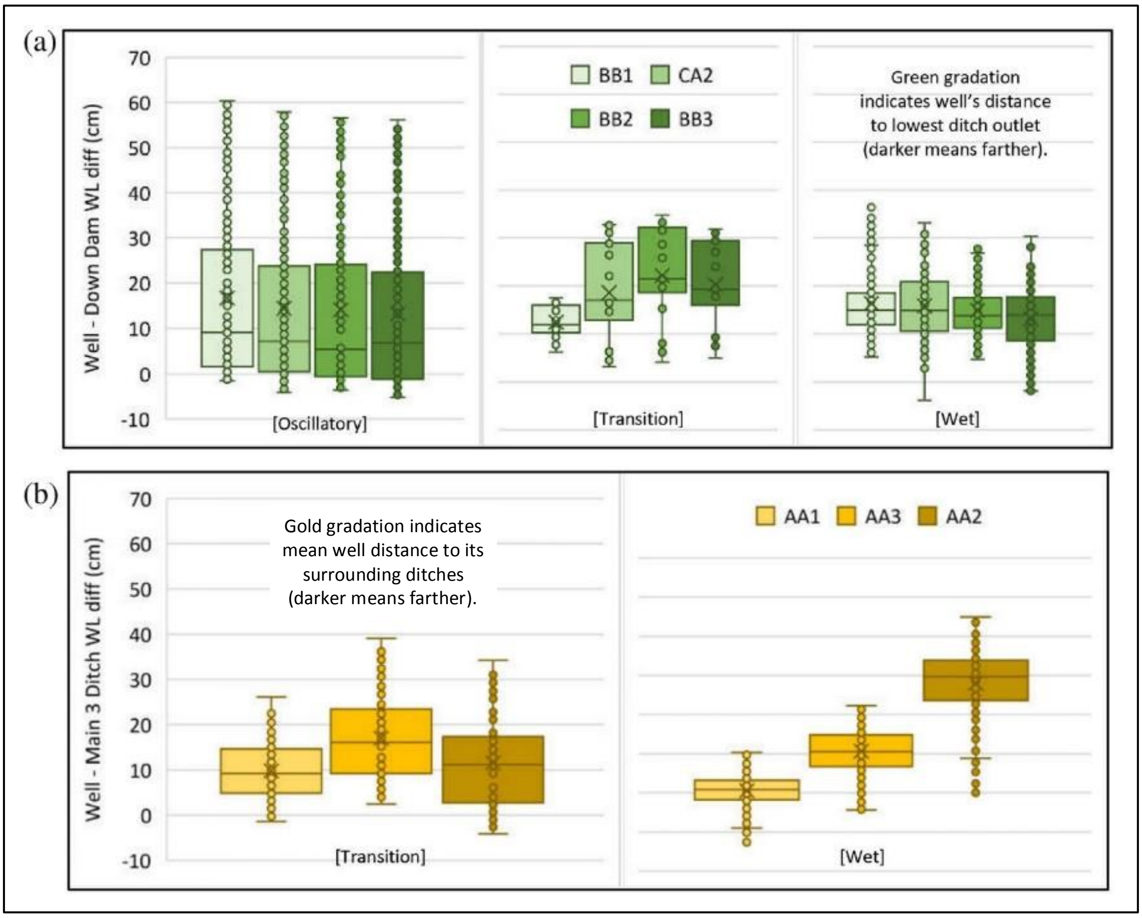

Figure 2.5 Differences between water level in the wells and water level in the lowest ditch on the site. The boxes represent monitoring wells. The lower, middle, and upper lines of the box indicate first quartile (Q1), median (Q2), and third quartile (Q3) of the data. The top and bottom ends of the whisker show $Q3 + 0.5\times|Q1–Q3|$ and $Q1 - 0.5\times|Q1–Q3|$ values. The cross symbol indicates the mean of the data, circles represent the data, but circles outside the whisker are the outliers. Blocked site data are from the oscillatory, transition, and wet periods, from left to right (a). Green gradation indicates a well's distance to lowest ditch outlet, which are 21 m (BB1), 275 m (CA2), 424 m (BB2), and 465 m (BB3). Drained data are from the transition and wet periods, from left to right (b). Gold gradation indicates mean well distance to its surrounding ditches, which are 21 m (AA1), 29 m (AA3), and 173 m (AA2). Negative values indicate that ditch water level is higher than peat water level. Water-level difference is abbreviated as WL diff.

The interquartile range of the hydraulic gradient between the water level at the lowest ditch outlet and in the farthest well from ditches was 3.7×10^{-4} to 7.8×10^{-4} m m^{-1} at Blocked, but 1.9×10^{-3} to 2.6×10^{-3} m m^{-1} at Drained. At Blocked, the median water-level difference upstream and downstream of the dam was 4 cm during the oscillatory period (Figure 2.6). Later in the study, during the transition and wet periods, ditch water levels at Blocked fluctuated substantially. At Blocked, the interquartile range of water-level differences between upstream and downstream of the dam was 8–32 cm in the transition period, but it was 14–24 cm in the wet period (Figure 2.6). At Drained, during the transition period, Small ditch 5 had a faster response to rainfall than Main ditch 3, with flow occurring from the former into the latter after each rainfall event. At Drained, the ditch water levels were generally lower than the peat water levels in the transition period (Figure 2.6); hence, water flowed from the peat into the ditches. At Drained, during the wet period, water levels on Main ditch 3 and on Small ditch 5 were approximately the same.

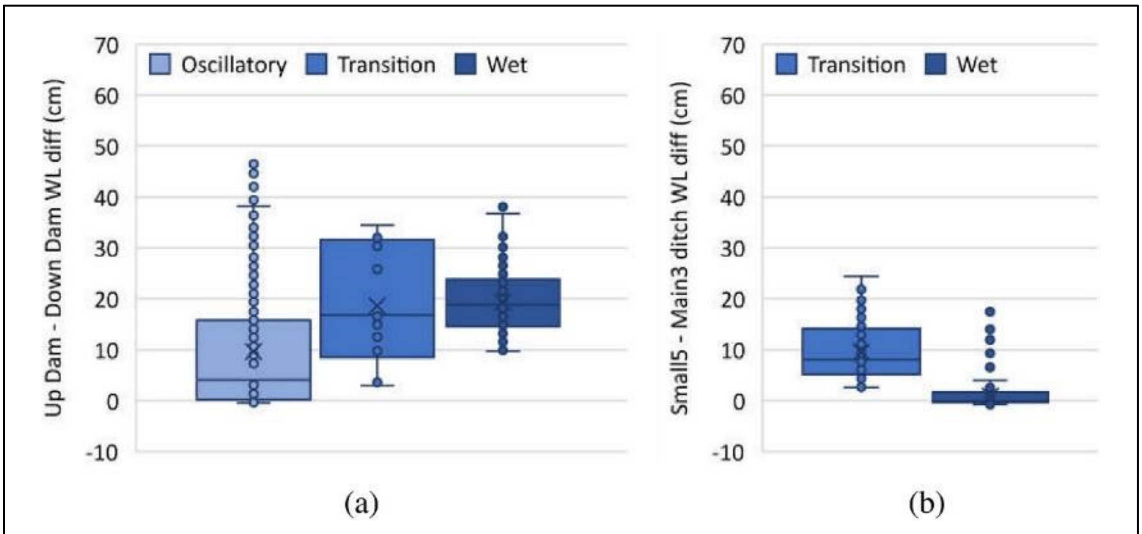

Figure 2.6 Differences in water levels in ditch sections for different time periods. The boxes represent seasonal periods. The lower, middle, and upper lines of the box indicate first quartile (Q1), median (Q2), and third quartile (Q3) of the data. The top and bottom ends of the whisker show Q3 + 0.5×|Q1–Q3| and Q1 − 0.5×|Q1–Q3| values. The cross symbol indicates the mean of the data, circles represent the data, but circles outside the whisker are the outliers. a) Blocked, in which negative values indicate that downstream water level is higher than upstream water level. b) Drained, in which negative values indicate that Main ditch 3 water level is higher than Small ditch 5 water level. Note that ditches were dry before mid-November 2019. Water level difference is abbreviated as WL diff.

2.3.4 Water-table residence times

Figure 2.7 presents water-table depth (i.e., relative to the peatland surface) residence times while Figure 2.8 shows the ditch inundation height residence times for the three study sites. During the dry period, the water table at all sites was deeper than the 40 cm limit specified in Indonesia's national regulations (Figure 2.7). At Forested, the dry season maximum water-table depth was 119 cm, about the same as at Drained (121 cm), while it was 98 cm at Blocked. A quantitative summary of water-table residence times is presented in Table 2.1.

The oscillatory period was undetectable in the forested peatland. In response to rainfall, Forested water tables did not return to their initial level but shifted to a shallower level. After the second large rainfall event, around 2 January 2020, Forested entered the ponding period. There was no significant water-table drop at Forested in the ponding period. Although the water-table depths still ranged from 9 to 18 cm in depth from the surface, in the six monitoring wells (Figure 2.7a), some inundations shallower than 10 cm depth were noted on small depressions in the microtopography at Forested. At Forested, during the ponding period, the ditch water table was above the ditch bed, whereas it was below the ditch bed at all other periods.

In the wet period, the mean water-table depth in the wells near the ditches at Drained (AA1 and AA3) was around 37.2 (SD = 9.5) cm, but 20.5 (SD = 6.7) cm in the well that was farthest from the ditches (AA2). Surface inundation was not indicated by any of the logged wells at Drained during the wet period. However, visually we observed some inundated areas, which were farther from ditches, and had shallow ponding (1–5 cm) during the wet period. In contrast, a large portion of Blocked was ponded in the wet period, especially near ditches, and the inundation height reached 23 cm (Figure 2.7b).

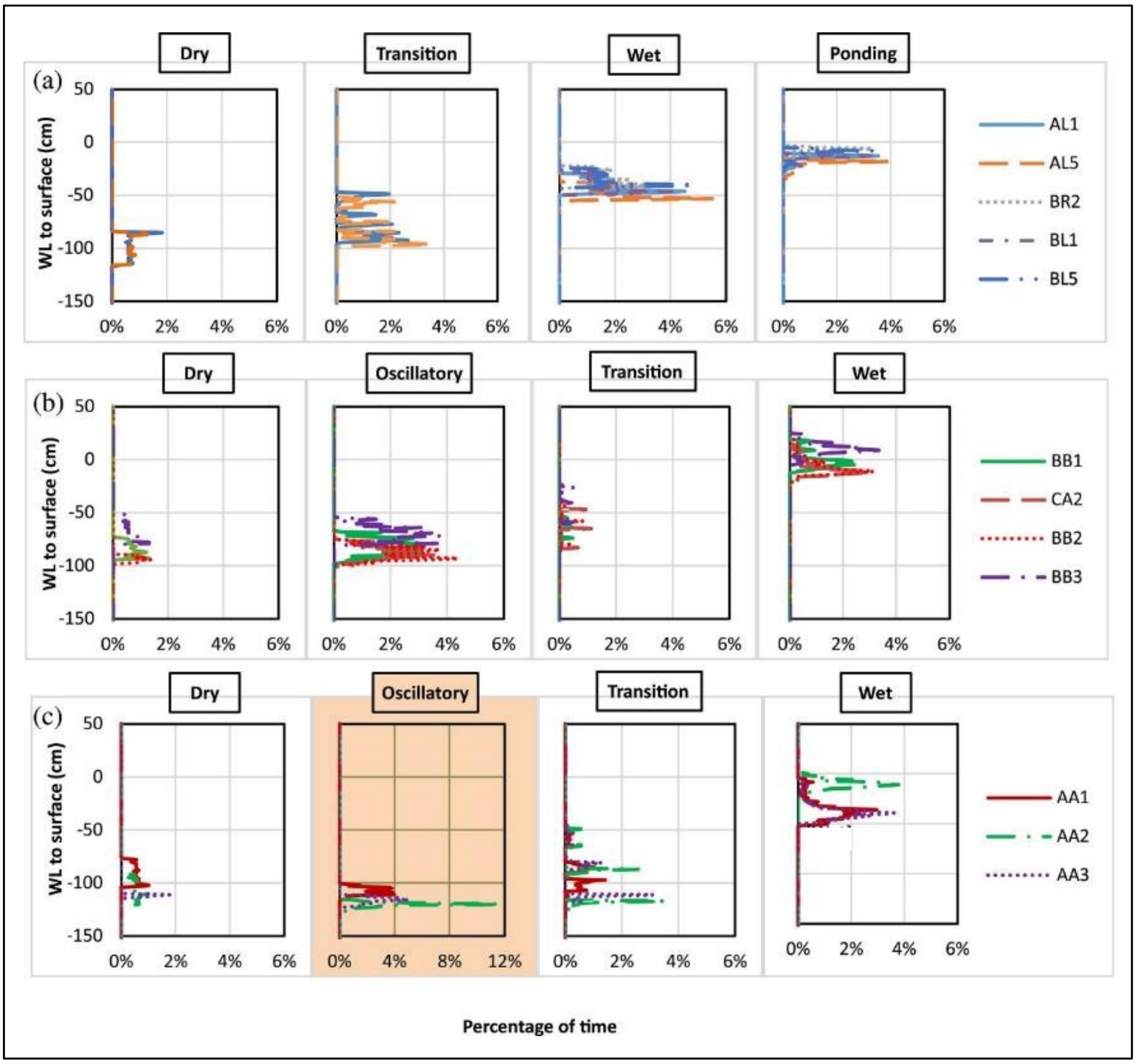

Figure 2.7 Water-level residence time for the whole study period in the study sites. Forested data (a) are presented for the dry, transition, wet, and ponding periods, from left to right. The data from Blocked (b) and Drained (c) are for the dry, oscillatory, transition, and wet periods, left to right. The orange shaded plot has a different *x*-axis scale. The peat surface level is referenced as level zero. Negative values indicate that water levels were below the peatland surface. Water level is abbreviated as WL.

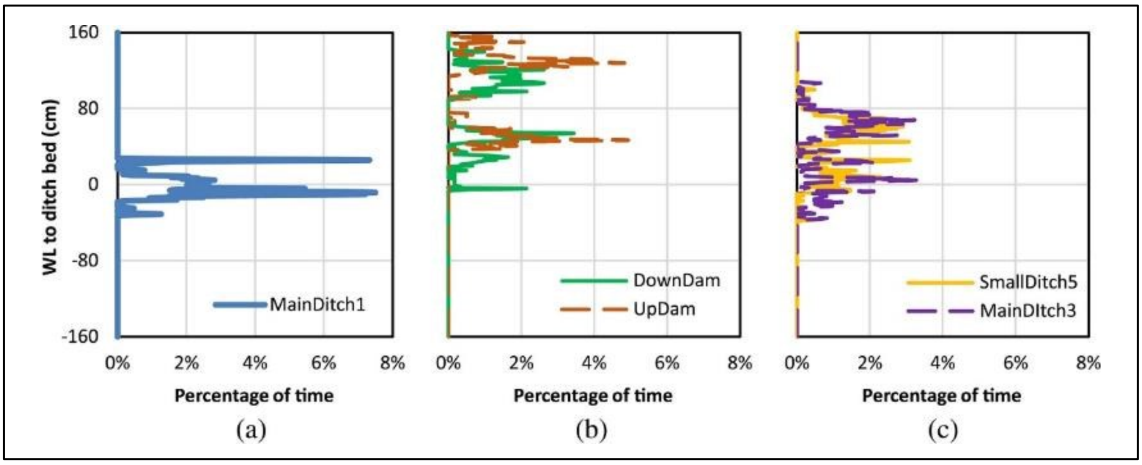

Figure 2.8 Ditch water-level residence time for the early wet seasons in the three study sites: (a) is for Forested, (b) is for Blocked, and (c) is for Drained. The data are from 30 October 2019 to 15 January 2020. Ditch AL0 bed depth at Forested is 64 cm; ditch bed depths at Blocked are both 125 cm; ditch bed depths at Drained are 107 cm in the larger ditch and 90 cm in the smaller ditch. Water levels are given relative to the ditch bed. The ditch bed level is referenced as level zero. Negative values indicate that water levels were below the ditch bed. The ditch bed depth is the distance between the top of the ditch bank and the ditch base. Water level is abbreviated as WL.

The ditch water height at Forested fluctuated less than at the other sites (Figure 2.8a), with a mean value of −0.4 cm (SD = 13.8 cm and interquartile range from −9.9 to 6.4 cm). The mean ditch water height (78.7 cm, SD = 40.4 cm, and interquartile range from 47.7 to 113.3 cm) at downstream of the dam, at Main ditch 2 in Blocked was higher than at Main Ditch 3 in Drained (38.6 cm, SD = 35.4 cm, and interquartile range from 5.5 to 67.1 cm). The residence time curves for Forested and Blocked ditch water levels appeared to form a bimodal distribution, whereas the residence time curve for Drained was more spread. At Blocked, the ditch water height upstream of the dam was generally higher than downstream of the dam, and both locations had different residence time patterns (Figure 2.8b).

2.4 Discussion

2.4.1 Water tables in different peatland settings

Water-table dynamics were different among the three studied sites. The mean water-table depths during the whole monitoring period at Drained (78.2 cm, SD = 38.9 cm), at Blocked (54.8 cm, SD = 38.8 cm), and at Forested (48.8 cm, SD = 24.6 cm) were generally in line with values reported from other drained or intact tropical peatlands (Table 2.2). However, our study was able to provide additional detail on temporal variability and spatial patterns relative to ditches.

Table 2.2 Water-table values reported from tropical peatlands. Water-table data are in cm, with negative values indicating levels below the peat surface. The 'Num' column gives the number of automatic wells (A) or the number of manual measurement wells (M). The Q1, Q2, and Q3 represent the quartiles of the data.

No.	Location	Num	Mean	Min	Q1	Q2	Q3	Max	Period
1	Drained, Kalampangan, Indonesia, 50 to 150 m from primary canal (Lampela et al., 2017)								November 2012 to July 2014
	• Wet peat (2°20′24″ S, 114°02′11″ E)	1A	-20	-60	-	-	-	0	
	• Medium wet peat (2°19′18″ S, 114°01′05″ E)	1A	-30	-80	-	-	-	0	
	• Dry peat (2°19′32″ S, 114°00′59″ E)	1A	-50	-100	-	-	-	-20	
2	Drained, Kalampangan, Indonesia (Ritzema et al., 2014) Transect 3 (2°21'22.23" S, 114° 3'8.30" E)	968 M	-14	-52	-	-	-	9	2006 to 2009
3	Drained, Kalampangan, Indonesia (Santoso and Qirom, 2020) Transect 3 (2°21'22.23" S, 114° 3'8.30" E)	75 M	-15	-55	-	-	-	5	March 2013 to September 2013
4	Kalampangan, Palangka Raya, Indonesia (Jauhiainen et al., 2014)								May 2012 to September 2012
	• Agricultural peatland (Mr. Edi's field)	1A	-50	-	-40	-52	-59	-	
	• Degraded peatland (2°19′24″ S, 114°1′14″ E)	1A	-52	-	-47	-51	-59	-	

(Continues)

Table 2.2 (Continued)

No.	Location	Num	Mean	Min	Q1	Q2	Q3	Max	Period
5	Forest, LAHG CIMTROP, floodplain (FP) to forest transect (Lampela et al., 2017) (02°18.843′ S, 113°54.159’ E)								November 2012 to July 2014
	• On FP	1A	-100	-120	-	-	-	20	
	• At 40 m from FP	48 M	-85	-100	-	-	-	20	
	• At 80 m from FP	48 M	-70	-80	-	-	-	20	
	• At 120 m from FP	48 M	-55	-60	-	-	-	20	
	• At 160 m from FP	48 M	-40	-50	-	-	-	10	
	• At 200 m from FP	48 M	-10	-30	-	-	-	10	
6	Forest, LAHG CIMTROP (Takahashi et al., 2002) (2°18'59.6" S, 113°54'28.9" E)	-	-40	-98	-	-	-	20	1993 to 2000
7	Block A, Ex-MRP, Mawas (Sinclair et al., 2020) (2°15′ S, 114°30′ E)								2012
	• Intact forest	55 M	-15	-70	-	-	-	5	
	• Far from canal	33 M	-30	-80	-	-	-	-5	
	• Near canal	55 M	-50	-85	-	-	-	-25	
8	Block A, Ex-MRP, Mawas, Large Canal Drainage (Hooijer et al., 2014) (2°15′ S, 114°30′ E)								March 2010 to December 2012
	• Burned peatland transect	220 M	-26	-63	-37	-	-	-	
	• Degraded forest transect	62 M	-43	-86	-53	-	-	-	
9	Forest transect, Block E, Ex-MRP, Mawas, Small Canal Drainage (Hooijer et al., 2014) (2°10′ S, 114°30′ E)	107 M	-34	-89	-49	-	-	-	March 2010 to December 2012
10	Palm Oil Plantation, Wajok Hilir Peatland, Siantan, Mempawah, Indonesia (Herawati et al., 2018). The study was on a tertiary and quaternary canal, with a depth of around 2 m and 1.5 m respectively. (0°07'04.3" N, 109°17'42.8" E)								February 2018 to March 2018
	• Nearby quaternary canal with dam	60 M	-	-73	-	-	-	-46	
	• Nearby quaternary canal without dam	60 M	-	-70	-	-	-	-15	
	• Nearby tertiary canal with dam	60 M	-	-80	-	-	-	-45	
	• Nearby tertiary canal without dam	60 M	-	-78	-	-	-	-23	

Hydraulic gradients at the studied sites appeared to be strongly affected by peatland management. Prior studies have indicated that drainage increases peatland hydraulic gradients, but ditch and canal blocking combined with other restoration measures might lessen gradients (Baird et al., 2017; Young et al., 2017; Dohong et al., 2018a; Urzainki et al., 2020). At Blocked, during the wet period, water tended to flow towards the lowest outlet from the peatland, which was near to well BB1 (Figure 2.2). At Drained, there was no dominant hydraulic gradient, as water tended to flow towards all available ditches. These findings show that if the water levels in the ditches can be maintained, then the peatland is likely to stay wetter for longer. At Forested, the water levels were similar to each other across the studied plot, although there was a shallow trench on the plot edge (~60 cm in depth). There did not appear to be obvious impacts from this trench on water-level variation among wells at Forested.

In the dry period, water levels decreased at different rates, and were affected by drainage to ditches, as well as evapotranspiration which varied strongly between Forested and Drained/Blocked. The water-level drawdown rate decreased as the hydraulic gradient lessened. Accordingly, the drainage effect was barely perceptible during the late dry season, as the ditch water levels were at least on or below the ditch bed. These findings suggest that lateral flow loss from the peat mass into ditches did not exist during this period. Instead, evapotranspiration solely lowered the water levels in the dry period, as found in other tropical peatland studies (Kumagai et al., 2005; Hirano et al., 2015; Lion et al., 2017).

In the wet period, variations in land management appeared to cause differences in inundation depth, duration, and distribution. Surface inundation was most pronounced at Blocked (up to 23 cm), especially around the ditches. The inundation at Blocked may be related to the dam position and size (see Ditch dam effects on water levels subsection), rainfall, and the wet season water-level condition at the lowest outlet of the peatland area (see also Ritzema et al. (2014), Kasih et al. (2016), and Urzainki et al. (2020)). Inundation may also be related to river water levels, especially at Forested which was quite close to the Punggualas river, because a higher river level will reduce hydraulic gradients for areas alongside the river (see also the study by Itakura et al. (2016)).

Near-surface water-tables (including inundation) are needed to buffer water levels during the dry season, as also suggested in other studies (Evers et al., 2017; Wijedasa et al., 2017). Our study showed that the wet season water tables at Drained remained relatively deep and were not suitable for new peat accumulation. In the wet season, the interquartile range in water tables in wells at Drained was between 22.5 and 41.3 cm below the surface. The water-table depth in wells near ditches receded mostly near to the 40 cm regulatory limit (median 35.6 cm), after around 5–7 days from a storm event (Figure 2.4c). This condition does not give confidence that there would be enough buffered water at Drained for the following dry season. In contrast, the interquartile range for the wet season water table at Blocked was between 7.9 cm above the surface and 10.2 cm below the surface. Nevertheless, long periods of inundation may be undesirable, because they may enhance methane release from peatlands (Teh et al., 2017; Wong et al., 2018).

The impact of greater rainfall at Drained/Blocked (937 mm between 11 September 2019 and 13 January 2020) than at Forested (586 mm across the same period) might affect the comparisons in water-table dynamics. This would suggest that even stronger differences in water-table behaviour (e.g., longer periods of deeper water table, or less frequent rise of water table at Drained/Blocked than at Forested) might have occurred if rainfall were equivalent between sites. However, when net rainfall (rainfall minus evapotranspiration) was calculated for Forested and Drained/Blocked it was approximately the same (272 mm between 11 September 2019 and 13 January 2020 at Drained/Blocked and 285 mm at Forested across the same period). Additionally, as Drained and Blocked were co-located, water-table comparisons between those treatments should be unaffected by rainfall variation. In Chapter III non-parametric analysis of the response of water tables to individual rainfall events is undertaken which should further control for differences in rainfall totals between sites.

2.4.2 Ditch dam effects on water levels

It was notable that Blocked had deeper water tables than the governmental target water table during the dry season. In the dry season, the dams did not raise the peatland water table because the ditches were dry, and no excess water was retained. At Blocked, peatland water tables and water levels downstream of the dams rose rapidly after rainfall events during the transition period and stayed high during the wet period (Figure

2.4b). Our data show that the effect of ditch dams depends on rainfall inputs and also ditch water levels downstream of the dams. The areas downstream of a dam where water levels are lowest, provide zones where water flow from the peat mass can concentrate, as hydraulic gradients are highest, confirming suggestions from previous tropical peatland modelling studies (Ochi et al., 2016; Urzainki et al., 2020).

The ditch dams raised ditch water height upstream of the dam, reducing peat plot—ditch water-level differences, which resulted in a reduction of the hydraulic gradient in the peatland, and therefore lower rates of flow between the peat mass and the ditch. By slowing water losses, water had a longer residence time in the peatland, so that water tables were closer to the peat surface. Peatland water tables at Blocked during the wet period ranged between 9 cm below to 11 cm above the surface. The main cause of the lower hydraulic gradient was the sustained higher water levels in the ditches behind the dams (see also the studies by Susilo et al. (2013) and Kasih et al. (2016)).

The effect of ditch dams in raising water levels was spatially confined and limited in coverage, and ditch dam effects were temporary. They were conditional on rainfall inputs and the location of the dams relative to the lowest outlet water level from the peatland, as also indicated in modelling studies by Ochi et al. (2016) and Urzainki et al. (2020). The ditch dams failed to have an effect during much of the dry season with little water buffering available against PET loss. As the water tables were deeper during the dry season at Drained than at Blocked, it is likely that the Drained site had less water than the Blocked site at the beginning of the dry season, before the water-level monitoring started, as also indicated by our wet season data. The ditch dams only partly buffered the Blocked site against water losses via evapotranspiration. Thus, additional measures are required on such drained tropical peatland systems to retain water from the late wet season into the early dry season. For example, bunds could be used to help store surface water across the site (Price et al., 2003; Wichmann et al., 2017; Payne et al., 2018).

2.4.3 Water-table management implications

The dry season water-table data at all sites did not meet the limit of the Indonesian peatland regulation (Republic of Indonesia Government Regulation No. 57 Year 2016 about Peatland Ecosystem Protection and Management, 2016), as the water tables were deeper than 40 cm below the surface. This limit was generally developed in relation to high fire risk in Indonesian tropical peatland (Taufik et al., 2015; Putra et al., 2018); therefore, it can be suggested that all of the studied sites were prone to fire in the dry season. More data are required to improve the water-table regulation policy and to ensure that the regulation reflects natural dry season and wet season water-table dynamics in tropical peatlands. This is particularly so, given that water tables at Forested, a near natural site, did not conform to the 40 cm limit. Moreover, Sinclair et al. (2020) reported water-table depths deeper than 40 cm in a relatively undisturbed tropical peatland site (Table 2.2). While human-induced actions in lowering peatland water level exacerbate fire risk (Evers et al., 2017; Wijedasa et al., 2017), further research is required to understand natural system water-table variability for forested tropical peatlands.

If the 40 cm water-level regulation is to be maintained, ditch dam construction on drained topical peatlands needs careful site design and layout (Dohong et al., 2018a; Urzainki et al., 2020). There is a need to test different spatial designs of dam installations to understand whether adequate dry season buffering can be provided. Some studies have suggested that bunding without peat excavation may enhance ditch dam performance in drained northern peatlands (Mackin et al., 2017; Payne et al., 2018). Such additional techniques may need to be further tested and refined for higher permeability tropical peat landscapes. It is essential to see how well those techniques may hold back water from the wet season and maintain water tables near the ground surface into the dry season.

We did not consider water-level dynamics across different dry and wet years, which may differ substantially in terms of rainfall pattern and rainfall depth compared to the monitored period. In particular, water-level dynamics in El Niño, La Niña, and ENSO-neutral years may be very different and so longer monitoring periods would be desirable. Further research is required to understand more detailed spatial interactions of water-level dynamics and dam locations or ditch layouts in tropical peat systems;

modelling may be beneficial in this regard. While our block drainage systems were representative of management in the study region, not all tropical peatlands will have the same peat properties and land-use settings as the three sites in this study (Ilek et al., 2017; Kurnianto et al., 2019; Sinclair et al., 2020). Therefore, establishing the interactions between peat properties, drainage and restoration designs and hydrological function remains a key priority. In degraded peatland, it will be very important to link water-table dynamics to peat properties (Sinclair et al., 2020; Hoyt et al., 2020), so that modelling of hydrological response to different restoration measures or climate change impacts can be undertaken with more confidence. Such work would enable improved advice on the spatial design of restoration interventions that provide buffering to dry season conditions for tropical peatlands.

2.5 Conclusions

It was found that ditch dams buffered a tropical peatland restoration site against water losses via evapotranspiration, compared to a nearby drained site. If water levels in the ditches can be maintained, then the drainage effect would be smaller. Nevertheless, water-level dynamics at the drained and the blocked sites still differed from those at a near-natural site. Ditch damming alone was not enough to ensure water tables remained close to the peatland surface in the dry season. The effect of ditch dams in raising water tables was spatially confined, limited in coverage, and also temporary. It was conditional on the rainfall input and the lowest water level in the drain network surrounding the peatland.

Additional measures may be required at some sites, in accordance with revised spatial designs of dams or bunds. These measures are needed to store water from the wet season for a longer period of time into the dry season. However, it should also be noted that, in the dry season, the water tables were deeper than the Indonesian regulatory 40 cm depth limit in the drained peatland with ditch dams and even in the near-natural forested system. In the wet season, water tables among the Forested site wells were deeper than the 40 cm depth limit for 45% of the time, suggesting that: i) the Indonesian regulatory limit may need refining based on further hydrological research from more natural sites to better represent natural processes that align with conservation needs, and ii) widespread assumptions about the nature of near-surface water tables in forested tropical peatlands may need to be supported by further multi-year monitoring.

2.6 Data availability statement

The data that are presented in this study are available via The University of Leeds repository (https://doi.org/10.5518/960), which are publicly accessible under Creative Commons BY-NC Licence.

2.7 References

Agus, C., Azmi, F.F., Widiyatno, Ilfana, Z.R., Wulandari, D., Rachmanadi, D., Harun, M.K. and Yuwati, T.W. 2019. The Impact of Forest Fire on the Biodiversity and the Soil Characteristics of Tropical Peatland *In*: W. L. Filho, J. Barbir and R. Preziosi, eds. *Handbook of Climate Change and Biodiversity. Climate Change Management*. Springer Cham, pp.287–303.

Baird, A.J., Low, R., Young, D., Swindles, G.T., Lopez, O.R. and Page, S.E. 2017. High permeability explains the vulnerability of the carbon store in drained tropical peatlands. *Geophysical Research Letters*. **44**(3), pp.1333–1339.

Ballhorn, U., Siegert, F., Mason, M. and 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*. **106**(50), pp.21213–21218.

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

Cattau, M.E., Harrison, M.E., Shinyo, I., Tungau, S., Uriarte, M. and DeFries, R. 2016. Sources of anthropogenic fire ignitions on the peat-swamp landscape in Kalimantan, Indonesia. *Global Environmental Change*. **39**, pp.205–219.

Cattau, M.E., Husson, S. and Cheyne, S.M. 2015. Population status of the Bornean orang-utan Pongo pygmaeus in a vanishing forest in Indonesia: the former Mega Rice Project. *Oryx*. **49**(3), pp.473–480.

Dargie, G.C., Lewis, S.L., Lawson, I.T., Mitchard, E.T.A., Page, S.E., Bocko, Y.E. and Ifo, S.A. 2017. Age, extent and carbon storage of the central Congo Basin peatland complex. *Nature*. **542**(7639), pp.86–90.

Davis Instruments 2006. Derived Variables in Davis Weather Products *In*: *Davis Vantage Pro2 Application Notes*. Hayward, CA: Davis Instruments, pp.1–23.

Dohong, A., Aziz, A.A. and Dargusch, P. 2018a. A review of techniques for effective tropical peatland restoration. *Wetlands*. **38**(2), pp.275–292.

Dohong, A., Aziz, A.A. and Dargusch, P. 2017. A review of the drivers of tropical peatland degradation in South-East Asia. *Land Use Policy*. **69**(1), pp.349–360.

Dohong, A., Aziz, A.A. and Dargusch, P. 2018b. Carbon emissions from oil palm development on deep peat soil in Central Kalimantan Indonesia. *Anthropocene*. **22**, pp.31–39.

Dohong, A., Cassiophea, L., Sutikno, S., Triadi, B.L., Wirada, F., Rengganis, P. and Sigalingging, L. 2017. *Modul pelatihan pembangunan infrastruktur pembasahan gambut sekat kanal berbasis masyarakat (Training Module for community based peatland rewetting infrastructure development)*. [Online]. Jakarta, Indonesia. Available from: https://brg.go.id/files/Publikasi/Modul_Sekat_Kanal_Berbasis_Masyarakat_Final_30082017.pdf.

Dommain, R., Couwenberg, J. and Joosten, H. 2011. Development and carbon sequestration of tropical peat domes in south-east Asia: links to post-glacial sea-level changes and Holocene climate variability. *Quaternary Science Reviews*. **30**(7–8), pp.999–1010.

Dommain, R., Couwenberg, J. and Joosten, H. 2010. Hydrological self-regulation of domed peatlands in south-east Asia and consequences for conservation and restoration. *Mires and Peat*. **6**, pp.1–17.

Evans, M. G., Burt, T. P., Holden, J. and Adamson, J. K. 1999. Runoff generation and water table fluctuations in blanket peat: evidence from UK data spanning the dry summer of 1995. *Journal of Hydrology*, **221**(3-4), pp.141-160.

Evans, C.D., Williamson, J.M., Kacaribu, F., Irawan, D., Suardiwerianto, Y., Hidayat, M.F., Laurén, A. and Page, S.E. 2019. Rates and spatial variability of peat subsidence in Acacia plantation and forest landscapes in Sumatra, Indonesia. *Geoderma*. **338**, pp.410–421.

Evers, S., Yule, C.M., Padfield, R., O'Reilly, P. and Varkkey, H. 2017. Keep wetlands wet: the myth of sustainable development of tropical peatlands - implications for policies and management. *Global Change Biology*. **23**(2), pp.534–549.

Giesen, W. and Sari, E.N.N. 2018. *Tropical Peatland Restoration Report: the Indonesian Case*. [Online]. Jambi, Indonesia. Available from: https://www.researchgate.net/publication/323676663_Tropical_Peatland_Restoration_Report_the_Indonesian_case.

Glauber, A.J., Moyer, S., Adriani, M. and Gunawan, I. 2016. *The Cost of Fire: An*

Economic Analysis of Indonesia's 2015 Fire Crisis. [Online]. [Accessed 1 November 2020]. Jakarta, Indonesia: World Bank, Jakarta. Available from: https://olc.worldbank.org/content/cost-fire-economic-analysis-indonesia's-2015-fire-crisis.

Goodbrand, A., Westbrook, C.J. and Kamp, G. 2019. Hydrological functions of a peatland in a Boreal Plains catchment. *Hydrological Processes*. **33**(4), pp.562–574.

Harris, L.I., Roulet, N.T. and Moore, T.R. 2020. Drainage reduces the resilience of a boreal peatland. *Environmental Research Communications*. **2**(6), article no.: 065001.

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. and Veen, F.J.F. 2020. Tropical forest and peatland conservation in Indonesia: Challenges and directions. *People and Nature*. **2**(1), pp.4–28.

Herawati, H., Ali Akbar, A., Farastika, D. and Azmeri 2018. Water table evaluation post the construction of canal blocks on peatland in West Kalimantan, Indonesia *In*: P. Hajek, A. L. Han, S. Kristiawan, W. T. Chan, M. bin Ismail, B. S. Gan, R. Sriravindrarajah and B. A. Hidayat, eds. *The 4th International Conference on Rehabilitation and Maintenance in Civil Engineering (ICRMCE 2018)*. Surakarta, Indonesia: MATEC Web of Conferences, article no.: 03016.

Hirano, T., Kusin, K., Limin, S. and Osaki, M. 2014. Carbon dioxide emissions through oxidative peat decomposition on a burnt tropical peatland. *Global Change Biology*. **20**(2), pp.555–565.

Hirano, T., Kusin, K., Limin, S. and Osaki, M. 2015. Evapotranspiration of tropical peat swamp forests. *Global Change Biology*. **21**(5), pp.1914–1927.

Holden, J., Green, S.M., Baird, A.J., Grayson, R.P., Dooling, G.P., Chapman, P.J., Evans, C.D., Peacock, M. and Swindles, G. 2017. The impact of ditch blocking on the hydrological functioning of blanket peatlands. *Hydrological Processes*. **31**(3), pp.525–539.

Holden, J., Wallage, Z.E., Lane, S.N. and McDonald, A.T. 2011. Water table dynamics in undisturbed, drained and restored blanket peat. *Journal of Hydrology*. **402**(1–2), pp.103–114.

Hooijer, A., Page, S.E., Jauhiainen, J., Lee, W.A., Lu, X.X., Idris, A. and Anshari, G. 2012. Subsidence and carbon loss in drained tropical peatlands. *Biogeosciences*. **9**(3), pp.1053–1071.

Hooijer, A., Page, S.E.E., Navratil, P., Vernimmen, R., van der Vat, M., Tansey, K., Konecny, K., Siegert, F., Ballhorn, U. and Mawdsley, N. 2014. *Carbon Emissions from Drained and Degraded Peatland in Indonesia and Emission Factors for Measurement, Reporting and Verification (MRV) of Peatland Greenhouse Gas Emissions*. [Online]. Jakarta, Indonesia. Available from: https://www.researchgate.net/publication/323616904.

Hoscilo, A., Page, S.E., Tansey, K.J. and Rieley, J.O. 2011. Effect of repeated fires on land-cover change on peatland in southern Central Kalimantan, Indonesia, from 1973 to 2005. *International Journal of Wildland Fire*. **20**(4), pp.578–588.

Hoscilo, A., Tansey, K.J. and Page, S.E. 2013. Post-fire vegetation response as a proxy to quantify the magnitude of burn severity in tropical peatland. *International Journal of Remote Sensing*. **34**(2), pp.412–433.

Hoyt, A.M., Chaussard, E., Seppalainen, S.S. and Harvey, C.F. 2020. Widespread subsidence and carbon emissions across Southeast Asian peatlands. *Nature Geoscience*. **13**(6), pp.435–440.

Husson, S.J., Limin, S.H., Boyd, N.S., Brousseau, J.J., Collier, S., Cheyne, S.M., D'Arcy, L.J., Dow, R.A. and Schreven, S. 2018. Biodiversity of the Sebangau tropical peat swamp forest, Indonesian Borneo. *Mires and Peat*. **22**, pp.1–50.

Ilek, A., Kucza, J. and Szostek, M. 2017. The effect of the bulk density and the decomposition index of organic matter on the water storage capacity of the surface layers of forest soils. *Geoderma*. **285**, pp.27–34.

International Tropical Peatlands Center 2018. *RePeat: Generating youth and community for sustainable peatlands in Tumbang Nusa, Pulang Pisau, Central of Kalimantan*. [Online]. [Accessed 19 June 2022]. Jakarta, Indonesia: International Tropical Peatlands Center. Available from: https://www.tropicalpeatlands.org/corporate-news/generating-youth-and-community-for-sustainable-peatlands.

Inubushi, K., Furukawa, Y., Hadi, A., Purnomo, E. and Tsuruta, H. 2003. Seasonal changes of CO_2, CH_4 and N_2O fluxes in relation to land-use change in tropical peatlands located in coastal area of South Kalimantan. *Chemosphere*. **52**(3), pp.603–608.

Ishii, Y., Koizumi, K., Fukami, H., Yamamoto, K., Takahashi, H., Limin, S.H., Kusin, K., Usup, A. and Susilo, G.E. 2016. Groundwater in Peatland *In*: M. Osaki and N. Tsuji, eds. *Tropical Peatland Ecosystems*. Tokyo: Springer Japan, pp.265–279.

Itakura, T., Nakatsugawa, M., Sugimoto, H. and Watanabe, Y. 2016. Characteristics of Watershed in Central Kalimantan *In*: M. Osaki and N. Tsuji, eds. *Tropical Peatland Ecosystems*. Tokyo: Springer Japan, pp.247–263.

Jaenicke, J., Rieley, J.O., Mott, C., Kimman, P. and Siegert, F. 2008. Determination of the amount of carbon stored in Indonesian peatlands. *Geoderma*. **147**(3–4), pp.151–158.

Jaenicke, J., Wösten, H., Budiman, A. and Siegert, F. 2010. Planning hydrological restoration of peatlands in Indonesia to mitigate carbon dioxide emissions. *Mitigation and Adaptation Strategies for Global Change*. **15**(3), pp.223–239.

Jauhiainen, J., Kerojoki, O., Silvennoinen, H., Limin, S. and Vasander, H. 2014. Heterotrophic respiration in drained tropical peat is greatly affected by temperature—a passive ecosystem cooling experiment. *Environmental Research Letters*. **9**(10), article no.: 105013.

Jensen, M.E. and Allen, R.G. 2016. *Evaporation, Evapotranspiration, and Irrigation Water Requirements*. [Online] 2nd ed. American Society of Civil Engineers. Available from: https://ascelibrary.org/doi/abs/10.1061/9780784414057.

Kasih, R.C., Simon, O., Ansori, M., Pratama, M.P. and Wirada, F. 2016. Rewetting on Degraded Tropical Peatland by Canal Blocking Technique in Sebangau National Park, Central Kalimantan, Indonesia *In*: *Proceedings on 15th International Peat Congress 2016. Extended Abstract No: A-065*.

Kelly, T.J., Baird, A.J., Roucoux, K.H., Baker, T.R., Honorio Coronado, E.N., Ríos, M. and Lawson, I.T. 2014. The high hydraulic conductivity of three wooded tropical peat swamps in northeast Peru: measurements and implications for hydrological function. *Hydrological Processes*. **28**(9), pp.3373–3387.

Kelly, T.J., Lawson, I.T., Roucoux, K.H., Baker, T.R., Jones, T.D. and Sanderson, N.K. 2017. The vegetation history of an Amazonian domed peatland. *Palaeogeography, Palaeoclimatology, Palaeoecology*. **468**, pp.129–141.

Kumagai, T., Saitoh, T.M., Sato, Y., Takahashi, H., Manfroi, O.J., Morooka, T., Kuraji, K., Suzuki, M., Yasunari, T. and Komatsu, H. 2005. Annual water balance and seasonality of evapotranspiration in a Bornean tropical rainforest. *Agricultural and Forest Meteorology*. **128**(1–2), pp.81–92.

Kurnianto, S., Selker, J., Boone Kauffman, J., Murdiyarso, D. and Peterson, J.T. 2019.

The influence of land-cover changes on the variability of saturated hydraulic conductivity in tropical peatlands. *Mitigation and Adaptation Strategies for Global Change*. **24**(4), pp.535–555.

Kurnianto, S., Warren, M., Talbot, J., Kauffman, B., Murdiyarso, D. and Frolking, S. 2015. Carbon accumulation of tropical peatlands over millennia: a modeling approach. *Global Change Biology*. **21**(1), pp.431–444.

Lampela, M., Jauhiainen, J., Kämäri, I., Koskinen, M., Tanhuanpää, T., Valkeapää, A. and Vasander, H. 2016. Ground surface microtopography and vegetation patterns in a tropical peat swamp forest. *CATENA*. **139**, pp.127–136.

Lampela, M., Jauhiainen, J., Sarkkola, S. and Vasander, H. 2017. Promising native tree species for reforestation of degraded tropical peatlands. *Forest Ecology and Management*. **394**, pp.52–63.

Limin, S.H., Jentha and Ermiasi, Y. 2007. History of the Development of Tropical Peatland in Central Kalimantan, Indonesia. *Tropics*. **16**(3), pp.291–301.

Lion, M., Kosugi, Y., Takanashi, S., Noguchi, S., Itoh, M., Katsuyama, M., Matsuo, N. and Shamsuddin, S.-A. 2017. Evapotranspiration and water source of a tropical rainforest in peninsular Malaysia. *Hydrological Processes*. **31**(24), pp.4338–4353.

Mackin, F., Barr, A., Rath, P., Eakin, M., Ryan, J., Jeffrey, R. and Fernandez Valverde, F. 2017. *Best practice in raised bog restoration in Ireland. Irish Wildlife Manuals, No. 99*. [Online]. Dublin, Ireland. Available from: http://edepositireland.ie/handle/2262/82140.

Millar, D. J., Cooper, D. J. and Ronayne, M. J. 2018. Groundwater dynamics in mountain peatlands with contrasting climate, vegetation, and hydrogeological setting. *Journal of Hydrology*. **561**, pp.908-917.

Morrogh-Bernard, H., Husson, S., Page, S.E. and Rieley, J.. 2003. Population status of the Bornean orang-utan (Pongo pygmaeus) in the Sebangau peat swamp forest, Central Kalimantan, Indonesia. *Biological Conservation*. **110**(1), pp.141–152.

Murdiyarso, D., Hergoualc'h, K. and Verchot, L. V. 2010. Opportunities for reducing greenhouse gas emissions in tropical peatlands. *Proceedings of the National Academy of Sciences*. **107**(46), pp.19655–19660.

Ochi, N., Okada, M., Hayashi, H., Shimokura, H. and Kamiya, M. 2016. Arrangement and Structure of Weirs on the Kalampangan Canal *In*: M. Osaki and N. Tsuji, eds. *Tropical Peatland Ecosystems*. Tokyo: Springer Japan, pp.313–325.

Page, S.E., Hoscilo, A., Langner, A., Tansey, K., Siegert, F., Limin, S. and Rieley, J.

2009. Tropical peatland fires in Southeast Asia *In*: M. A. Cochrane, ed. *Tropical Fire Ecology*. Springer Berlin Heidelberg, pp.263–287.

Page, S.E., Rieley, J. and Banks, C. 2011. Global and regional importance of the tropical peatland carbon pool. *Global Change Biology*. **17**(2), pp.798–818.

Page, S.E., Rieley, J.O., Shotyk, Ø.W. and Weiss, D. 1999. Interdependence of peat and vegetation in a tropical peat swamp forest D. M. Newbery, T. H. Clutton–Brock, & G. T. Prance, eds. *Philosophical Transactions of the Royal Society of London. Series B: Biological Sciences*. **354**(1391), pp.1885–1897.

Page, S.E., Wűst, R.A.J., Weiss, D., Rieley, J.O., Shotyk, W. and Limin, S.H. 2004. 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. *Journal of Quaternary Science*. **19**(7), pp.625–635.

Payne, R.J., Anderson, A.R., Sloan, T., Gilbert, P., Newton, A., Ratcliffe, J., Mauquoy, D., Jessop, W. and Andersen, R. 2018. The future of peatland forestry in Scotland: balancing economics, carbon and biodiversity. *Scottish Forestry*. **72**(1), pp.34–40.

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

Prananto, J.A., Minasny, B., Comeau, L., Rudiyanto, R. and Grace, P. 2020. Drainage increases CO_2 and N_2O emissions from tropical peat soils. *Global Change Biology*. **26**(8), pp.4583–4600.

President of Indonesia 2020. *President of the Republic of Indonesia Regulation No. 120 Year 2020 about Peatland Restoration and Mangrove Agency (BRGM)*. Ministry of State Secretariat, Indonesia.

Price, J.S., Heathwaite, A.L. and Baird, A.J. 2003. Hydrological processes in abandoned and restored peatlands: An overview of management approaches. *Wetlands Ecology and Management*. **11**, pp.65–83.

Puspitaloka, D., Kim, Y., Purnomo, H. and Fulé, P.Z. 2020. Defining ecological restoration of peatlands in Central Kalimantan, Indonesia. *Restoration Ecology*. **28**(2), pp.435–446.

Putra, E.I., Cochrane, M.A., Vetrita, Y., Graham, L. and Saharjo, B.H. 2018. Determining critical groundwater level to prevent degraded peatland from severe peat fire. *IOP Conference Series: Earth and Environmental Science*. **149**, article no.: 012027.

Rahajoe, J.S., Alhamd, L., Atikah, T.D., Pratama, B.A., Shiodera, S. and Kohyama, T.S.

2016. Floristic Diversity in the Peatland Ecosystems of Central Kalimantan *In*: M. Osaki and N. Tsuji, eds. *Tropical Peatland Ecosystems*. Springer Japan, pp.167–196.

Republic of Indonesia Government 2016. *Indonesian Government Regulation No. 57 Year 2016 about Peatland Ecosystem Protection and Management*. Jakarta, Indonesia: Ministry of State Secretariat, Indonesia.

Ritzema, H., Limin, S., Kusin, K., Jauhiainen, J. and Wösten, H. 2014. Canal blocking strategies for hydrological restoration of degraded tropical peatlands in Central Kalimantan, Indonesia. *CATENA*. **114**, pp.11–20.

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. and Vriesendorp, C.F. 2017. Threats to intact tropical peatlands and opportunities for their conservation. *Conservation Biology*. **31**(6), pp.1283–1292.

Santoso, P.B. and Qirom, M.A. 2020. The fluctuation of peatland water table at Tumbang Nusa, Central Kalimantan. *Jurnal Galam*. **1**(1), pp.27–40.

Sari, A.P., Dohong, A. and Wardhana, B. 2021. Innovative Financing for Peatland Restoration in Indonesia *In*: R. Djalante, J. Jupesta and E. Aldrian, eds. *Climate Change Research, Policy and Actions in Indonesia*. Springer Cham, pp.247–264.

Sinclair, A.L., Graham, L.L.B., Putra, E.I., Saharjo, B.H., Applegate, G., Grover, S.P. and Cochrane, M.A. 2020. Effects of distance from canal and degradation history on peat bulk density in a degraded tropical peatland. *Science of The Total Environment*. **699**, article no.: 134199.

Susilo, G.E., Yamamoto, K., Imai, T., Inoue, T., Takahashi, H., Ishii, Y., Fukami, H., Koizumi, K. and Kusin, K. 2013. Effect of Canal Damming on the Surface Water Level Stability in the Tropical Peatland Area. *Journal of Water and Environment Technology*. **11**(4), pp.263–274.

Takahashi, H., Ibie, B.F., Usup, A., Shimada, S. and Limin, S.H. 2002. Annual changes of water balance and a drought index in a tropical peat swamp forest of Central Kalimantan, Indonesia *In*: J. O. Rieley and S. E. Page, eds. *International Symposium on Tropical Peatlands, Jakarta (Indonesia), 22-23 Aug 2002*. Jakarta, Indonesia: BPPT.

Taufik, M., Minasny, B., McBratney, A.B., Van Dam, J.C., Jones, P.D. and van Lanen, H.A.J. 2020. Human-induced changes in Indonesian peatlands increase drought severity. *Environmental Research Letters*. **15**(8), article no.: 084013.

Taufik, M., Setiawan, B.I. and van Lanen, H.A.J. 2015. Modification of a fire drought index for tropical wetland ecosystems by including water table depth. *Agricultural and Forest Meteorology*. **203**, pp.1–10.

Teh, Y.A., Murphy, W.A., Berrio, J.-C., Boom, A. and Page, S.E. 2017. Seasonal variability in methane and nitrous oxide fluxes from tropical peatlands in the western Amazon basin. *Biogeosciences*. **14**(15), pp.3669–3683.

Uda, S.K., Hein, L. and Atmoko, D. 2019. Assessing the health impacts of peatland fires: a case study for Central Kalimantan, Indonesia. *Environmental Science and Pollution Research*. **26**(30), pp.31315–31327.

Uda, S.K., Hein, L. and Sumarga, E. 2017. Towards sustainable management of Indonesian tropical peatlands. *Wetlands Ecology and Management*. **25**(6), pp.683–701.

Urzainki, I., Laurén, A., Palviainen, M., Haahti, K., Budiman, A., Basuki, I., Netzer, M. and Hökkä, H. 2020. Canal blocking optimization in restoration of drained peatlands. *Biogeosciences*. **17**(19), pp.4769–4784.

Warren, M., Hergoualc'h, K., Kauffman, J.B., Murdiyarso, D. and Kolka, R. 2017. An appraisal of Indonesia's immense peat carbon stock using national peatland maps: uncertainties and potential losses from conversion. *Carbon Balance and Management*. **12**, article no.: 12.

Wichmann, S., Prager, A. and Gaudig, G. 2017. Establishing Sphagnum cultures on bog grassland, cut-over bogs, and floating mats: procedures, costs and area potential in Germany. *Mires and Peat*. **20**, pp.1–19.

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., Wösten, H., Manuri, S., Müller, 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. and Andersen, R. 2017. Denial of long-term issues with agriculture on tropical peatlands will have devastating consequences. *Global Change Biology*. **23**(3), pp.977–982.

Wijedasa, L.S., Vernimmen, R., Page, S.E., Mulyadi, D., Bahri, S., Randi, A., Evans, T.A., Lasmito, Priatna, D., Jensen, R.M. and Hooijer, A. 2020. Distance to forest, mammal and bird dispersal drive natural regeneration on degraded tropical peatland. *Forest Ecology and Management*. **461**, article no.: 117868.

Witts, D., Hall, R., Nichols, G. and Morley, R. 2012. A new depositional and provenance model for the Tanjung Formation, Barito Basin, SE Kalimantan, Indonesia. *Journal of Asian Earth Sciences*. **56**, pp.77–104.

Wong, G.X., Hirata, R., Hirano, T., Kiew, F., Aeries, E.B., Musin, K.K., Waili, J.W., Lo, K.S. and Melling, L. 2018. Micrometeorological measurement of methane flux above a tropical peat swamp forest. *Agricultural and Forest Meteorology*. **256–257**, pp.353–361.

Wooster, M., Gaveau, D., Salim, M., Zhang, T., Xu, W., Green, D., Huijnen, V., Murdiyarso, D., Gunawan, D., Borchard, N., Schirrmann, M., Main, B. and Sepriando, A. 2018. New Tropical Peatland Gas and Particulate Emissions Factors Indicate 2015 Indonesian Fires Released Far More Particulate Matter (but Less Methane) than Current Inventories Imply. *Remote Sensing*. **10**(4), article no.: 495.

Wösten, H., Van Den Berg, J., Van Eijk, P., Gevers, G.J.M., Giesen, W.B.J.T., Hooijer, A., Idris, A., Leenman, P.H., Rais, D.S., Siderius, C., Silvius, M.J., Suryadiputra, N. and Wibisono, I.T. 2006. Interrelationships between Hydrology and Ecology in Fire Degraded Tropical Peat Swamp Forests. *International Journal of Water*

Resources Development. **22**(1), pp.157–174.

Wösten, H., Clymans, E., Page, S.E., Rieley, J.O. and Limin, S.H. 2008. Peat–water interrelationships in a tropical peatland ecosystem in Southeast Asia. *CATENA*. **73**(2), pp.212–224.

Xu, J., Morris, P.J., Liu, J. and Holden, J. 2018. PEATMAP: Refining estimates of global peatland distribution based on a meta-analysis. *CATENA*. **160**, pp.134–140.

Young, D.M., Baird, A.J., Morris, P.J. and Holden, J. 2017. Simulating the long-term impacts of drainage and restoration on the ecohydrology of peatlands. *Water Resources Research*. **53**(8), pp.6510–6522.

Chapter III

Water-table responses to storms in Sebangau tropical peatland, Kalimantan, Indonesia

Abstract

The role of tropical peatlands in carbon capture and storage has been severely diminished due to drainage. Governments have promoted hydrological restoration in some regions, usually using ditch dams. However, the effects of ditch dams on water-table responses to storms in tropical peatlands are poorly understood. We collected hourly rainfall and water-level data during the dry and wet seasons (August 2019 to January 2020) at a forested peatland (Forested), a drained peatland with ditch dams (Blocked), and a drained peatland without ditch dams (Drained) in Sebangau National Park, Kalimantan, Indonesia. Hydraulic conductivity of the surface peat and bulk density of the peat profiles were also measured. The two main components of a Principal Component Analysis (PCA) could explain between 62% and 68% of the variation of water-table responses to storms in the studied sites. We found responses were very different between the intact, drained and ditch-dammed systems. In the Forested site, the mean of the post-storm water-level drawdown speed (DSpeed) was 0.039 cm $hour^{-1}$ (SD = 0.024 cm $hour^{-1}$) when the water table was deeper than 50 cm below the surface but 0.047 cm $hour^{-1}$ (SD = 0.039 cm $hour^{-1}$) when the water table was within the upper 50 cm. In the Drained/Blocked sites, DSpeed varied greatly with depth, distance to ditches, and distance to the main outlet of ditches. Water-table responses to storms were particularly related to the initial water-table position, the depth and the duration of the storm, and the position with respect to ditches. These factors need to be considered in hydrological modelling studies to better represent the water-level dynamics in tropical peatlands with different restoration conditions. We show that ditch dams alone may not be sufficient to promote restoration of hydrological functions in drained peatlands toward those observed in undrained forested systems.

3.1 Introduction

Tropical peatlands cover ~ 1 million km^2 (Ruwaimana et al., 2020; Xu et al., 2018), storing an estimated 102.24 Pg of carbon (Dargie et al., 2017; Honorio Coronado et al., 2021; Miettinen et al., 2017), around 19% of the global peatland carbon store (528 Pg of carbon) (Hodgkins et al., 2018; Yu et al., 2010). They are also important for biodiversity and a range of ecosystem services (Husson et al., 2018; Schulz et al., 2019; Wijedasa et al., 2020). Many tropical peatlands are dome shaped and ombrotrophic, bounded by open water bodies, such as the Sebangau peat dome in Indonesia (Berninger and Siegert, 2020), and domes in the *Cuvette Centrale* in the Congo Basin (Davenport et al., 2020), in Changuinola in the Province of Bocas del Toro, Panamá (Phillips et al., 1997), and in Pastaza-Marañón, Peru (Roucoux et al., 2017). These domes range between 2 and 20 km in width (Dommain et al., 2014; Ishii et al., 2016; Dargie et al., 2017; Kelly et al., 2020), with peat depths typically between 2 and 12 m at the midpoint of the dome (Page et al., 2004; Warren et al., 2012; Lähteenoja et al., 2013; Hapsari et al., 2017). The development of a peat dome has a limit that is related to not only physical but also biochemical factors of the system (Anderson and Muller, 1975; Winston, 1994; Patterson and Anderson, 2000; Anderson and Peace, 2017).

Many tropical peatlands have been drained with canals and ditches to lower the water table (Lilleskov et al., 2019; Dadap et al., 2021). Deepening of water tables makes drained peatlands more prone to fire and ecological degradation (Dohong et al., 2017; Putra et al., 2018; Agus et al., 2020). Peatland drainage, such as in the ex-Mega Rice Project area in Central Kalimantan, Indonesia, may also affect the water-table regime (Wösten et al., 2008; Ishii et al., 2016; Vernimmen et al., 2020; Putra et al., 2021). Some governments have committed to undertake topical peatland restoration, as addressed by representatives of Democratic Republic of the Congo, Republic of the Congo, Republic of Peru, and Republic of Indonesia during the "Brazzaville Declaration", 21 to 23 March 2018 (Desai, 2017; International Climate Initiative, 2021). International Climate Initiative (2021) stated that the commitment in preserving peat carbon is one of the main drivers behind government investments in peatland restoration. The restoration measures, such as installation of ditch dams, are used to hold water on site by lowering hydraulic gradients (Ritzema et al., 2014; Kasih et al., 2016; Putra et al., 2021).

While several studies have examined tropical peatland water tables (e.g., Cobb et al. (2017), Cobb & Harvey (2019), Deshmukh et al. (2021), Marwanto et al. (2018), Mezbahuddin et al. (2015), Sutikno et al. (2019), Wösten, Van Den Berg, et al. (2006), Wösten, Hooijer, et al. (2006), and Putra et al. (2021)), as far as we are aware, there have been no studies examining the effect of different storm variables, such as duration and intensity, on water-table dynamics in tropical peatlands. Such information may help practitioners understand more fully (than just simple seasonal measures of water-table depth) whether ditch blocking promotes restoration of the hydrological functioning of peatlands. Understanding how peatland water tables respond to rainfall events, and the controls on these responses, can also be important to test and possibly improve some models of peatland hydrology, development and carbon cycling that currently are operating at sub-daily scales, for example Ecosys, DigiBog_Hydro, PDM, TROPP-CAT, and Linear Emission Model (Mezbahuddin et al., 2014; Farmer et al., 2014; Hoyt et al., 2019; Young et al., 2019). Additionally, such an understanding may aid predictions of how tropical peatlands will respond to different intensities and durations of rainfall under future climate change (Li et al., 2007; IPCC, 2021).

In temperate peatlands, the responses of water tables to rainfall events have been found to be different between intact and restored systems, suggesting there is, at least, a very long lag time before the hydrological function recovers (Holden et al., 2011; Williamson et al., 2017; Kreyling et al., 2021). It is therefore hypothesised that the responses of water table to storms will be different between intact, drained and ditch-dammed tropical peatlands.

This chapter examines the responses of water tables to storms in the Sebangau tropical peatland, Indonesia, considering the variation of restoration condition among sites (undrained forest, drained, and ditch-dammed systems). The main research questions are:

1. What storm variables significantly influence the water-table dynamics in tropical peatlands with different management conditions?
2. How do ditches and ditch dams alter the response of water tables to storms in tropical peatlands?
3. How does the variation in peat properties contribute to the differences in the response of water tables to storms in tropical peatlands?

3.2 Methodology

3.2.1 Study sites

We studied three peatland sites in Sebangau, Kalimantan, Indonesia. These were a drained peatland with open ditches (Drained), a peatland where the ditches had dams installed (Blocked), and a relatively intact forested peatland (Forested) (see Putra et al. (2021)). The monitoring period of this study was between 22 August 2019 and 17 January 2020, covering water-table variations during the late dry and early wet periods.

The layout of the study sites is presented in Figure 3.1. The Drained and Blocked sites were deforested systems. The ditches in Drained and Blocked were between 1.5 to 2 m deep and 2 to 4 m wide. There were four ditch dams constructed in Blocked between 2016 and 2018, still in operation during the study. The Forested site was a relatively intact forested system. Other than that, a single narrow 60 cm deep (but dammed) trench was present in Forested, formed by logs being dragged from the site during former forestry operations in the 1990s, and the land remained forested throughout the monitoring period.

Putra et al. (2021) found that during the end of dry season 2019, water tables at the study sites were deeper than 40 cm from the peat surface and the ditches were dry. It was also found that in the wet season, water flowed from the peatland to ditches in Drained and Blocked. In Drained, in the wet season, patchy inundation (shallow ponding between 1 and 5 cm) was observed in areas furthest from ditches. In Blocked, in the wet season, large areas around ditches had surface water to depths up to 23 cm. In Forested, for a short period of the wet season (2 January 2020 to 15 January 2020), some shallow inundation (less than 10 cm) was noted, especially in microtopographic depressions, while the water-table depths at monitoring wells ranged between 9 and 18 cm.

A topographical survey was conducted to provide absolute water-level profiles across the study sites. In this study, in Drained and Blocked, the absolute water levels for wells were calculated based on local benchmarks (note that these benchmarks were different from those presented by Putra et al. (2021)). The benchmarks were the surface elevations of well AA2 in Drained and well BB2 in Blocked (Figure 3.1), which were the wells with the highest water level at each site in this analysis. The benchmark for Forested was beside well AL0 (2.3894°S, 113.4524°E), as in Putra et al. (2021). The elevations of these benchmarks were set at 0 cm.

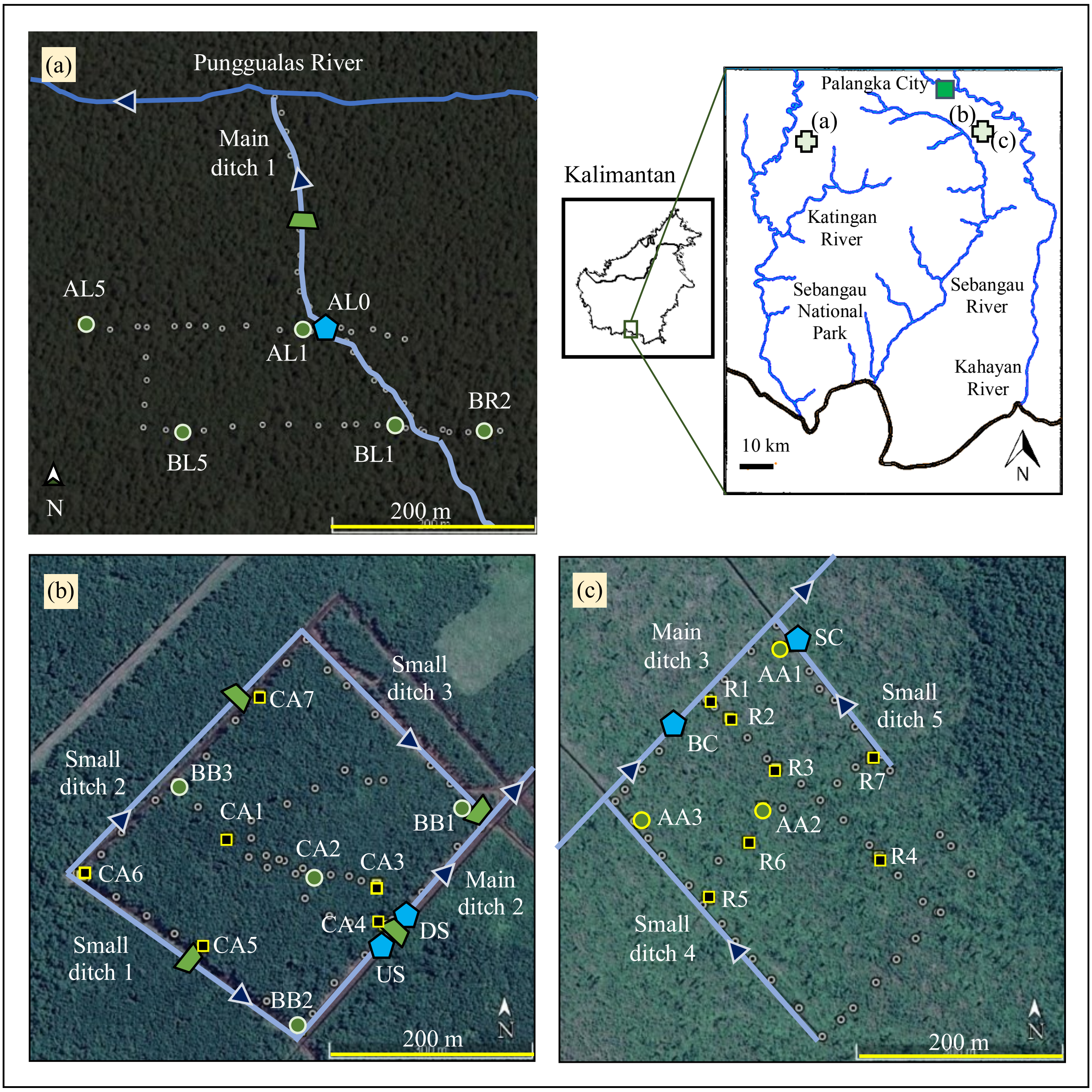

Figure 3.1 The schematization of the study sites, which were Forested (a), Blocked (b), and Drained (c). The green square in the top right map shows the location of Palangka City, and the plus signs are the study locations. In (a) to (c) larger circles symbolize logger wells, smaller circles are levelling points, and squares are testing points. Blue pentagons are ditch loggers and trapezoids are dams. The dark blue continuous line represents a river, and the light blue lines are ditches. The main flow directions are shown by arrows. Peat cores were taken at AA1, AA2, AA3, BB1, BB2, BB3, AL0, AL1, and AL5. Surface hydraulic conductivity tests (*K*) in Drained/Blocked were conducted at all logger wells, all points with label CA, and all points with label R. Surface *K* tests in Forested were conducted only at well AL1 and AL5.

3.2.2 Rainfall and water-level data

Rainfall data were obtained from two automatic gauges (gauges not shown in Figure 3.1). One was installed at Tumbang Nusa Camp (2.3556°S, 114.0896°E), which was 2.7 km from Drained and 3.5 km from Blocked. The second was located at Punggualas Camp (2.3865°S, 113.4453°E), which was 0.8 km from Forested. Rainfall data were collected at 30-minute intervals. Water-level data at 180-minute intervals were collected from automatically-logged monitoring wells: thirteen in the peat at various distances from the ditches and four in the ditches themselves (Figure 3.1). The water-table loggers were vented (In-situ Level TROLL 500), but the ditch water-level loggers were non-vented (Schlumberger Diver), accompanied by a barometric pressure logger installed above the ground. The monitoring wells were lined with perforated plastic tubing that was 6.4 cm in diameter and 200 cm deep. The top of the tubing was 50 cm above the ground surface, allowing for access to the well during periods when the peatland was inundated.

Storm events were identified from the rainfall data. A storm event was defined as a series of 30-minute rainfall values, separated by zero mm of rainfall at the beginning and end of the event. If there were two or more adjacent rainfall events within 24 hours, the rainfall events were considered as a single storm event. If there was a single rainfall event (no other rainfall event within 24 hours before and after) followed by an increase of water table at any well (within 24 hours after the rainfall event), that single rainfall event was considered as a storm event but otherwise it was discarded. The number of storms considered in a principal components analysis (see below, section 3.2.3) was different at each well, as a storm did not always result in a rise in water tables at all wells. In total, for the whole monitoring period, there were 87 identified storm events in Drained, 142 in Blocked, and 151 in Forested. The storm variables listed in Table 3.1 were then extracted for each identified storm event.

The water-level recession patterns during the periods without rain were extracted from the records from the automatically logged wells (13 wells) and from the ditches (at the Main ditch 3 monitoring point –BC– in Drained and at the downstream of the main-dam monitoring point –DS– in Blocked, see Figure 3.1). Water-level drawdown was calculated by subtracting water level at time j from the water level at time i during the recession period, where time j equals time i plus 3 hours. Water-level drawdown speed

(DSpeed) was calculated by dividing the water-level drawdown value by 3 hours. The hydraulic head difference (HHead) was calculated by subtracting the water level at the ditch (at point BC in Drained and DS in Blocked) from the peatland water level. A negative HHead value indicates that the ditch water level was higher than the peatland water level at that specific time.

3.2.3 Storm-event data analysis

The storm variables listed in Table 3.1 were included in a principal component analysis (PCA) (Westra et al., 2007; Jolliffe and Cadima, 2016; Maity, 2018; Lai and Kuok, 2019), to identify the main controls on the overall variation of the water table in response to storms. The PCA was also used to establish whether the set of variables which were significantly correlated to the top two principal components, were different for each land management condition.

For each analysis, a matrix with individual storms on the *x*-axis and the eleven storm variables on the *y*-axis was processed using the R package FactoMineR (Lê et al., 2008), following the protocol of Kassambara (2017). The PCA was conducted to examine the responses of water tables to storms in all three peatland sites (Drained, Blocked, and Forested), for two water-table categories (shallow: water-table data within 50 cm of the surface; and deep: water table deeper than 50 cm below surface). If the water-table rise crossed the 50-cm level, and if the rise above the 50-cm level was larger than the rise below, that water-table rise was categorized in the shallow category. The PCAs were conducted for individual wells to examine spatial variations of the response of water tables to storms but only included the shallow water-table category, because the number of storms recorded in the deep water-table category was less than fifteen at each well. Rainfall rates greater than 7 mm hour^{-1} in Drained/Blocked were associated with surface inundation during shallow water-table conditions and were not included in the analysis.

Table 3.1 Hydrological variables extracted from the storm profiles.

Variables	**Units**	**Description**
Storm variables		
Initial-vars		Variables that are related to conditions before the water table started to rise
• rR	mm	Storm rainfall before rise (storm rainfall sum before first water-level rise)
• dLS	hour	Duration from the last rainfall event to first rainfall in this storm
• dSR	hour	Duration from first rainfall to first rise in water level
• dP2	hour	Duration from first rainfall to peak rainfall
Rising-vars		Variables that were related to conditions between the start of rise and the water-table peak
• dSP	hour	Duration from first rainfall to peak water level
• dPP	hour	Duration from peak rainfall to peak water level
• dRP	hour	Duration from first rise in water level to peak water level
Peak-vars		Variables that were related to peak rainfall conditions
• 30MP	$mm.hour^{-1}$	Peak rainfall intensity in 30 minutes interval
• 1HP	$mm.hour^{-1}$	Peak rainfall intensity in hourly interval
• pR	mm	Storm rainfall before peak (storm rainfall sum before peak water level was reached)
Unique variable		Variable that cannot be grouped with others
• sRI	$mm.hour^{-1}$	Storm rainfall intensity (total storm rainfall depth divided by storm duration)
Recession variables		
• DSpeed	$cm\ hour^{-1}$	Result of (water level at time j minus at time i) divided by (time j minus time i), where $j > i$
• HHead	cm	Result of subtracting (water level at the lowest ditch) from (water level at well) for time i

3.2.4 Dry bulk density and hydraulic conductivity

In order to support the interpretation of the PCA results, three peat cores were extracted from each site (Figure 3.1) and sampled for dry bulk density and organic matter content. The sample locations were chosen to capture the spatial variability of peat properties with reference to the layout of the ditches. The cores were extracted in 50 cm sections, up to 200 cm deep, using a Russian peat sampler (Eijkelkamp Soil and Water, 2020) of 52 mm internal diameter. The cores were stored in PVC casing, wrapped with cling film, and transported to the soil laboratory at the University of Palangka Raya. The samples were then cut into 2 cm lengths for the dry bulk density determination, although longer samples up to 10 cm in length were used if the samples were too fibric (poorly decomposed peat). The bulk density samples were dried at 105 °C for at least 24 hours (Chambers et al., 2011). The organic matter content was determined for samples of 2 cm length taken at upper depths of 2 cm, 23 cm, 46 cm, 96 cm, 146 cm, and 196 cm from the top of each peat core. However, the sample at 23 cm depth for point BB3 (Figure 3.1), at 146 cm for AL1, and at 2 cm for AL5 were not tested for organic matter as those were too fibric. The samples were heated in the soil furnace at 850 to 900 °C (Hoogsteen et al., 2015) to remove organics (for at least 5 hours) and later weighed at room temperature (25 °C).

Minidisk tension infiltrometers of 4.4 cm diameter (Meter Group, 2020) were used to measure saturated hydraulic conductivity at and close to the peatland surface. The infiltrometer tests were conducted at the peat surface in the Drained (ten locations), Blocked (eleven locations), and Forested (only at point AL1 and AL5) sites (see Figure 3.1). Six 'sub-surface' tests, at 20 cm below the original surface were conducted at point CA1, CA2, CA3, CA4, R2, and R3. The top peat was carefully removed to undertake the sampling with minimal disturbance of the peat at 20 cm below the surface. Two pressure head states (minimum) were required to calculate near-saturated hydraulic conductivity values using the technique outlined by Reynolds & Elrick (1991) and Baird (1997); pressure heads of 0 cm and -1 cm were used in this study.

3.3 Results

Overall, water levels in the peatlands had different responses to storms, depending on the initial water-table condition, the depth-duration patterns of the storm, and the position with respect to ditches. The timing and magnitude of water-table increase in response to storm events were unique for each monitoring well. Table 3.2 shows the mean values for the storm variables for each monitoring well during the shallow water-table condition.

Table 3.2 Mean values for storm variables for each monitoring well during shallow water-table conditions. Standard deviations are in brackets. The variable codes are as given in Table 3.1. Column ns contains the number of storms included in the PCA.

Well	Location	ns	sRI mm hour^{-1}	30MP mm hour^{-1}	1HP mm hour^{-1}	rR mm	pR mm	dLS hour	dSR hour	dSP hour	dPP hour	dRP hour	dP2 hour
AA1	Drained	15	3.8[2.9]	14.3[11.1]	19.2[16.9]	4.9[8.5]	34.9[28.9]	23.0[14.1]	2.0[2.8]	11.0[6.1]	9.4[5.8]	9.0[5.9]	1.6[1.9]
AA2	Drained	15	2.9[2.3]	9.5[6.7]	12.5[9.5]	1.1[2.2]	22.3[18.3]	22.1[14.7]	1.2[2.1]	7.4[3.4]	5.6[3.6]	6.2[3.5]	1.8[2.0]
AA3	Drained	19	3.4[2.8]	12.0[10.9]	15.9[16.4]	3.7[6.8]	27.3[27.4]	20.9[13.5]	3.2[7.4]	11.7[8.1]	10.2[8.5]	8.5[6.2]	1.4[1.8]
BB1	Blocked	17	3.4[2.9]	10.3[9.1]	14.8[15.9]	3.3[7.2]	27.0[28.3]	21.5[14.2]	1.2[2.1]	11.1[8.6]	9.6[8.9]	9.9[9.0]	1.5[1.8]
BB2	Blocked	23	3.1[2.7]	10.3[10.6]	13.8[15.6]	3.2[7.5]	24.9[26.5]	20.1[13.6]	1.3[2.8]	9.3[4.5]	7.8[4.5]	8.0[4.8]	1.5[1.8]
BB3	Blocked	20	3.2[2.7]	11.2[10.9]	15.0[16.2]	5.0[8.4]	28.8[27.9]	21.9[13.5]	2.6[5.2]	15.3[9.4]	14.1[9.2]	12.8[8.7]	1.2[1.4]
CA2	Blocked	22	2.8[2.7]	8.4[8.9]	11.9[15.1]	3.4[7.5]	21.0[26.8]	20.2[12.8]	1.2[1.7]	10.1[8.0]	9.1[7.8]	8.9[7.0]	1.0[1.3]
AL0	Forested	19	8.7[12.5]	11.7[11.0]	15.6[16.4]	10.7[20.4]	25.0[23.6]	39.7[28.8]	1.6[2.0]	27.9[14.9]	26.2[14.4]	26.4[14.6]	1.7[2.0]
AL1	Forested	20	9.2[12.5]	10.9[11.1]	14.6[16.5]	11.4[22.2]	23.6[24.3]	41.6[27.3]	1.4[2.0]	23.0[17.2]	21.4[16.5]	21.6[16.4]	1.5[2.0]
AL5	Forested	17	9.5[13.0]	12.3[11.5]	16.6[17.1]	5.1[10.5]	27.3[25.1]	43.6[28.6]	1.2[2.1]	28.1[14.5]	26.3[13.8]	26.8[14.5]	1.8[2.0]
BL1	Forested	23	8.5[11.5]	10.6[10.4]	14.0[15.4]	9.1[18.7]	21.6[22.6]	37.9[27.4]	1.7[1.9]	21.1[12.7]	19.7[12.4]	19.4[12.1]	1.4[1.9]
BL5	Forested	23	8.5[11.5]	10.6[10.4]	14.0[15.4]	9.0[18.7]	21.4[22.7]	37.6[27.7]	1.7[2.1]	24.4[16.9]	22.9[16.4]	22.7[16.1]	1.4[1.9]
BR2	Forested	22	8.7[11.7]	11.0[10.4]	14.5[15.5]	10.0[19.0]	22.5[22.7]	37.2[27.8]	1.9[2.1]	13.4[30.7]	18.0[11.1]	17.6[10.6]	1.5[1.9]

3.3.1 Storm controls on water-table response between sites

Figure 3.2 shows the contributions of each storm variable to Principal Components 1 and 2 (PC1 and PC2) that were different between sites and seasonal conditions. The responses to storms varied between the sites and the responses when water tables were deep differed from the responses when the water table was shallow (Figure 3.2). For the site and season (shallow/deep water table) categories, PC1 and PC2 represented the variation in the responses to storms by between 62% (Drained-deep) and 68% (Forested-shallow). In terms of individual variable contributions to PC1 and PC2, the eleven storm variables can be placed into three groups plus a further individual variable (it had a unique response for each category of the analysis). The groups are Initial-vars, Rising-vars, and Peak-vars, whereas the ungrouped variable is sRI (Storm rainfall intensity) (see Table 3.1). In general, Rising-vars and Peak-vars were important and gave substantial contributions to PC1 and PC2, while Initial-vars did not. Storm rainfall intensity was important and provided substantial contributions to PC1 and PC2 for Forested-deep, Blocked-shallow, and Drained-shallow, but not for other categories.

The vector directions for the storm variables varied among categories of analysis. The vectors that positively contributed to both PC1 and PC2 are presented with a darker colour in Figure 3.2 (see the blue scale bar). For Forested-deep, Peak-vars and Storm rainfall intensity contributed positively to both PC1 and PC2 but Rising-vars only contributed positively to PC1. The responses in Drained were different from those in Forested and Blocked because no storm variables contributed positively to PC1, except for dSR (the duration from first rainfall to first water-level rise) that contributed only 3% of the variance. In Drained-deep, Rising-vars strongly contributed to PC1 and Peak-vars to PC2, but in Drained-shallow the reverse was the case.

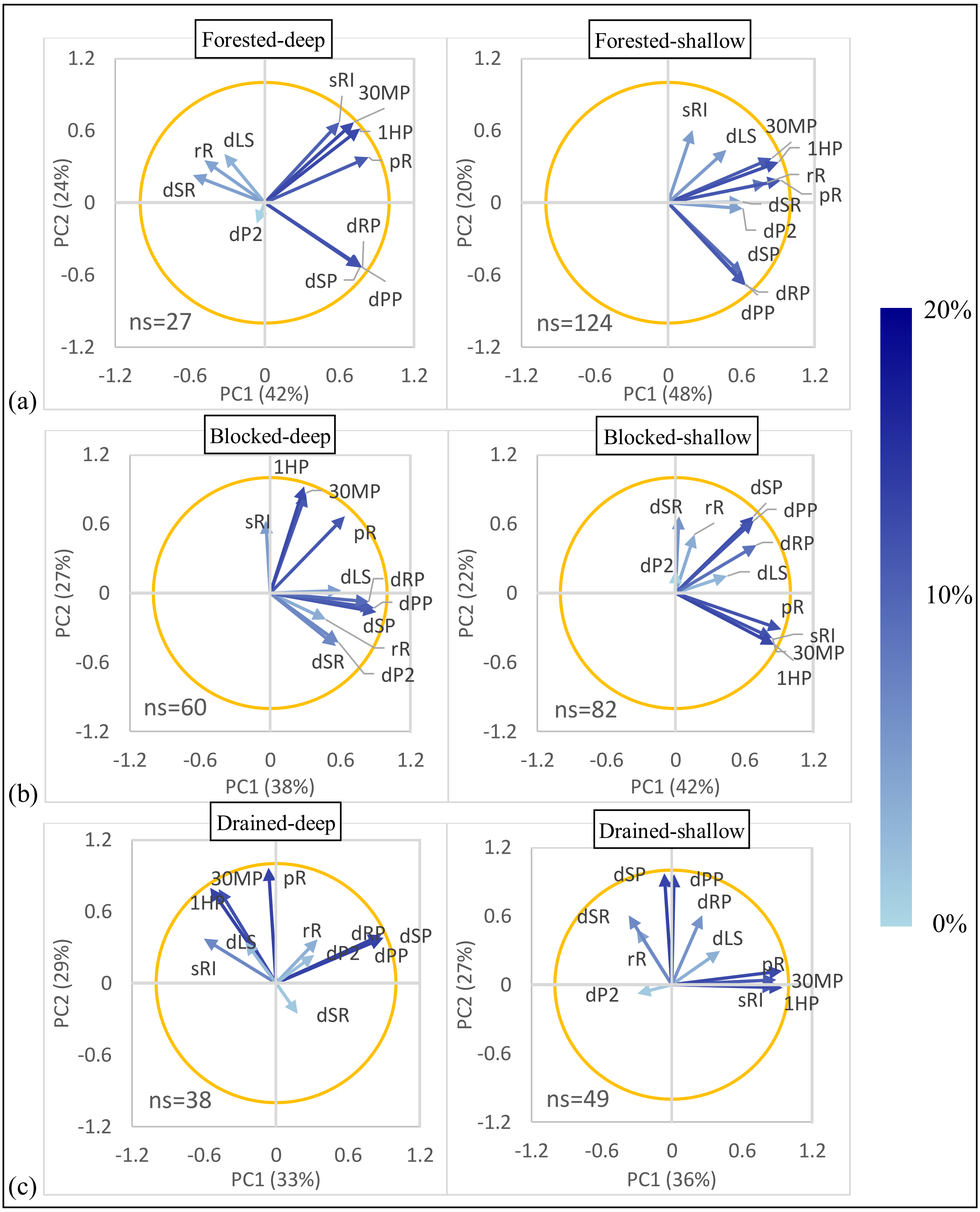

Figure 3.2 Principal Components 1 and 2 (PC1 and PC2) with the distribution of each hydrological variable for the three sites. The variable codes are provided in Table 3.1. The left-hand figures show the variations under the deep water-table conditions (deep/dry period), while the right-hand figures show the shallower water-table conditions (shallow/wet period). The percentage attributed to the axes indicates the variance that is represented by the PC. The arrows show the direction and the quality of representation for each variable to PC1 and PC2 within the maximum quality radius of 1 (yellow circle). The colour of the arrows shows the contribution of each variable to PC1 and PC2, in which the shaded bar indicates the percentage of the contributions. The number of storms (ns) used in the analysis for each location is stated.

3.3.2 Storm controls on water-table response between wells

The responses of water tables to storms varied spatially within sites. Table 3.3 shows the contribution of each storm variable to PC1 and PC2 at each individual well, during the shallow water-table conditions. The variances of the response to storms at each individual well accounted for by PC1 and PC2, were between 64.1% (at AA3) and 73.9% (at AL1). For wells in Drained, during the shallow water-table condition, the storm variables with the largest contribution were generally the Rising-vars, showing the dominant effect of storm duration over storm intensity. The Peak-vars provided the largest contribution to the water-table variations at wells in Blocked during the shallow water-table conditions, suggesting that storm intensity contributed more than storm duration. Initial-vars did not provide substantial contributions to water-table variations at all sites.

Table 3.3 The contributions of storm variables to Principal Components 1 and 2 at individual wells, during the shallow water-table conditions. The codes are explained in Table 3.1. Column ns contains the number of storms considered. Column sum.pov contains the variances of the response to storms that are represented by PC1 and PC2. For each well, storm variables with the largest contribution are indicated by †.

Well	Location	ns	sum.pov (%)	Contribution to PC1 and PC2 (%)										
				------Initial-vars------					-----Rising-vars-----			-----Peak-vars-----		
				sRI	rR	dLS	dSR	dP2	dSP	dPP	dRP	30MP	1HP	pR
AA1	Drained	15	64.3	10.4	3.0	2.2	7.1	2.4	13.7†	12.9	11.3	11.4	12.8	13.0
AA2	Drained	15	72.0	9.0	7.7	7.4	9.1	8.2	7.6	10.8	11.2†	9.7	11.0	8.5
AA3	Drained	19	64.1	11.7	6.2	4.9	9.3	1.4	13.5†	13.3	2.4	11.7	12.9	12.7
BB1	Blocked	17	68.7	10.8	11.1	7.1	8.1	1.3	9.2	9.3	10.5	10.3	10.3	11.8†
BB2	Blocked	23	66.6	9.8	10.6	4.0	11.9†	7.1	7.3	7.6	9.1	10.2	11.0	11.4
BB3	Blocked	20	64.4	12.4	5.1	3.6	5.7	0.6	12.9	12.7	7.8	12.3	13.4	13.5†
CA2	Blocked	22	70.8	11.7	3.3	3.3	9.6	1.9	11.9	11.1	10.7	11.7	12.1	12.6†
AL0	Forested	19	72.9	5.0	9.7	4.3	6.9	5.0	12.2†	11.8	11.9	10.3	11.5	11.5
AL1	Forested	20	73.9	5.7	9.6	5.5	7.4	4.9	11.3	10.7	10.8	11.2	11.7†	11.2
AL5	Forested	17	67.6	8.1	8.4	7.0	6.6	4.3	11.7†	11.2	11.6	10.4	10.8	9.8
BL1	Forested	23	70.9	4.7	9.8	5.3	5.6	4.6	12.2†	11.7	11.5	11.0	12.1	11.4
BL5	Forested	23	69.2	2.1	9.5	5.4	5.1	4.8	12.9†	12.6	12.4	11.1	12.4	11.7
BR2	Forested	22	67.8	9.3	10.8	6.3	6.8	5.6	3.2	10.8	10.6	11.9	12.7†	11.9

In Drained, the contributions of storm variables to the variation in PC1 and PC2 at wells near to ditches (AA1 and AA3) were different from those at the more distant well (AA2) (Table 3.3). The duration from first rainfall to peak water level (dSP) provided the largest contribution to variance in PC1 and PC2 at AA1 and AA3, while the duration from first rise in water level to peak water level (dRP) provided the greatest contribution at AA2. In Blocked, the contributions of storm variables to the variation of the water table at wells near to the main ditch outlet (BB1) were different from more distant wells (CA2 and BB3). Although all of the wells in Blocked had strong contributions to water-table response from pR (Storm rainfall before peak), CA2 and BB3 had low contributions from Initial-vars and high contributions from Storm rainfall intensity. In Forested, storm response at all wells had strong contributions from Rising-vars and Peak-vars, with low contributions from Initial-vars and Storm rainfall intensity to PC1 and PC2. Only well BR2 had water-table responses that were not strongly affected by Rising-vars.

3.3.3 Water-table drawdown

Figure 3.3 shows the distribution of standardized water-level drawdown speed (DSpeed) and hydraulic head (HHead) data in the study sites. In brief, Figure 3.3 conveys that there were different patterns of water-table drawdown between seasons and between wells in the study sites.

In periods without rain, the water tables of tropical peatlands decreased as a result of the on-site differences in hydraulic head, the high permeability of peat (Baird et al., 2017; Kurnianto et al., 2019), and the evapotranspirative demand (evapotranspiration data were presented by Putra et al. (2021)). Our results show that the distribution of water-level drawdown speed (DSpeed) and hydraulic head (HHead) varied with depth and depended on site conditions (Figure 3.3). In Forested, drawdown speed varied slightly with depth, with deep peat having a lower variability of drawdown speed values than shallow peat. In Forested, the variation of drawdown speed values was similar among wells. In Forested, the mean drawdown speed values for deep peat were 0.038 (SD = 0.020) cm hour^{-1} at well AL0, 0.039 (SD = 0.024) cm hour^{-1} at well AL1, and 0.039 (SD = 0.026) cm hour^{-1} at well AL5. Furthermore, in Forested, the mean drawdown speed values for shallow peat were 0.045 (SD = 0.032) cm hour^{-1} at well AL0, 0.047 (SD = 0.039) cm hour^{-1} at well AL1, and 0.048 (SD = 0.057) cm hour^{-1} at well AL5. In

Forested, the hydraulic head data were not calculated because water-level differences among wells were relatively small (less than 10 cm).

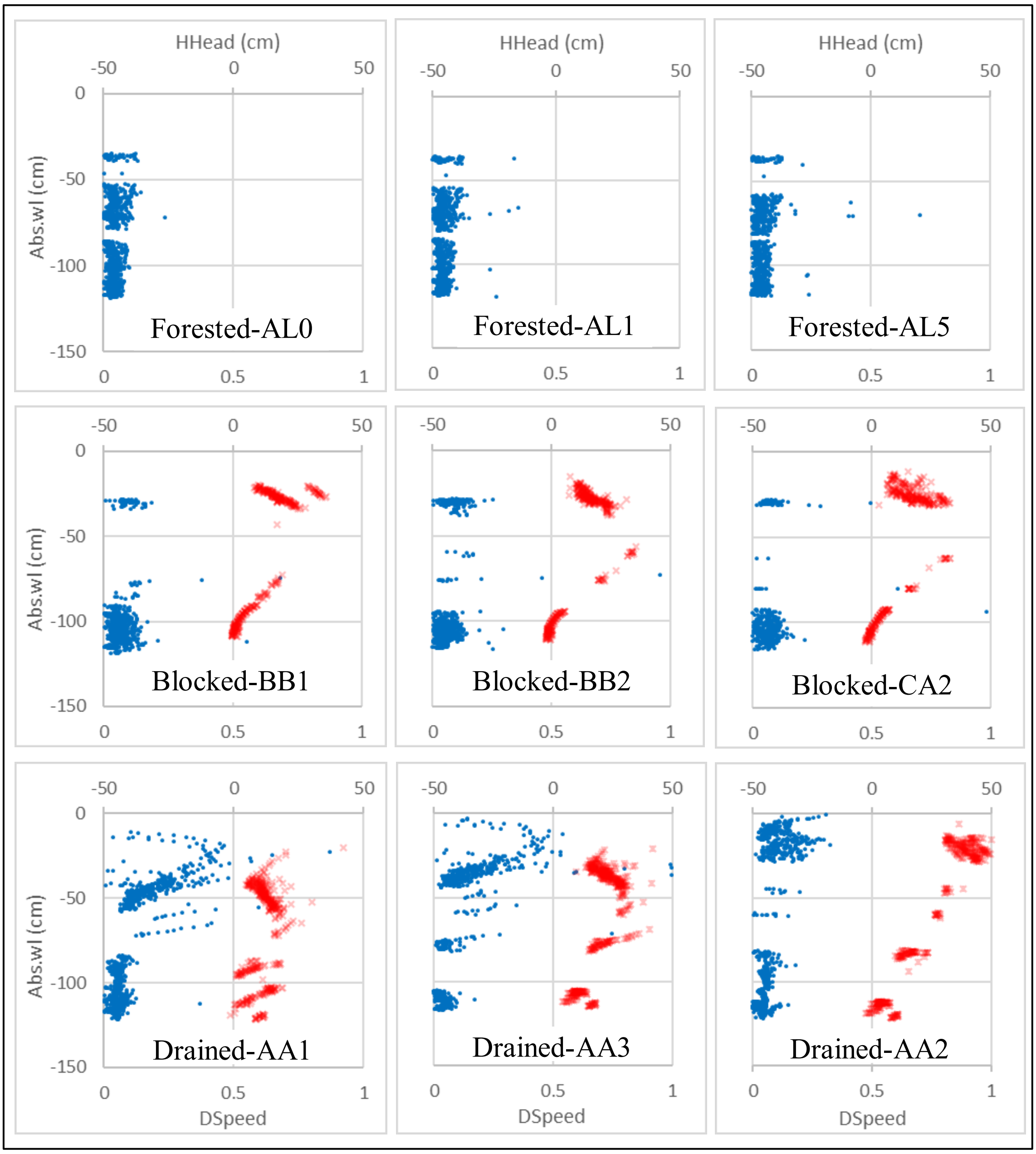

Figure 3.3 The distribution of standardized water-level drawdown speed (DSpeed) and hydraulic head (HHead) data in Forested, Blocked and Drained. The scattered blue-filled circles are the DSpeed data (left side in each graph). The scattered red-crosses are the HHead data (right side in each graph). Negative HHead data indicates that the ditch water level was higher than the well water level at that time. No HHead data were available for Forested.

In Blocked, water-level drawdown speed (DSpeed) differed with depth and with distance to the main ditch outlet. Drawdown speed for deeper peat in Blocked was more variable than in Forested and Drained (Figure 3.3). Also, drawdown speed at the well furthest from the main ditch outlet (CA2) was more variable (larger SD) than at the nearest well (BB1). In Blocked, the mean drawdown speed values for deeper peat were 0.061 (SD = 0.037) cm $hour^{-1}$ at well BB1, 0.050 (SD = 0.035) cm $hour^{-1}$ at well BB2, and 0.065 (SD = 0.067) cm $hour^{-1}$ at well CA2. In Blocked, the mean drawdown speed values for shallower peat were 0.109 (SD = 0.085) cm $hour^{-1}$ at well BB1, 0.107 (SD = 0.125) cm $hour^{-1}$ at well BB2, and 0.087 (SD = 0.070) cm $hour^{-1}$ at well CA2.

In Drained, drawdown speed (DSpeed) varied with depth and with distance to ditches, especially for shallower depths (see Figure 3.3). In Drained, for either shallow or deep peat, the variation of drawdown speed data at well AA2 (farther from ditches) was less than at well AA1 and AA3 (closer to ditches). In Drained, the mean drawdown speed values for shallow peat were 0.200 (SD = 0.112) cm $hour^{-1}$ at well AA1, 0.187 (SD = 0.195) cm $hour^{-1}$ at well AA3, and 0.114 (SD = 0.061) cm $hour^{-1}$ at well AA2. In Drained, the drawdown speed rates for deep peat were similar to those in Forested and Blocked, with a mean of 0.052 (SD = 0.026) cm $hour^{-1}$ at well AA1, 0.035 (SD = 0.023) cm $hour^{-1}$ at well AA3, and 0.042 (SD = 0.024) cm $hour^{-1}$ at well AA2.

Hydraulic head (HHead) varied with depth but differently between Blocked and Drained. In Blocked, hydraulic head values were small (near to zero) for either the deepest or the shallowest water tables. In Drained, hydraulic head at AA2 had a different pattern from that at AA1 and AA3. The hydraulic head data at AA2 varied with depth, with the hydraulic head values being larger when the water table was deeper (mean = 37.3 cm, SD = 8.2 cm) than when it was shallow (mean = 4.4 cm, SD = 3.3 cm). In Drained, during deep water-table periods, the hydraulic head data of well AA1 and AA3 increased as the water level of the peatland increased. At several intervals, hydraulic head dropped near to zero, indicating that there were some rapid increases of the ditch water level during the intervals (see Figure 3.3). During shallow water-table periods, the hydraulic head data of well AA1 and AA3 decreased with reductions in water-table depth, due to gradual increases of the ditch water level.

3.3.4 Peat properties at well locations

Table 3.4 presents near-surface saturated hydraulic conductivity (*K*) data for the studied wells. In Table 3.4, each datum from the Drained site is accompanied by the mean distance of each well to nearest ditch and each datum from the Blocked site by the distance of each well to lowest ditch outlet. The distances of well AL1 and well AL5 to the small trench at Forested are not presented in Table 3.4. Table 3.5 contains simple statistical data of mean peat dry bulk density (DBD) and peat organic matter content (OM) at the study sites.

The highest surface *K* values were measured in Forested (Table 3.4). The surface *K* values in Drained (mean = 9.2 m day^{-1}, SD = 3.2 m day^{-1}) were generally higher than in Blocked (mean = 6.0 m day^{-1}, SD = 2.7 m day^{-1}). In Drained, the five points with shortest distances to ditches (AA1, AA3, R1, R2, and R7) had lower surface *K* values compared to other sampling points. In Blocked, the three points with shortest distances to the main outlet of the peat plot (BB1, CA3, and CA2) had lower surface *K* values compared to the three points at greater distances (BB2, CA5, and CA6). Infiltrometer tests data indicated that sub-surface *K* values taken at a depth of 20 cm from surface were consistently lower than the surface *K* values (Table 3.4).

In Drained and Forested, the dry bulk density of shallow peat (surface to 50 cm depth) tended to be lower than that of deep peat (Table 3.5). In Forested, the deep peat dry bulk density for AL0 was smaller than for AL1 or AL5, which may be because point AL0 was located on the bed of a small trench (64 cm below the trench's bank). In Blocked, the dry bulk density of shallow peat was similar to that of deep peat (mean of 0.137 g cm^{-3} and 0.131 g cm^{-3} respectively). The organic matter content data were high with very little variability between wells or sites, which ranged between 98.6% and 99.8%. However, for each well, organic matter content was greater in deep peat than in the upper 50 cm, except at point AL0 (at the small trench).

Table 3.4 Near-surface saturated hydraulic conductivity (K) for the studied wells.

Well	Location	Mean distance to nearest ditch (m)	Distance to lowest ditch outlet (m)	K (m day^{-1})	
				Surface	20-cm depth
AL1	Forested	-	-	16.87	-
AL5	Forested	-	-	10.05	-
BB1	Blocked	-	21	4.71	-
CA3	Blocked	-	195	3.12	0.04
CA4	Blocked	-	222	10.10	0.23
CA2	Blocked	-	275	4.21	0.05
CA7	Blocked	-	387	9.84	-
CA1	Blocked	-	390	5.39	1.39
BB2	Blocked	-	424	6.07	-
BB3	Blocked	-	465	2.06	-
CA5	Blocked	-	469	5.07	-
CA6	Blocked	-	612	9.59	-
AA1	Drained	21	-	5.40	-
AA3	Drained	29	-	6.11	-
R5	Drained	112	-	10.70	-
R1	Drained	122	-	9.90	-
R7	Drained	129	-	3.41	-
R2	Drained	136	-	9.27	0.25
R3	Drained	168	-	15.38	0.81
AA2	Drained	173	-	10.52	-
R6	Drained	181	-	10.63	-
R4	Drained	249	-	11.03	-

Table 3.5 Mean dry bulk density (DBD) and organic matter content (OM) of peat at the study sites. Standard deviation values are in brackets. The surface at AL0 (†) was on the bed of a small trench, which was 64 cm below the surrounding peat surface.

Well	Location	Number of samples	Shallow peat		Deep peat	
			DBD (g cm^{-3})	OM (%)	DBD (g cm^{-3})	OM (%)
AA1	Drained	22	0.142[0.05]	98.8	0.128[0.03]	99.1
AA2	Drained	26	0.132[0.03]	99.3	0.175[0.05]	99.8
AA3	Drained	24	0.106[0.07]	99.4	0.142[0.03]	99.8
BB1	Blocked	22	0.140[0.04]	99.0	0.139[0.03]	99.5
BB2	Blocked	24	0.118[0.04]	99.3	0.116[0.03]	99.5
BB3	Blocked	23	0.154[0.05]	99.0	0.137[0.05]	99.6
AL0†	Forested	23	0.093[0.04]	99.2	0.095[0.03]	98.6
AL1	Forested	19	0.093[0.03]	98.6	0.112[0.03]	98.8
AL5	Forested	23	0.055[0.02]	99.1	0.121[0.03]	98.9

3.4 Discussion

3.4.1 Influential storm variables to water-table variations

Section 3.3 examined influential storm variables to the water-table dynamics in tropical peatlands. We provide evidence that the influence of each storm variable varied spatially and seasonally. In Forested, variables associated with the starting phase of the storm (Initial-vars) had less contribution to water-table variation, perhaps because of forest interception in the initial phases of storms. Interception in tropical rainforests can be around 14.5% to 18% of the annual rainfall (Dykes, 1997; Manfroi et al., 2004; Moore et al., 2013). In Blocked, Initial-vars did not strongly contribute to water-table variation during the deep water-table condition because ditch water levels were low and some portions of rainfall would have been lost via seepage to the ditch (hydraulic heads were increasing, see Figure 3.3). In contrast, during shallow water-table conditions in Blocked, the Initial-vars strongly contributed because the drainage effects were minimal. Both Rising-vars and Peak-vars showed significant contribution to water-table variations in Forested and Blocked (Figure 3.2), suggesting those sites had the capacity for a rapid rise of the water table. In Drained, the responses to storms when water tables resided in shallow layers had less contribution from Peak-vars (high rainfall depth) than when water tables resided in peat more than 50 cm deep (see Figure 3.2).

3.4.2 Effect of ditches and ditch dams to the responses to storms

Our findings provide evidence that ditches significantly altered the responses of water table to storms in the Drained compared to the Forested site. Drainage reduced the peatland's capacity to retain water provided by rainfall. We found that in Drained, less rainwater can be stored by shallow peat when the water table is near to the surface (little extra room for rainwater), with excess rainwater draining to the ditches via overland flow. The finding that storm response varied with restoration state is in line with studies of water table and riverflow response in temperate peatlands (Grayson et al., 2010; Holden et al., 2011, 2018; Shuttleworth et al., 2019).

The construction of ditch dams may have been responsible for the responses to storms in Blocked being more like those in Forested than those in Drained. In Blocked, in the wet period, the ditch dams could not fully minimise the hydraulic head (Figure 3.3), given that water was still being drained to the lowest outlet of the peat plot (Putra et al., 2021). To reduce water loss further the water level at the outlet needs to be raised whenever possible, and further restoration measures are needed to supplement the functioning of the ditch dams. Up to the time when this study was conducted (around five years after the ditch dams were built), the rewetting efforts did not result in fully recovered responses to storms in Blocked, assuming that Forested was a close to natural benchmark. Several studies from temperate peatlands have come to similar conclusions, affirming that the recovery of the hydrological dynamics of drained peatlands cannot be achieved in a short period after the start of restoration (Holden et al., 2011; Williamson et al., 2017; Kreyling et al., 2021).

The findings suggest that ditch dams only partially restore the responses to storms in a drained tropical peatland when compared with a near-natural forested peatland. However, we recognise that we studied only one site representing each condition of Drained, Blocked, and Forested, and that further studies of storm response across more sites would be useful. The rainfall and water-table data in this study covered the second half of 2019, which was a typical ENSO (El Niño–Southern Oscillation) year in a neutral condition (WMO, 2019; Becker, 2020). Further work on storm responses to different depth – duration patterns of rainfall during El Niño and La Niña years, may reveal additional components of hydrological functioning that we were unable to ascertain.

3.4.3 Possible effect of peat properties to the storm responses

Our study provided dry bulk density data from tropical peatlands and such data are sparse in the tropical peatland literature. The dry bulk density of the peat in Drained (Table 3.5) is comparable to values reported from another highly degraded and non-forested peatland in Sebangau (Könönen et al., 2015) and at locations near to a canal in Mawas peatland (Block A of MRP, see Sinclair et al. (2020)), but is higher than in the drained peatland in Selangor, Malaysia (Tonks et al., 2017). The dry bulk density of peat in Forested was comparable to that measured in the western side of Sebangau (Lampela et al., 2014), and also in the Amazon basin (Lähteenoja et al., 2013), and Sumatra (Shimamura and Momose, 2005), but lower than values reported for Congo peatlands (Dargie et al., 2017). As far as we are aware, there are no comparable dry bulk density data from tropical peat plots where dams have been installed in the surrounding ditches. We also provided some surface hydraulic conductivity data (circa 2.1–16.9 m day^{-1}) for Indonesian peatland, which may enrich the earlier hydraulic conductivity data (circa 0.001–13.9 m day^{-1}) collected by Kurnianto et al. (2019).

In Blocked, the dry bulk density of shallower peat was comparable to that of deeper peat (mean of 0.137 g cm^{-3} compared to 0.131 g cm^{-3}) but the contribution from Peak-vars (high rainfall depth) differed between the shallow and deeper layers, which may indicate that differences in responses to storms in Blocked were not closely related to dry bulk density. Therefore, we think that the pre-storm water storage in the peat profile, approximately reflected by the initial water-table condition, might be important in determining the response of water tables to rainfall in Blocked. The water-table retention time profiles presented by Putra et al. (2021) and the typical seepage pattern in the area near to ditches with ditch dams presented by Putra et al. (2022) showed that the upstream area of the ditch dam was wetter than around the outlet, which may result in the high surface K and low dry bulk density conditions of peat at the upstream area of the dam.

In Forested, the dry bulk density of shallow peat was comparable to that of deep peat (Table 3.5), yet those values were far smaller than those in Drained and Blocked, suggesting more storm water was potentially stored by the peat layers in Forested than in the other sites. However, this presumption needs to be supported by more data on total porosity and drainable porosity. Putra et al. (2021) reported that the water-table differences among wells in Forested were less than 10 cm, indicating that rapid drawdowns of the water table after storms might be part of natural functioning. The only two measured K values in Forested are similar to the K values at points farther from ditches in Drained (e.g., at AA2 and at R3, see Table 3.4). In accordance with other tropical peatland studies, see Baird et al. (2017) and Kurnianto et al. (2019), it is possible that the K at Forested may actually be higher than the measured K values in Table 3.4. The high K values at Forested could be the reason for the small differences in water tables between wells at Forested. Therefore, we think that subsurface flows in Forested could be greater than in Drained.

3.5 Conclusions

Understanding responses of water tables to storms is important for evaluating the hydrological condition of tropical peatlands. This study showed that responses of water table to storms were different between the studied intact, drained and ditch-dammed tropical peatlands. This chapter also provides evidence that several elements of hydrological functioning were seemingly not restored in the blocked site, at least when compared to the forested site. We also show that the responses to storms are spatially and seasonally variable, meaning that these factors need to be considered in tropical peatland hydrological modelling studies to better represent water-level dynamics rather than using constant seasonal boundary conditions or assuming spatially invariable responses. The simple measure of mean water-table depth may hide more detailed differences in hydrological functioning between sites under different types of management.

3.6 References

Agus, C., Ilfana, Z.R., Azmi, F.F., Rachmanadi, D., Widiyatno, Wulandari, D., Santosa, P.B., Harun, M.K., Yuwati, T.W. and Lestari, T. 2020. The effect of tropical peat land-use changes on plant diversity and soil properties. *International Journal of Environmental Science and Technology*. **17**(3), pp.1703–1712.

Anderson, J.A.R. and Muller, J. 1975. Palynological study of a holocene peat and a miocene coal deposit from NW Borneo. *Review of Palaeobotany and Palynology*. **19**(4), pp.291–351.

Anderson, R. and Peace, A. 2017. Ten-Year results of a comparison of methods for restoring afforested blanket bog. *Mires and Peat*. **19**, pp.1–23.

Baird, A.J. 1997. Field estimation of macropore functioning and surface hydraulic conductivity in a fen peat. *Hydrological Processes*. **11**(3), pp.287–295.

Baird, A.J., Low, R., Young, D., Swindles, G.T., Lopez, O.R. and Page, S.E. 2017. High permeability explains the vulnerability of the carbon store in drained tropical peatlands. *Geophysical Research Letters*. **44**(3), pp.1333–1339.

Becker, E. 2020. February 2020 ENSO update: mind reading. *NOAA ENSO Blog*. [Online]. [Accessed 1 January 2022]. Available from: https://www.climate.gov/news-features/blogs/enso/february-2020-enso-update-mind-reading.

Berninger, A. and Siegert, F. 2020. The Potential of ICESat-2 to Identify Carbon-Rich

Peatlands in Indonesia. *Remote Sensing*. **12**(24), article no.: 4175.

Chambers, F.M., Beilman, D.W. and Yu, Z. 2011. Methods for determining peat humification and for quantifying peat bulk density, organic matter and carbon content for palaeostudies of climate and peatland carbon dynamics. *Mires and Peat*. **7**(7), pp.1–10.

Cobb, A.R. and Harvey, C.F. 2019. Scalar Simulation and Parameterization of Water Table Dynamics in Tropical Peatlands. *Water Resources Research*. **55**(11), pp.9351–9377.

Cobb, A.R., Hoyt, A.M., Gandois, L., Eri, J., Dommain, R., Abu Salim, K., Kai, F.M., Haji Su'ut, N.S. and Harvey, C.F. 2017. How temporal patterns in rainfall determine the geomorphology and carbon fluxes of tropical peatlands. *Proceedings of the National Academy of Sciences*. **114**(26), pp.E5187–E5196.

Dadap, N.C., Hoyt, A.M., Cobb, A.R., Oner, D., Kozinski, M., Fua, P. V., Rao, K., Harvey, C.F. and Konings, A.G. 2021. Drainage Canals in Southeast Asian Peatlands Increase Carbon Emissions. *AGU Advances*. **2**(1), article no.: e2020AV000321.

Dargie, G.C., Lewis, S.L., Lawson, I.T., Mitchard, E.T.A., Page, S.E., Bocko, Y.E. and Ifo, S.A. 2017. Age, extent and carbon storage of the central Congo Basin peatland complex. *Nature*. **542**(7639), pp.86–90.

Davenport, I.J., McNicol, I., Mitchard, E.T.A., Dargie, G., Suspense, I., Milongo, B., Bocko, Y.E., Hawthorne, D., Lawson, I., Baird, A.J., Page, S.E. and Lewis, S.L. 2020. First Evidence of Peat Domes in the Congo Basin using LiDAR from a Fixed-Wing Drone. *Remote Sensing*. **12**(14), article no.: 2196.

Desai, B.H. 2017. United Nations Environment Program (UNEP). *Yearbook of International Environmental Law*. **28**(14), pp.498–505.

Deshmukh, C.S., Julius, D., Desai, A.R., Asyhari, A., Page, S.E., Nardi, N., Susanto, A.P., Nurholis, N., Hendrizal, M., Kurnianto, S., Suardiwerianto, Y., Salam, Y.W., Agus, F., Astiani, D., Sabiham, S., Gauci, V. and Evans, C.D. 2021. Conservation slows down emission increase from a tropical peatland in Indonesia. *Nature Geoscience*. **14**(7), pp.484–490.

Dohong, A., Aziz, A.A. and Dargusch, P. 2017. A review of the drivers of tropical peatland degradation in South-East Asia. *Land Use Policy*. **69**, pp.349–360.

Dommain, R., Couwenberg, J., Glaser, P.H., Joosten, H. and Suryadiputra, I.N.N. 2014. Carbon storage and release in Indonesian peatlands since the last deglaciation.

Quaternary Science Reviews. **97**, pp.1–32.

Dykes, A.P. 1997. Rainfall interception from a lowland tropical rainforest in Brunei. *Journal of Hydrology*. **200**(1–4), pp.260–279.

Eijkelkamp Soil and Water 2020. Peat sampler operating instructions. *Augering & soil sampling equipment*. [Online], pp.1–7. [Accessed 23 March 2022]. Available from: https://en.eijkelkamp.com/products/augering-soil-sampling-equipment/tech-specs-peat-sampler.html.

Farmer, J., Matthews, R., Smith, P. and Smith, J.U. 2014. The Tropical Peatland Plantation-Carbon Assessment Tool: estimating CO_2 emissions from tropical peat soils under plantations. *Mitigation and Adaptation Strategies for Global Change*. **19**(6), pp.863–885.

Grayson, R., Holden, J. and Rose, R. 2010. Long-term change in storm hydrographs in response to peatland vegetation change. *Journal of Hydrology*. **389**(3–4), pp.336–343.

Hapsari, K.A., Biagioni, S., Jennerjahn, T.C., Reimer, P.M., Saad, A., Achnopha, Y., Sabiham, S. and Behling, H. 2017. Environmental dynamics and carbon accumulation rate of a tropical peatland in Central Sumatra, Indonesia. *Quaternary Science Reviews*. **169**, pp.173–187.

Hodgkins, S.B., Richardson, C.J., Dommain, R., Wang, H., Glaser, P.H., Verbeke, B., Winkler, B.R., Cobb, A.R., Rich, V.I., Missilmani, M., Flanagan, N., Ho, M., Hoyt, A.M., Harvey, C.F., Vining, S.R., Hough, M.A., Moore, T.R., Richard, P.J.H., De La Cruz, F.B., Toufaily, J., Hamdan, R., Cooper, W.T. and Chanton, J.P. 2018. Tropical peatland carbon storage linked to global latitudinal trends in peat recalcitrance. *Nature Communications*. **9**, article no.: 3640.

Holden, J., Moody, C.S., Edward Turner, T., McKenzie, R., Baird, A.J., Billett, M.F., Chapman, P.J., Dinsmore, K.J., Grayson, R.P., Andersen, R., Gee, C. and Dooling, G. 2018. Water-level dynamics in natural and artificial pools in blanket peatlands. *Hydrological Processes*. **32**(4), pp.550–561.

Holden, J., Wallage, Z.E., Lane, S.N. and McDonald, A.T. 2011. Water table dynamics in undisturbed, drained and restored blanket peat. *Journal of Hydrology*. **402**(1–2), pp.103–114.

Honorio Coronado, E.N., Hastie, A., Reyna, J., Flores, G., Grández, J., Lähteenoja, O., Draper, F.C., Åkesson, C.M., Baker, T.R., Bhomia, R.K., Cole, L.E.S., Dávila, N., Del Águila, J., Del Águila, M., Del Castillo Torres, D., Lawson, I.T., Martín

Brañas, M., Mitchard, E.T.A., Monteagudo, A., Phillips, O.L., Ramírez, E., Ríos, M., Ríos, S., Rodriguez, L., Roucoux, K.H., Tagle Casapia, X., Vasquez, R., Wheeler, C.E. and Montoya, M. 2021. Intensive field sampling increases the known extent of carbon-rich Amazonian peatland pole forests. *Environmental Research Letters*. **16**(7), article no.: 074048.

Hoogsteen, M.J.J., Lantinga, E.A., Bakker, E.J., Groot, J.C.J. and Tittonell, P.A. 2015. Estimating soil organic carbon through loss on ignition: effects of ignition conditions and structural water loss. *European Journal of Soil Science*. **66**(2), pp.320–328.

Hoyt, A.M., Gandois, L., Eri, J., Kai, F.M., Harvey, C.F. and Cobb, A.R. 2019. CO_2 emissions from an undrained tropical peatland: Interacting influences of temperature, shading and water table depth. *Global Change Biology*. **25**(9), pp.2885–2899.

Husson, S.J., Limin, S.H., Boyd, N.S., Brousseau, J.J., Collier, S., Cheyne, S.M., D'Arcy, L.J., Dow, R.A. and Schreven, S. 2018. Biodiversity of the Sebangau tropical peat swamp forest, Indonesian Borneo. *Mires and Peat*. **22**, pp.1–50.

International Climate Initiative 2021. The Global Peatlands Initiative: Assessing, Measuring and Preserving Peat Carbon. *Food and Agriculture Organization of the United Nations*. [Online]. [Accessed 1 February 2021]. Available from: https://www.international-climate-initiative.com/en/details/project/the-global-peatlands-initiative-assessing-measuring-and-preserving-peat-carbon-18_III_096-3039.

IPCC 2021. Summary for Policymakers *In*: V. Masson-Delmotte, P. Zhai, A. Pirani, S. L. Connors, C. Péan, S. Berger, N. Caud, Y. Chen, L. Goldfarb, M. I. Gomis, M. Huang, K. Leitzell, E. Lonnoy, J. B. R. Matthews, T. K. Maycock, T. Waterfield, O. Yelekçi, R. Yu and B. Zhou, eds. *Climate Change 2021: The Physical Science Basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change*. [Online]. Cambridge, UK: Cambridge University Press. In Press., pp.1–40. Available from: https://www.ipcc.ch/report/ar6/wg1/.

Ishii, Y., Koizumi, K., Fukami, H., Yamamoto, K., Takahashi, H., Limin, S.H., Kusin, K., Usup, A. and Susilo, G.E. 2016. Groundwater in Peatland *In*: M. Osaki and N. Tsuji, eds. *Tropical Peatland Ecosystems*. Tokyo: Springer Japan, pp.265–279.

Jolliffe, I.T. and Cadima, J. 2016. Principal component analysis: a review and recent developments. *Philosophical Transactions of the Royal Society A: Mathematical, Physical and Engineering Sciences*. **374**(2065), article no.: 20150202.

Kasih, R.C., Simon, O., Ansori, M., Pratama, M.P. and Wirada, F. 2016. Rewetting on Degraded Tropical Peatland by Canal Blocking Technique in Sebangau National Park, Central Kalimantan, Indonesia *In*: *Proceedings on 15th International Peat Congress 2016. Extended Abstract No: A-065*.

Kassambara, A. 2017. *Practical guide to principal component methods in R: PCA, M (CA), FAMD, MFA, HCPC, factoextra* 1st ed. Sthda.

Kelly, T.J., Lawson, I.T., Roucoux, K.H., Baker, T.R. and Honorio Coronado, E.N. 2020. Patterns and drivers of development in a west Amazonian peatland during the late Holocene. *Quaternary Science Reviews*. **230**, article no.: 106168.

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

Kreyling, J., Tanneberger, F., Jansen, F., van der Linden, S., Aggenbach, C., Blüml, V., Couwenberg, J., Emsens, W.-J., Joosten, H., Klimkowska, A., Kotowski, W., Kozub, L., Lennartz, B., Liczner, Y., Liu, H., Michaelis, D., Oehmke, C., Parakenings, K., Pleyl, E., Poyda, A., Raabe, S., Röhl, M., Rücker, K., Schneider, A., Schrautzer, J., Schröder, C., Schug, F., Seeber, E., Thiel, F., Thiele, S., Tiemeyer, B., Timmermann, T., Urich, T., van Diggelen, R., Vegelin, K., Verbruggen, E., Wilmking, M., Wrage-Mönnig, N., Wołejko, L., Zak, D. and Jurasinski, G. 2021. Rewetting does not return drained fen peatlands to their old selves. *Nature Communications*. **12**, article no.: 5693.

Kurnianto, S., Selker, J., Boone Kauffman, J., Murdiyarso, D. and Peterson, J.T. 2019. The influence of land-cover changes on the variability of saturated hydraulic conductivity in tropical peatlands. *Mitigation and Adaptation Strategies for Global Change*. **24**(4), pp.535–555.

Lähteenoja, O., Flores, B. and Nelson, B. 2013. Tropical Peat Accumulation in Central Amazonia. *Wetlands*. **33**(3), pp.495–503.

Lai, W.Y. and Kuok, K.K. 2019. A Study on Bayesian Principal Component Analysis for Addressing Missing Rainfall Data. *Water Resources Management*. **33**(8), pp.2615–2628.

Lampela, M., Jauhiainen, J. and Vasander, H. 2014. Surface peat structure and chemistry in a tropical peat swamp forest. *Plant and Soil*. **382**(1–2), pp.329–347.

Lê, S., Josse, J. and Husson, F. 2008. FactoMineR: An R Package for Multivariate Analysis. *Journal of Statistical Software*. **25**(1), pp.1–18.

Li, W., Dickinson, R.E., Fu, R., Niu, G.-Y., Yang, Z.-L. and Canadell, J.G. 2007. Future precipitation changes and their implications for tropical peatlands. *Geophysical Research Letters*. **34**(1), article no.: L01403.

Lilleskov, E., McCullough, K., Hergoualc'h, K., del Castillo Torres, D., Chimner, R., Murdiyarso, D., Kolka, R., Bourgeau-Chavez, L., Hribljan, J., del Aguila Pasquel, J. and Wayson, C. 2019. Is Indonesian peatland loss a cautionary tale for Peru? A two-country comparison of the magnitude and causes of tropical peatland degradation. *Mitigation and Adaptation Strategies for Global Change*. **24**(4), pp.591–623.

Maity, R. 2018. *Statistical methods in hydrology and hydroclimatology* (T. G. Sitharam, ed.). Springer Singapore.

Manfroi, O.J., Koichiro, K., Nobuaki, T., Masakazu, S., Nakagawa, M., Nakashizuka, T. and Chong, L. 2004. The stemflow of trees in a Bornean lowland tropical forest. *Hydrological Processes*. **18**(13), pp.2455–2474.

Marwanto, S., Watanabe, T., Iskandar, W., Sabiham, S. and Funakawa, S. 2018. Effects of seasonal rainfall and water table movement on the soil solution composition of tropical peatland. *Soil Science and Plant Nutrition*. **64**(3), pp.386–395.

Meter Group 2020. *Minidisk Infiltrometer Manual*. [Online]. Pullman, WA. Available from: http://publications.metergroup.com/Manuals/20421_Mini_Disk_Manual_Web.pdf.

Mezbahuddin, M., Grant, R.F. and Hirano, T. 2015. How hydrology determines seasonal and interannual variations in water table depth, surface energy exchange, and water stress in a tropical peatland: Modeling versus measurements. *Journal of Geophysical Research: Biogeosciences*. **120**(11), pp.2132–2157.

Mezbahuddin, M., Grant, R.F. and Hirano, T. 2014. Modelling effects of seasonal variation in water table depth on net ecosystem CO_2 exchange of a tropical peatland. *Biogeosciences*. **11**(3), pp.577–599.

Miettinen, J., Hooijer, A., Vernimmen, R., Liew, S.C. and Page, S.E. 2017. From carbon sink to carbon source: extensive peat oxidation in insular Southeast Asia since 1990. *Environmental Research Letters*. **12**(2), article no.: 024014.

Moore, S., Evans, C.D., Page, S.E., Garnett, M.H., Jones, T.G., Freeman, C., Hooijer, A., Wiltshire, A.J., Limin, S.H. and Gauci, V. 2013. Deep instability of deforested tropical peatlands revealed by fluvial organic carbon fluxes. *Nature*. **493**(7434), pp.660–663.

Page, S.E., Wűst, R.A.J., Weiss, D., Rieley, J.O., Shotyk, W. and Limin, S.H. 2004. 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. *Journal of Quaternary Science*. **19**(7), pp.625–635.

Patterson, G. and Anderson, R. 2000. *Forests and Peatland Habitats: Guideline Note*. [Online]. Edinbrugh, UK. Available from: https://forestry.gov.scot/publications/2-forests-and-peatland-habitats.

Phillips, S., Rouse, G.E. and Bustin, R.M. 1997. Vegetation zones and diagnostic pollen profiles of a coastal peat swamp, Bocas del Toro, Panamá. *Palaeogeography, Palaeoclimatology, Palaeoecology*. **128**(1–4), pp.301–338.

Putra, E.I., Cochrane, M.A., Vetrita, Y., Graham, L. and Saharjo, B.H. 2018. Determining critical groundwater level to prevent degraded peatland from severe peat fire. *IOP Conference Series: Earth and Environmental Science*. **149**, article no.: 012027.

Putra, S.S., Baird, A.J. and Holden, J. 2022. Modelling the performance of bunds and ditch dams in the hydrological restoration of tropical peatlands. *Hydrological Processes*. **36**(1), article no.: e14470.

Putra, S.S., Holden, J. and Baird, A.J. 2021. The effects of ditch dams on water-level dynamics in tropical peatlands. *Hydrological Processes*. **35**(5), article no.: e14174.

Reynolds, W.D. and Elrick, D.E. 1991. Determination of hydraulic conductivity using a tension infiltrometer. *Soil Science Society of America Journal*. **55**(3), pp.633–639.

Ritzema, H., Limin, S., Kusin, K., Jauhiainen, J. and Wösten, H. 2014. Canal blocking strategies for hydrological restoration of degraded tropical peatlands in Central Kalimantan, Indonesia. *CATENA*. **114**, pp.11–20.

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. and Vriesendorp, C.F. 2017. Threats to intact tropical peatlands and opportunities for their conservation. *Conservation Biology*. **31**(6), pp.1283–1292.

Ruwaimana, M., Anshari, G.Z., Silva, L.C.R. and Gavin, D.G. 2020. The oldest extant tropical peatland in the world: a major carbon reservoir for at least 47 000 years. *Environmental Research Letters*. **15**(11), article no.: 114027.

Schulz, C., Martín Brañas, M., Núñez Pérez, C., Del Aguila Villacorta, M., Laurie, N., Lawson, I.T. and Roucoux, K.H. 2019. Peatland and wetland ecosystems in Peruvian Amazonia: indigenous classifications and perspectives. *Ecology and Society*. **24**(2), article no.: 12.

Shimamura, T. and Momose, K. 2005. Organic matter dynamics control plant species coexistence in a tropical peat swamp forest. *Proceedings of the Royal Society B: Biological Sciences*. **272**(1571), pp.1503–1510.

Shuttleworth, E.L., Evans, M.G., Pilkington, M., Spencer, T., Walker, J., Milledge, D. and Allott, T.E.H. 2019. Restoration of blanket peat moorland delays stormflow from hillslopes and reduces peak discharge. *Journal of Hydrology X*. **2**, article no.: 100006.

Sinclair, A.L., Graham, L.L.B., Putra, E.I., Saharjo, B.H., Applegate, G., Grover, S.P. and Cochrane, M.A. 2020. Effects of distance from canal and degradation history on peat bulk density in a degraded tropical peatland. *Science of The Total Environment*. **699**, article no.: 134199.

Sutikno, S., Rinaldi, R., Saputra, E., Kusairi, M., Saharjo, B.H. and Putra, E.I. 2019. Water management for hydrological restoration and fire prevention in tropical peatland *In*: T. B. Aulia, ed. *The 2nd Aceh International Symposium on Civil Engineering (AISCE) ,18-19 September 2019*. Banda Aceh, Indonesia: IOP Conference Series: Materials Science and Engineering, article no.: 012053.

Tonks, A.J., Aplin, P., Beriro, D.J., Cooper, H., Evers, S., Vane, C.H. and Sjögersten, S. 2017. Impacts of conversion of tropical peat swamp forest to oil palm plantation on peat organic chemistry, physical properties and carbon stocks. *Geoderma*. **289**, pp.36–45.

Vernimmen, R., Hooijer, A., Mulyadi, D., Setiawan, I., Pronk, M. and Yuherdha, A.T. 2020. A New Method for Rapid Measurement of Canal Water Table Depth Using Airborne LiDAR, with Application to Drained Peatlands in Indonesia. *Water*. **12**(5), article no.: 1486.

Warren, M.W., Kauffman, J.B., Murdiyarso, D., Anshari, G., Hergoualc'h, K., Kurnianto, S., Purbopuspito, J., Gusmayanti, E., Afifudin, M., Rahajoe, J., Alhamd, L., Limin, S. and Iswandi, A. 2012. A cost-efficient method to assess carbon stocks in tropical peat soil. *Biogeosciences*. **9**(11), pp.4477–4485.

Westra, S., Brown, C., Lall, U. and Sharma, A. 2007. Modeling multivariable hydrological series: Principal component analysis or independent component analysis? *Water Resources Research*. **43**(6), article no.: W06429.

Wijedasa, L.S., Vernimmen, R., Page, S.E., Mulyadi, D., Bahri, S., Randi, A., Evans, T.A., Lasmito, Priatna, D., Jensen, R.M. and Hooijer, A. 2020. Distance to forest, mammal and bird dispersal drive natural regeneration on degraded tropical peatland. *Forest Ecology and Management*. **461**, article no.: 117868.

Williamson, J., Rowe, E., Reed, D., Ruffino, L., Jones, P., Dolan, R., Buckingham, H., Norris, D., Astbury, S. and Evans, C.D. 2017. Historical peat loss explains limited short-term response of drained blanket bogs to rewetting. *Journal of Environmental Management*. **188**, pp.278–286.

Winston, R.B. 1994. Models of the geomorphology, hydrology, and development of domed peat bodies. *Geological Society of America Bulletin*. **106**(12), pp.1594–1604.

WMO 2019. *The WMO El Niño/La Niña Update 29 November 2019*. [Online]. Geneva, Switzerland, Switzerland. Available from: https://community.wmo.int/activity-areas/climate/wmo-el-ninola-nina-updates#archive.

Wösten, H., Van Den Berg, J., Van Eijk, P., Gevers, G.J.M., Giesen, W.B.J.T., Hooijer, A., Idris, A., Leenman, P.H., Rais, D.S., Siderius, C., Silvius, M.J., Suryadiputra, N. and Wibisono, I.T. 2006. Interrelationships between Hydrology and Ecology in Fire Degraded Tropical Peat Swamp Forests. *International Journal of Water Resources Development*. **22**(1), pp.157–174.

Wösten, H., Clymans, E., Page, S.E., Rieley, J.O. and Limin, S.H. 2008. Peat–water interrelationships in a tropical peatland ecosystem in Southeast Asia. *CATENA*. **73**(2), pp.212–224.

Wösten, H., Hooijer, A., Siderius, C., Rais, D.S., Idris, A. and Rieley, J. 2006. Tropical Peatland water management modelling of the Air Hitam Laut catchment in Indonesia. *International Journal of River Basin Management*. **4**(4), pp.233–244.

Xu, J., Morris, P.J., Liu, J. and Holden, J. 2018. PEATMAP: Refining estimates of global peatland distribution based on a meta-analysis. *CATENA*. **160**, pp.134–140.

Young, D.M., Baird, A.J., Charman, D.J., Evans, C.D., Gallego-Sala, A. V., Gill, P.J., Hughes, P.D.M., Morris, P.J. and Swindles, G.T. 2019. Misinterpreting carbon accumulation rates in records from near-surface peat. *Scientific Reports*. **9**, article no.: 17939.

Yu, Z., Loisel, J., Brosseau, D.P., Beilman, D.W. and Hunt, S.J. 2010. Global peatland dynamics since the Last Glacial Maximum. *Geophysical Research Letters*. **37**(13), article no.: L13402.

Chapter IV

Modelling the performance of bunds and ditch dams in the hydrological restoration of tropical peatlands

Abstract

Many tropical peatlands are subjected to artificial drainage that leads to degradation. Hence, hydrological restoration has recently been prioritized. Nevertheless, as field monitoring data are limited, little is known about how restoration measures, such as ditch dams and bunds, can regulate tropical peatland water tables. We used a hydrodynamic model —DigiBog_Hydro— to simulate the effectiveness of ditch dams and bunds across three El Niño–Southern Oscillation (ENSO) scenarios, which are El Niño, La Niña and Neutral, in three typical sites. The sites were moderately degraded (Mod-Dgr) and severely degraded (Sev-Dgr) peatland plots (each 0.2 km^2), representing typical peatland conditions in Sebangau National Park, Kalimantan, Indonesia. Our fine-scale (1 m × 1 m spatial resolution) modelling revealed that in the dry season of any ENSO scenario, the significant effects of ditch-dams alone on peatland water-level were limited to lateral distances of 26 m (in Mod-Dgr) and 12 m (in Sev-Dgr) from the ditch. In the dry season of an El Niño year, the combination of ditch dams and bunds helped maintain water levels up to 72 cm (in Mod-Dgr) and 69 cm (in Sev-Dgr) higher than in the no-restoration condition. During the extreme-dry period of an El Niño year, the bunds reduced the number of days when the water table was deeper than 40 cm in Mod-Dgr and in Sev-Dgr by 50% and 73%, respectively. We suggest that bunds used in combination with ditch dams are a practical restoration measure for tropical peatlands, providing critical extra water storage and helping maintain water tables near the peatland surface in dry periods. We also demonstrate how fine-scale hydrodynamic modelling is beneficial for planning and assessment of restoration measures in tropical peatlands.

4.1 Introduction

Tropical peatlands are important globally as carbon stores and for hosting distinctive and biodiverse ecosystems (Page and Baird, 2016; Hapsari et al., 2017; Agus et al., 2019; Wijedasa et al., 2020; Girkin et al., 2020). Xu et al. (2018) estimated that tropical peatlands cover at least 3.38×10^5 km^2, representing 8% of global peatland coverage. The main tropical peatland areas are found in Southeast Asia (2.48×10^5 km^2), Peruvian Amazonia (0.22×10^5 km^2) and the Congo Basin (1.45×10^5 km^2), accounting for an estimated carbon storage of 68.5 Pg C, 3.14 Pg C and 30.6 Pg C, respectively (Page et al., 2011; Draper et al., 2014; Dargie et al., 2017; Warren et al., 2017; Honorio Coronado et al., 2021).

Since the 1980s, many tropical peatlands have been converted to agricultural land and plantation forestry, and these conversions have been associated with widespread drainage (Farmer et al., 2014; Graham et al., 2017; Dohong, Aziz, et al., 2017; Könönen et al., 2018). As an example, in 1995, the Indonesian Government implemented the Mega Rice Project (MRP) that resulted in the drainage of 3 million hectares of tropical peatland before the scheme was stopped in 1999 (Dohong, Aziz, et al., 2017; Dohong et al., 2018b). Drainage involved the construction of canals and ditches to lower water tables (Boehm and Siegert, 2001; Limin et al., 2007; Medrilzam et al., 2017; Ward et al., 2021). Canals are up to 25 m wide, 4.5 m deep, can extend for tens of km, and receive drainage water from neighbouring ditch networks (Page et al., 2009; Sinclair et al., 2020). Secondary canals divided peatland into compartments of roughly 6.25 km^2 each (Mawdsley et al., 2009; Blackham et al., 2014), in particular those in Mantangai (Block A of the Ex-MRP). Meanwhile, ditches tend to be between 2–4 m wide and 1.5–3 m deep, generally designed in a grid pattern, bounding small peat plots typically of 0.15–0.25 km^2. The depth of canals and ditches may vary over time due to subsidence.

Drainage causes rapid decay (oxidative loss) and an increase in the bulk density of peat, both leading to subsidence (Hooijer et al., 2012; Carlson et al., 2015; Sinclair et al., 2020). Couwenberg et al. (2010) suggested that each additional water-table lowering of 10 cm will trigger an approximate 0.25×10^{-6} Pg of C loss per year per km^2 of tropical peatland, and 0.9 cm of annual peat subsidence. Subsidence can be extreme, as in an abandoned area of the Ex-MRP, the peat surface subsided by 25 cm between 9 July 2007 and 1 September 2010 (Zhou et al., 2019; Hoyt et al., 2020). Drained peatlands are also susceptible to fire, and fire is used to clear natural forests as part of land conversions (Roucoux et al., 2017; Cooper et al., 2019; Dadap et al., 2019). Studies of fire distribution in Peninsular Malaysia, Sumatra and Borneo in 2015 showed that there was less fire in areas without artificial drainage than in drained areas (Miettinen et al., 2017). Peatland fires can last many weeks and lead to extremely high rates of C loss (as CO_2 and CH_4) (Page et al., 2002; Ballhorn et al., 2009). Accordingly, predictions suggest that drainage and peatland conversions for agriculture in Southeast Asia will cause CO_2 emissions of 4.43–11.45 Pg between 2010 and 2130 if restoration is not carried out (Hergoualc'h and Verchot, 2011; Roucoux et al., 2017; Wijedasa et al., 2018).

Because of the environmental damage caused by drainage, there have been recent government initiatives to restore or rehabilitate tropical peatlands. These initiatives formed a part of the 'Brazzaville Declaration', 21–23 March 2018, attended by representatives of Democratic Republic of the Congo, Republic of the Congo, Republic of Peru and Republic of Indonesia (Desai, 2017; International Climate Initiative, 2021). The Indonesian Government, for example, aims to maintain peatland water tables at depths shallower than 40 cm from the surface (the 40-cm limit) (The Regulation of The Republic of Indonesia No. 57 Year 2016 about Peatland Ecosystem Protection and Management, 2016). Measures being implemented to keep within this limit include restoring native peatland vegetation, banning new peatland drainage operations, installing canal dams and ditch dams and infilling of ditches (Dohong, Cassiophea, et al., 2017; Dohong et al., 2018a; Harrison et al., 2020). Little is known as to whether these restoration measures, specifically ditch damming, are sufficient to ensure peatland water tables stay within the 40-cm limit. However, a study by Putra et al. (2021) in Sebangau peatland, Kalimantan, indicated that dams alone could not keep water tables within the 40-cm limit, especially during the dry season. In summary, Putra et al. (2021) provided empirical evidence that the ditch dams do not maintain the ditch and peatland water levels in the dry period, because not enough water was retained in the peatland at the beginning of the dry season. Therefore, if the 40 cm water-table policy is to be achieved, extra water needs to be retained in the peatland at the end of a wet period, to act as buffer to maintain the water table of the peatland during subsequent dry periods.

Based on some initial studies in temperate peatlands, some researchers have proposed supplementing dams with bunds to keep a peatland wet (Shantz and Price, 2006; Land and Brock, 2017; Payne et al., 2018; Glenk et al., 2020). Bunds tend to be impermeable or very low permeability linear barriers installed on the peatland surface (not in canals or ditches). They are designed to store rainfall or surface water in the area behind the bunds. In temperate peatland studies, researchers have promoted two types of bund: i) cell design (for relatively flat peatland applications) and ii) contour bunds (for sloping peatland applications). The cell bunds might be suitable for application in Sebangau peatland, given the relatively flat terrain of the peatland.

Before bunds can be promoted and used more widely in tropical settings, it is important to appraise their effectiveness. However, undertaking field trials can be costly and time-consuming (Ritzema et al., 2014; Kasih et al., 2016; Novitasari et al., 2018). An alternative approach is to use a physically based hydrodynamic model to simulate the effect of bunds in combination with other restoration measures. Models based on the Boussinesq groundwater equation have been applied to simulate water-table response to peatland drainage and variations in meteorological conditions (e.g. see Baird et al. (2017)), and also to model restoration scenarios (e.g. see Urzainki et al. (2020)). In this study, we evaluated the combined and separate effects of dams and bunds on water tables in a typical Indonesian tropical peatland under different climate scenarios. The climate scenarios covered three El Niño–Southern Oscillation (ENSO) conditions, which are ENSO neutral (medium rainfall), El Niño (limited rainfall) and La Niña (abundant rainfall) conditions (more detailed explanation is available in the methods section). We focused on two research questions: how do different ditch dam and bunding arrangements affect seasonal and spatial water-level dynamics in tropical peatlands and how does the degree to which the peat has been degraded influence the effectiveness of restoration measures? Furthermore, we want to appraise the use of DigiBog_Hydro in modelling water-table restoration in tropical peatlands.

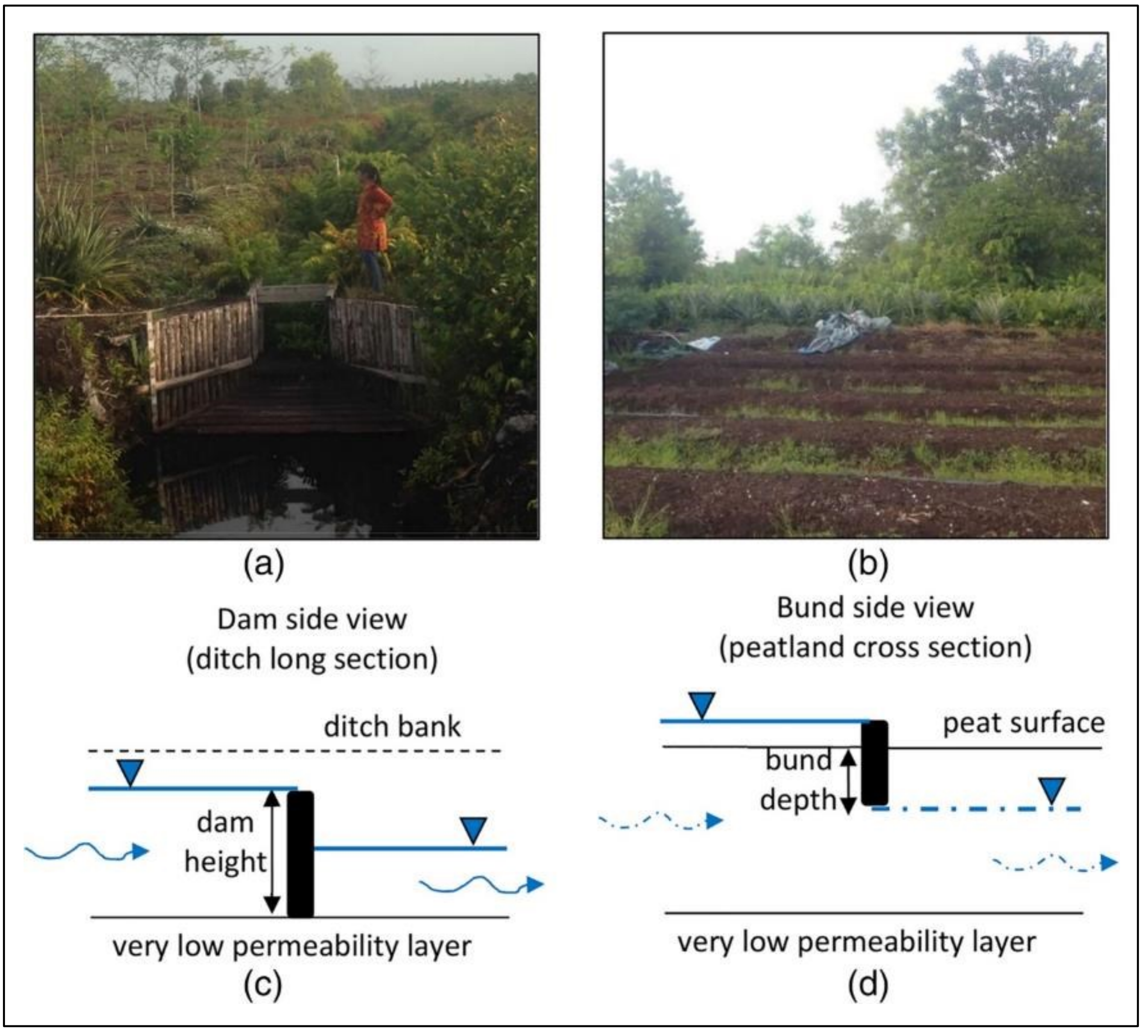

Figure 4.1 The ditch dam and agricultural bunds in the Sebangau area, Indonesia. The ditch dam (a) has been used to restore peatland water level in the studied area. Agricultural bunds (b) are commonly created in the area, but water-level restoration bunds have not been trialled (see Payne et al. (2018) and Wichmann et al. (2017) for the application of restoration bunds in temperate peatlands). The sectional water-level schematization in (c) is for a ditch dam and the one in (d) is for a restoration bund. The cross-section scheme for an agricultural bund is not shown. The lines with triangles represent water levels, in which the light blue lines are ditch/surface water levels, and the dash-dotted lines are the sub-surface water level. Wavy arrows show water flow direction.

4.2 Data and methods

4.2.1 Typical drained sites

We simulated drained peatlands, typical of those found in Sebangau, Kalimantan, Indonesia (see Putra et al. (2021)), using a hydrodynamic model —DigiBog_Hydro (Baird et al., 2012). In Sebangau, drained peatland areas are commonly divided into plots by a grid of ditches. There are typically two to four ditch dams installed for a peat plot in restored areas, but no ditch dams in other drained plots. In Sebangau, ditches are 2–3 m wide and 2–4 m depth (although ditch depths can change over time due to peat surface subsidence following drainage, see Hooijer et al. (2012)). The thickness of the ditch dams installed in ditches is approximately 1–2 m. The main body of ditch dams is designed as a single block of sand or compacted peat enclosed by plastics, whereas the outer shell of the ditch dam is created with a layer of wood slabs or poles (Suryadiputra et al., 2005; Dohong et al., 2018a). The ditch dam has wings, which are extensions of the main body that are about 2–3 m long, anchored sideways into the ditch banks. The ditch dams aim to be water deflectors (Dohong, Cassiophea, et al., 2017; Dohong et al., 2018a), diverting some water in ditches to the peatland, slowing channel flow. Restoration bunds have not been trialled yet in the Sebangau area.

In temperate peatlands, restoration bunds are sometimes built to enhance surface water storage, to help 'buffer' water tables during dry periods. These restoration bunds are made of poorly permeable material (plastic, clay or compacted peat), arranged in such a way to allow 30–50 cm of temporary surface inundation (Price et al., 2003; Wichmann et al., 2017; Payne et al., 2018). In the Sebangau area, local farmers create ridges to produce an elevated surface above a peatland so that crops can be planted on it, but not in a configuration to trap water on site (Figure 4.1). Thus, bunding for restoration might be acceptable to local communities as ridge structures are familiar. Here, we modelled a bund system that seeks to reduce drying of the peatland during the early phase of a dry season by retaining surface water on the peatland.

4.2.2 Modelling scenarios

In this study, the scenarios that were modelled included several components: degree of peatland degradation, climate, presence or absence of ditch dams and bunds, and bund depth (Figure 4.2). First, two peatland degradation conditions were modelled: moderately degraded (Mod-Dgr, assumed to be a drained peatland with a dense vegetation cover and with no fire record in the last 20 years) and severely degraded (Sev-Dgr, interpreted as a drained ex-agricultural peatland that has been burnt and that has bare peat or a sparse vegetation cover). Second, three climate scenarios were implemented: El Niño, Neutral and La Niña. Third, four drainage and restoration scenarios were used: a drained peatland restored with dams and bunds (Combined), a peatland restored with bunds but without dams (Bunded), a peatland restored with dams but without bunds (Dammed) and finally a peatland without dams and without bunds (Control). These scenarios were chosen to compare restoration strategies with each other and with the unrestored control condition. Fourth, in the Bunded scenario, two bund types were considered: On-Surface (bund depth 0 cm) and Extended (bund depth 50 cm), in which the bund depth values are the extent to which the bund penetrates below the peat surface. The height of the bund above ground in this modelling study was set to 30 cm.

In total, 32 model setups were used to represent all scenarios across the targeted modelling variables (see Figure 4.2), which were four dam/bund arrangements, two peat types, and three ENSO conditions, plus eight scenarios of bund type variation ($[4 \times 2 \times 3] + 8 = 32$ setups). For the model simulations, we assumed a rectangular tropical peat plot of 500 m × 400 m bounded by ditches. The selected typical peat plot dimensions mimicked the conditions of a drained peatland with ditch dams studied by Putra et al. (2021).

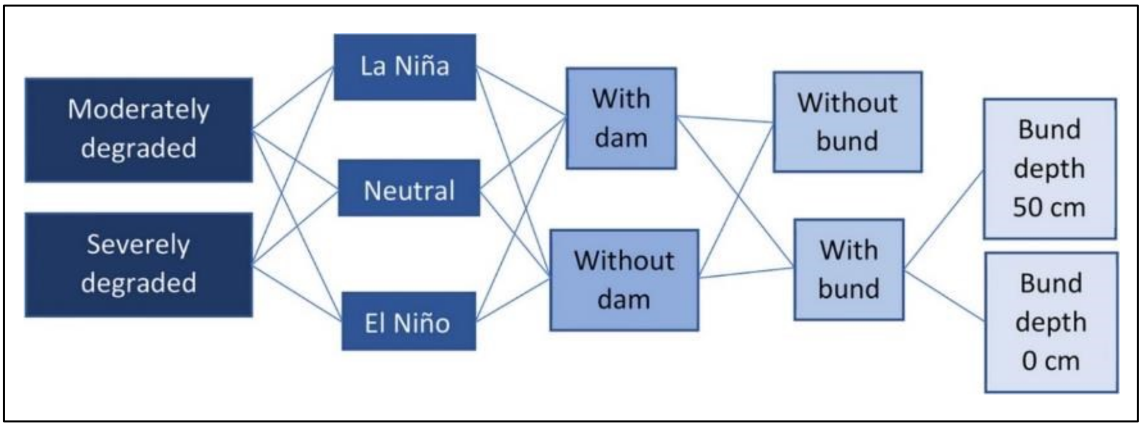

Figure 4.2 Schematic diagram of modelling scenarios.

Weather data for the model runs were obtained from the BMKG (Indonesian Meteorology, Climatology and Geophysics Agency) weather station in Palangka Raya City (2.2279°S, 113.9462°E, 10 m.a.s.l.) —WMO Weather Station ID: 96655, located 13 km from the northernmost tributary of the main canal in Block C MRP (near Kalampangan village). The weather station has long rainfall records (1978–2021) for the Sebangau area. The total daily rainfall records were collected from a ground-sited rain gauge. Three meteorological scenarios were chosen based on inter-annual ENSO variations: a La Niña year (1 February 2011 to 8 February 2012), a 'Neutral' year (1 February 2013 to 8 February 2014), and an El Niño year (1 February 2015 to 8 February 2016) (Figure 4.3). The rainfall totals for those years were 3594 mm (La Niña), 2844 mm (Neutral) and 2778 mm (El Niño). The particular years were selected based on recent meteorological studies and reports (WMO, 2012, 2014a, 2014b, 2016; Susilo, Yamamoto, Imai, Inoue, et al., 2013; Supari et al., 2018), but also considering the BMKG rainfall data availability and reliability. Years in the BMKG database with greater than 5% of days with no data during the associated wet period (November to April), or a total of yearly rainfall higher than 4000 mm, were not used in this study.

The weather data inputted to DigiBog_Hydro are in the form of daily net rainfall (rainfall minus evapotranspiration). Hirano et al. (2015) determined that the values of yearly evapotranspiration (ET) in tropical peatland were 1374 ± 75 mm in a Sev-Dgr site (site with peat fire records) and 1636 ± 53 mm in a Mod-Dgr site (less disturbed site). Hirano et al. (2015) also suggested that daily ET values were in the range 3.27–3.35 mm in a Sev-Dgr site, but 4.09–4.60 mm in a Mod-Dgr site. Therefore, in this study, it is assumed that there are differences in ET between typical moderately and severely degraded sites under the implementation of hydrological restoration management. However, given we had no actual daily ET measurements for the studied periods, the daily ET data of each typical site were assumed to be uniform for the whole year and across ENSO scenarios (Figure 4.3). The chosen daily ET values were 3.76 mm for Sev-Dgr and 4.48 mm for Mod-Dgr. In Sev-Dgr, the calculated yearly net rainfalls were 2190 mm (La Niña), 1440 mm (Neutral) and 1373 mm (El Niño). Accordingly, in Mod-Dgr, the calculated yearly net rainfalls were 1922 mm (La Niña), 1172 mm (Neutral) and 1106 mm (El Niño).

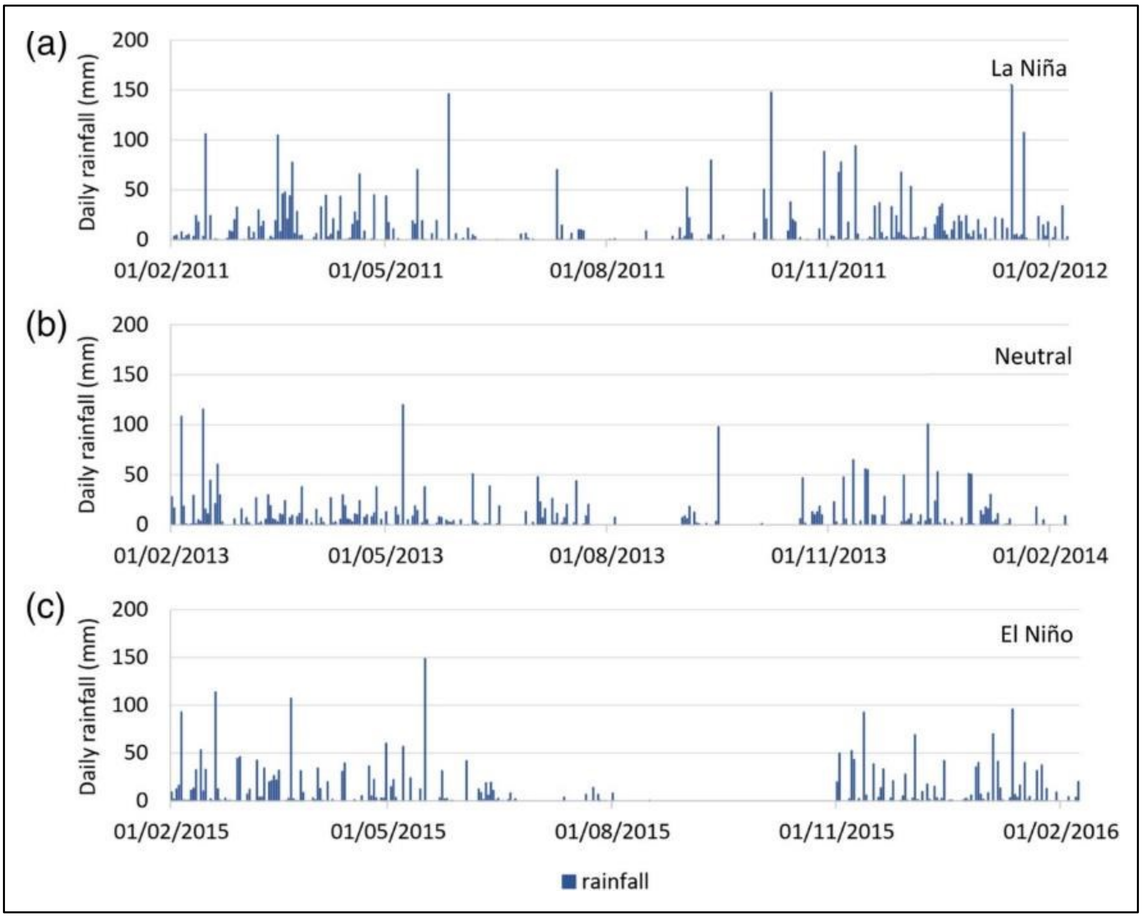

Figure 4.3 The time series of daily rainfall of a La Niña, a Neutral and an El Niño year that are used in the modelling scenarios.

The model was implemented with different peat properties across layers and peat degradation conditions (Table 4.1). Generally, there were four different layers set above the model base (a very low permeability layer), but five layers for the area behind the bund. The additional uppermost layer (the fifth layer) was used to simulate surface inundation on the area behind the bunds, as described in the DigiBog_Hydro User Manual (Baird et al., 2020). The layers varied in thickness and the peat properties were assumed to be homogeneous within a layer (Table 4.1). Some of the peat properties values were adopted from the literature (Kobayashi, 2016; Kurnianto et al., 2019) and others were estimated based on the assumption that hydraulic conductivity decreases with depth (Kelly et al., 2014; Baird et al., 2017; Cobb et al., 2017). Drainable porosity has not been measured in many tropical peatlands. Cobb and Harvey (2019) provided a drainable porosity profile for a pristine tropical peatland, based on the response of the water table to rainfall, which is similar to the Mod-Dgr peatland profile in Table 4.1. Wösten et al. (2008) estimated drainable porosity values from measurements, which showed 50% drainage of total pore spaces in the top peat layer with a drop of peatland water table by 40 cm from the surface. The same estimation was used by Mezbahuddin et al. (2015). Baird et al. (2017) set drainable porosity values at 0.6 (upper), 0.45 (middle) and 0.35 (lower) for each peat layer based upon expert judgement but their values were not based on field measurements. We chose drainable porosity values based on the values reported in the above studies. In scenarios involving bunds (Combined and Bunded), the additional layer was set to have high drainable porosity and high hydraulic conductivity to represent surface flow through dense vegetation.

Table 4.1 Peat properties used in the DigiBog_Hydro model scenarios. *K* is hydraulic conductivity and *s* is drainable porosity. There are bund depth 50 cm and 0 cm (on surface only) scenarios. A part of the presented *K* data († & ‡) are median values taken from Kurnianto et al. (2019) (†) and Kobayashi (2016) (‡). The other *K* data (§) were estimated, considering that *K* decreases with depth (Kelly et al., 2014; Baird et al., 2017; Cobb et al., 2017). The *s* data were chosen based on values presented in previous studies (Wösten et al., 2008; Mezbahuddin et al., 2015; Baird et al., 2017).

Layer depth	**Severely degraded site**		**Moderately degraded site**		**Bund-d-50cm**		**Bund-d-0cm**	
(cm)	***K* (cm s^{-1})**	***s***	***K* (cm s^{-1})**	***s***	***K* (cm s^{-1})**	***s***	***K* (cm s^{-1})**	***s***
-30 to 0	5	0.90	3	0.9	1×10^{-6}	0.11	1×10^{-6}	0.11
0 to 20§	2.4769×10^{-3} §	0.45	1.1631×10^{-2} §	0.6	1×10^{-6}	0.11	-	-
20 to 50	6.2847×10^{-4} §	0.42	4.3287×10^{-3} ‡	0.55	1×10^{-6}	0.11	-	-
50 to 80†	1.0995×10^{-4} †	0.37	2.3148×10^{-4} †	0.45	-	-	-	-
80 to 200†	3.4722×10^{-5} †	0.30	4.9769×10^{-5} †	0.3	-	-	-	-

4.2.3 DigiBog_Hydro model

The DigiBog_Hydro model simulates subsurface flow and water-table dynamics in the *x–y* plane. It also allows for variation in hydraulic conductivity and drainable porosity laterally and with depth below the surface. Therefore, despite not simulating vertical water flow, it can be described as a 2.5-dimensional model. It is based on a numerical solution to the following version of the Boussinesq equation (Baird et al., 2012):

$$\frac{\partial h}{\partial t} = \frac{\partial}{\partial x}\left(\frac{\kappa(d)}{s(d)} d \frac{\partial h}{\partial x}\right) + \frac{\partial}{\partial y}\left(\frac{\kappa(d)}{s(d)} d \frac{\partial h}{\partial y}\right) + \frac{P(t) - E(t)}{s(x, y, d)}$$

in which:

h is water-table elevation above a datum (set below the peat) [L];

d is the thickness of flow (i.e., the local height of the water table above an underlying assumed impermeable layer [mineral soil, sediment or rock]) [L];

t is time [T];

x is horizontal distance in the *x* coordinate direction [L];

y is horizontal distance in the *y* coordinate direction [L];

s is drainable porosity [dimensionless];

κ is depth-averaged hydraulic conductivity below the water table [L T^{-1}];

P is the rate of rainfall addition to the water table [L T^{-1}];

E is the rate of evapotranspiration from the water table [L T^{-1}].

In this study, the DigiBog_Hydro model was set up using cells and layers to represent the modelled domain. The setup of the model is shown in Figure 4.4. The model domain represents a typical rectangular peat plot surrounded by ditches, in which ditch dam and bund presence varies across scenarios. The typical peat plot was represented by a set of active cells in the model domain. The grid cell size used was 1 m × 1 m in plan dimensions and was not varied across scenarios. The water level for the active cells was calculated based on the full Boussinesq equation during the simulation period. Fixed head (water level) —Dirichlet — boundary cells were used to represent ditches. The dams were represented by two different water levels in a ditch segment. The set water levels behind the dam (upstream) were higher than the levels set in front (downstream) of the dam. The bunds were represented using active cells containing peat layers with a low hydraulic conductivity (1×10^{-6} cm s^{-1}) and a low drainable porosity (0.11) (Table

4.1). As the bund depth was not assumed to reach the impermeable layer, the peat layers below the bunds were set to have the same hydraulic properties as the surrounding peat at that depth. The overall thickness of the model was 2 m, below which an impermeable layer was assumed.

In this study, we simulated a 0.2 km^2 (500 m × 400 m) area of drained peatland. For the Combined and Bunded scenarios, the area that was enclosed by the bunds (220 m × 180 m) was one fifth of the surface area of the selected typical plot (3.96×10^{-2} km^2). The restoration bunds were 30 cm in height above the peat surface. The thickness of the bunds was 100 cm. We considered such a partial bunded option for our simulations because if it is applied in the field, it would require less budget and would be less radical to local people than constructing the bund around the whole peat plot. The bunded area was located near to the lowest outlet of the peat plot (Figure 4.4), in which the closest bund corner was 28.2 m from the outlet of the peat plot. The bunded area was placed in the downstream corner of the model domain because it is the driest zone of the peat plot, inferred from the water-level data of a ditch-dam-blocked area studied in the field by Putra et al. (2021). Time-series model output data were obtained for two model 'monitoring points' shown in Figure 4.4. Point 1 is near the drainage outlet and in front of the bund in those scenarios where the bund is present. Point 2 is located within the main block of peat and occurs within the bund enclosure in the bunding scenarios.

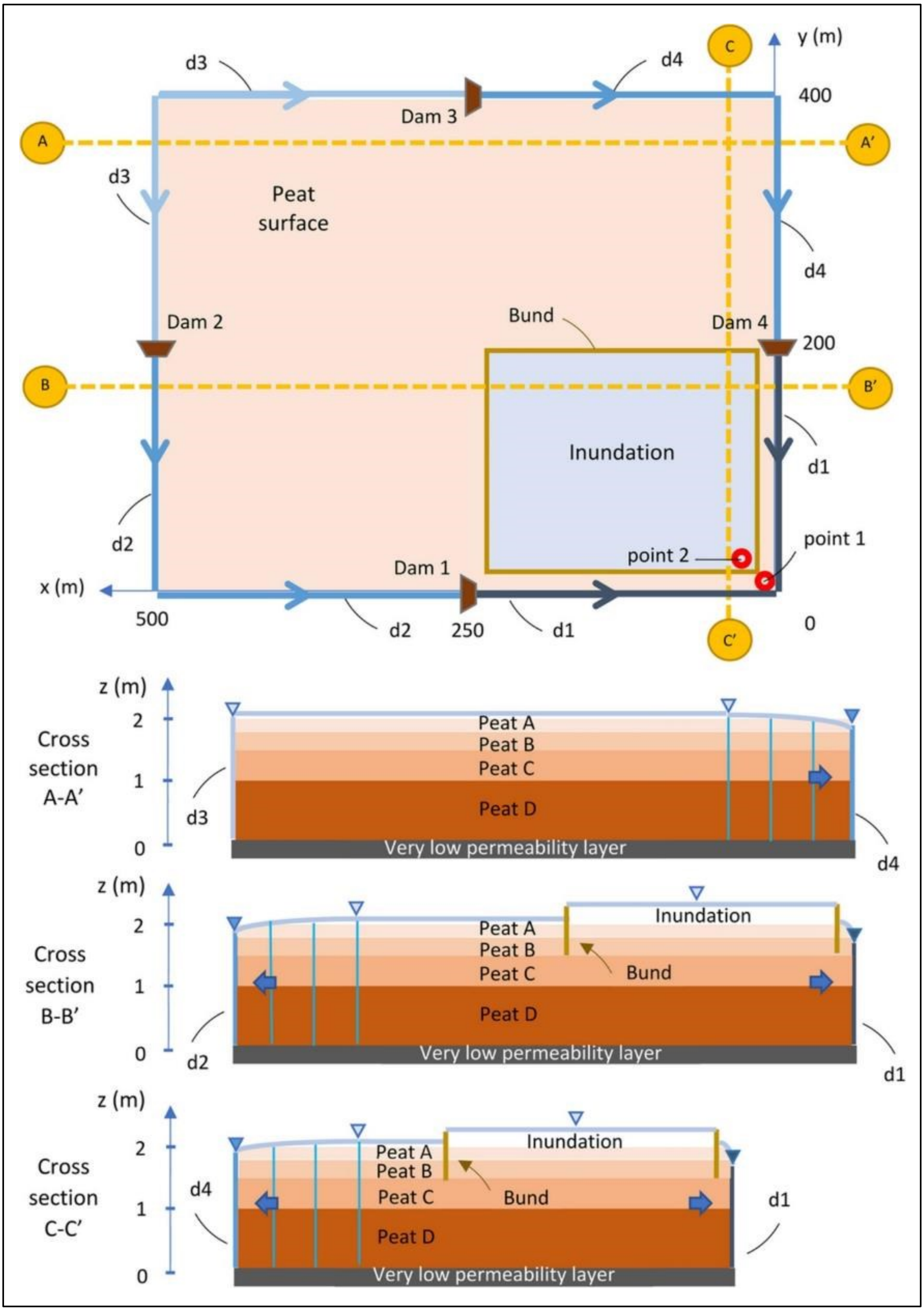

Figure 4.4 Plan and cross-sectional view of the DigiBog_Hydro modelling domain used in this study. The three cross-sections show the layers used in the simulations, considering the ditch water levels of Early Wet period. The triangles and light-blue curves show example water-table positions. The vertical light-blue lines and dark-blue arrows show the possible flow directions. There are four implemented types of Dirichlet boundary (d1 to d4). The red doughnut symbols are monitoring point 1 (in front of the bund, x = 10 m and y = 8 m) and monitoring point 2 (behind the bund, x = 40 m and y = 32 m). The main outlet of the modelled peat plot is located at x = 0 m and y = 0 m.

We represented surface water storage and movement in the model using virtual layers. In the un-bunded area, water was allowed to pond to a depth of 5 cm so that a virtual layer was used. This water could escape to the margin only by entering the peat below and moving laterally via subsurface flow. The 5-cm virtual layer (the uppermost layer) functioned to limit the ponding depth in the area, acting as a 'tank' that temporarily stored excess water from rainfall. No hydraulic conductivity value was applied to this layer and no surface runoff modelling was implemented for the virtual layer. Any rainfall causing the 5-cm ponding limit to be exceeded was assumed to be immediately lost to the model boundaries. In effect, this loss is the equivalent of rapid overland flow. In the area enclosed by the bunds, two layers were used to represent surface water storage and flow. To the top of the peat was added a 30-cm layer with a high hydraulic conductivity and drainable porosity (Table 4.1). Water could flow laterally 'through' this layer and through the lower-permeability bunds. It could also escape downwards and thence laterally through the deeper peat. On top of the 30-cm layer was a 5-cm layer the same as in the un-bunded area.

Table 4.2 DigiBog_Hydro model boundary conditions for different scenarios. All boundaries are Dirichlet cells, and the boundary water levels (WL) are measured in cm above the impermeable base of the 200-cm peat profile. The ditch locations are shown in Figure 4.4.

Scenarios	Modelling periods			
	Late Wet	**Early Dry**	**Late Dry**	**Early Wet**
With ditch-dams				
• WL at all d1 cells (outlet segment)	189	132	101	165
• WL at all d2 and d4 cells (upstream of Dam 1 and Dam 4)	195	137	110	187
• WL at all d3 cells (upstream of Dam 2 and Dam 3)	198	141	116	198
Without ditch-dams				
• All ditches WL	128	96	49	118

The total simulation time for each scenario was 364 days. In the Dammed and Combined scenarios, the effect of the dams on the ditch water levels was assumed to vary between dry and wet periods. Each yearly simulation was divided into four different 100-day periods, with 12 days overlap between periods (Table 4.2). The water-level output resulting from the preceding period was used as an initial condition for the next period. The results for periods of overlap were similar (fewer than 2 cm in difference), providing assurance that the model spin-up time was sufficient and indicating that the initial condition did not introduce artefacts to the results. Those periods were Late Wet (1 February to 11 May), Early Dry (1 May to 8 August), Late Dry (1 August to 8 November) and Early Wet (1 November to 1 February of the following year). The ditch water levels were set to different Dirichlet boundary water levels for each of the modelled periods and were kept constant during each period. The ditch water levels were based on data collected by Putra et al. (2021) and discussions with local forest rangers. The values that were used are given in Table 4.2, which were implemented across different climatic years.

4.3 Results

Our simulations show that, in the dry season, water levels in the Combined scenario (with ditch dams and bunds) were higher compared with water levels in the Control scenario. The effect of bunds in storing 'excess' water for the bunded area and its surroundings was distinct in the dry season. By contrast, in the dry season, the ditch-dam effects on water-level dynamics were limited around the ditches. The bund depth (in the On-surface and Extended scenarios) leads to different responses in terms of the amount of water stored behind the bunds, which depend on the state of peat degradation. Table 4.3 contains basic statistics from the modelling results (seasonal water-table depths and day counts of deep water-table condition) to support the graphical outputs in the figures.

Table 4.3 Basic statistics of the modelled scenario outputs. The water levels (WL) are measured in cm from the peat surface, in which negative values indicate those above the peat surface. The standard deviation (SD) is included for each mean value.

Results	Modelling periods			
	Late Wet	Early Dry	Late Dry	Early Wet
Mean water-table (with SD) in an El Niño year (cm)				
• In moderately degraded scenarios (cm)				
◦ Combined at Point 1	-4.0 (1.4)	6.1 (10.4)	74.1 (31.1)	19.0 (26.7)
◦ Combined at Point 2	-35.2 (1.0)	-30.4 (5.5)	6.5 (18.2)	-22.9 (13.6)
◦ Dammed at Point 1	-4.0 (1.4)	14.2 (16.1)	96.1 (32.2)	28.4 (33.7)
◦ Dammed at Point 2	-4.4 (0.8)	2.0 (8.1)	62.5 (26.5)	12.4 (22.2)
◦ Bunded at Point 1	-2.5 (3.4)	14.2 (16.2)	97.9 (33.3)	34.5 (34.4)
◦ Bunded at Point 2	-35.2 (1.0)	-30.4 (5.5)	6.6 (18.2)	-22.9 (13.6)
◦ Control at Point 1	-2.5 (3.4)	15.2 (16.9)	102.3 (33.9)	37.7 (35.6)
◦ Control at Point 2	-4.4 (0.8)	2.0 (8.1)	62.6 (26.5)	12.5 (22.2)
• In severely degraded scenarios (cm)				
◦ Combined at Point 1	0.5 (4.6)	9.4 (6.2)	76.8 (33.3)	32.1 (25.3)
◦ Combined at Point 2	-34.9 (1.2)	-28.4 (7.2)	19.8 (21.6)	-8.4 (20.2)
◦ Dammed at Point 1	0.5 (4.6)	27.9 (16.3)	116.5 (37.5)	48.7 (37.7)
◦ Dammed at Point 2	-4.2 (1.0)	4.4 (10.2)	69.6 (28.5)	28.7 (25.3)
◦ Bunded at Point 1	8.3 (7.6)	27.4 (15.2)	115.7 (38.8)	59.1 (37.4)
◦ Bunded at Point 2	-34.9 (1.3)	-28.4 (7.3)	20.9 (22.0)	-7.6 (20.6)
◦ Control at Point 1	8.4 (7.7)	28.6 (17.0)	124.7 (39.9)	67.0 (38.3)
◦ Control at Point 2	-4.2 (1.0)	4.3 (10.1)	69.5 (28.4)	28.7 (25.2)
Mean deep water-table days (with SD) during the dry periods (total 183 days) in an El Niño year				
• In front of bunds in moderately degraded scenarios (days)				
◦ Combined	-		97.0 (6.6)	
◦ Dammed	-		106.3 (9.8)	
◦ Bunded	-		109.8 (22.2)	
◦ Control	-		112.5 (21.5)	
• Behind the bunds in moderately degraded scenarios (days)				
◦ Combined	-		72.4 (4.3)	
◦ Dammed	-		102.7 (5.2)	
◦ Bunded	-		73.2 (5.6)	
◦ Control	-		102.6 (5.2)	
• In front of bunds in severely degraded scenarios (days)				
◦ Combined	-		88.1 (10.2)	
◦ Dammed	-		92.6 (7.0)	
◦ Bunded	-		93.1 (19.9)	
◦ Control	-		96.0 (17.7)	
• Behind the bunds in severely degraded scenarios (days)				
◦ Combined	-		20.1 (2.5)	
◦ Dammed	-		90.0 (0.0)	
◦ Bunded	-		20.7 (2.6)	
◦ Control	-		90.0 (0.0)	

4.3.1 Seasonal water-level dynamics

Figure 4.5 shows how water levels vary in relation to seasons, ENSO conditions, peatland degradation and restoration measures. In the Late Wet period, the water levels across different scenarios ranged from being above the peat surface to 10 cm below the peat surface. In the Early Dry period, water levels started to decrease, although inundation still occurred, depending on rainfall and water levels in the preceding Late Wet period. Water levels continued to fall in the Late Dry period to reach the lowest level of the year (levels varied across scenarios). Water levels rose again in the Early Wet period with the increase in net rainfall. Water levels in Sev-Dgr were normally higher than in Mod-Dgr (in all scenarios), especially in the dry period. In the wet period, water levels in both Sev-Dgr and Mod-Dgr at monitoring point 2 were mostly above the peat surface (> 95% of the time).

The scenarios with bunds had higher dry season water levels than the scenarios without bunds. The water-level responses to bunding were different behind and in front of the bund. Behind the bund (monitoring point 2), in the scenarios involving bunds, water levels were above the peatland surface in the dry periods of the La Niña and Neutral years. At point 2, the Combined and Bunded scenarios resulted in higher dry season water levels than the Control scenario, in which the largest differences were 72 cm (in Mod-Dgr) and 69 cm (in Sev-Dgr), for day 272 of the simulated El Niño period. In the wet period, at point 2, the water levels in the Combined and Bunded scenarios were 30–35 cm higher than in the Control scenario. At point 2, there was no difference in the water-level pattern between the Dammed and Control scenarios across seasons. In front of the bund (monitoring point 1), the water levels in the Dammed and Bunded scenarios were similar to those for the Control scenario (fewer than 5 cm in difference, see Figure 4.5a,c), but the water level for the Combined scenario was unique. In the dry periods of the La Niña and Neutral years, peatland water-table depths at point 1 in the Control scenario were deep (up to 74 cm in Mod-Dgr and 59 cm in Sev-Dgr), but those in the Combined scenario were shallower (up to 45 cm in Mod-Dgr and 50 cm in Sev-Dgr). In the Early Wet period, at point 1, the water level in the Dammed scenario rose for 20 days in Sev-Dgr and 30 days in Mod-Dgr, faster than in the Control scenario, and also the green dashed lines were not overlaid on the red dotted lines during this period (see Figure 4.5a,c).

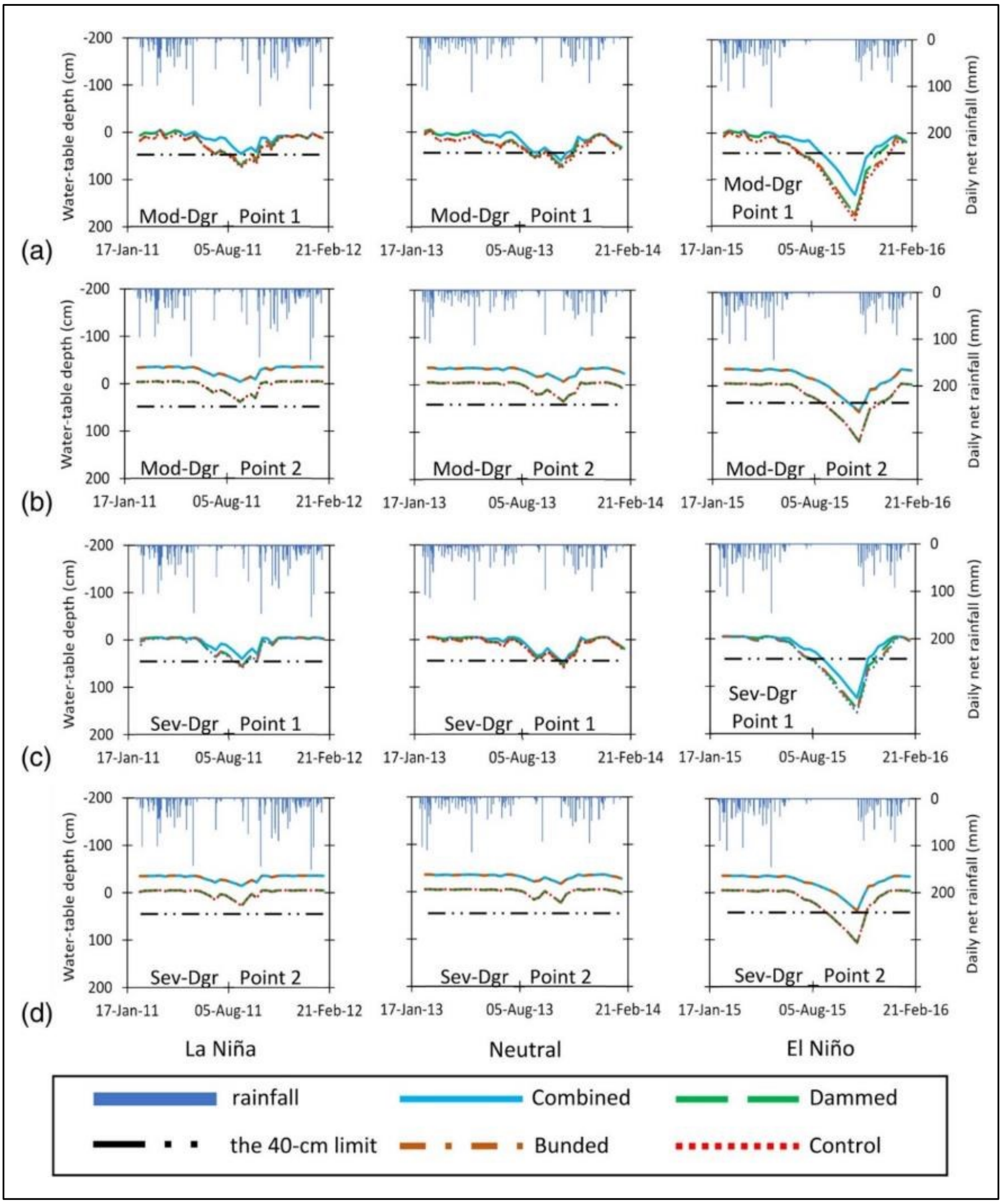

Figure 4.5 Water-table depth time series for different modelled scenarios.

There were different water-level patterns across the three ENSO conditions. In the modelled La Niña year of 2011, all water tables were shallower than the 40-cm depth limit, except around the ditches (e.g., monitoring point 1) for a part of the Late Dry period (for 50 days in Sev-Dgr and 70 days in Mod-Dgr). The La Niña water-level profiles show clearly that the supplied rainfall maintained water levels above the 40-cm limit, despite the drainage effect of the ditches. The Neutral year of 2013 resulted in similar water-level profiles to the La Niña year, but with fewer fluctuations during the dry periods. There was also a decrease in water level at the end of the Neutral year, which was not found in the La Niña year. The El Niño year of 2015 resulted in the steepest water-level decrease in the Late Dry period. The water-level fluctuation in the period without rain (1 August to 30 October 2015) was not distinct. The El Niño year resulted in large water-level recoveries after rainfall events in the Early Wet period, such as at monitoring point 1 (up to 150 cm in Sev-Dgr and 185 cm in Mod-Dgr compared with the condition at the beginning of the Early Dry period).

4.3.2 Spatial water-level profiles

Figure 4.6 presents the performance of the different restoration measures during the Late Dry and Early Wet periods. The performance is based on the accumulated number of deep water-table days during those periods, defined as days when the water table is deeper than 40 cm from the ground surface. The spatial variations of deep water-table days presented in Figure 4.6 are only for the Late Dry and Early Wet periods, as the water levels were above the 40-cm depth policy limit in the Late Wet and Early Dry periods. Figure 4.7 shows a typical water-level surface profile from each restoration scenario for day 232 of the simulation, an ordinary no-rain day within the Late Dry period.

The ditch drainage effect in lowering peatland water levels was more intense in Mod-Dgr than in Sev-Dgr. In Mod-Dgr, the zone with the intense drainage effect (>90 deep water-table days) was within 26 m of the ditch (see Figure 4.6). The intensely drained zone was just within 12 m of the ditch in Sev-Dgr. The number of days with deep water tables in Mod-Dgr was more than for Sev-Dgr (see Table 4.3). The water-table profile for a modelled dry day shows that the slope of the water-level in the area near the ditches was steeper in Sev-Dgr than in Mod-Dgr (see Figure 4.7). The water level at points distant from dams, bunds, and ditches, had a relatively flat profile (fewer than 5 cm water-level difference).

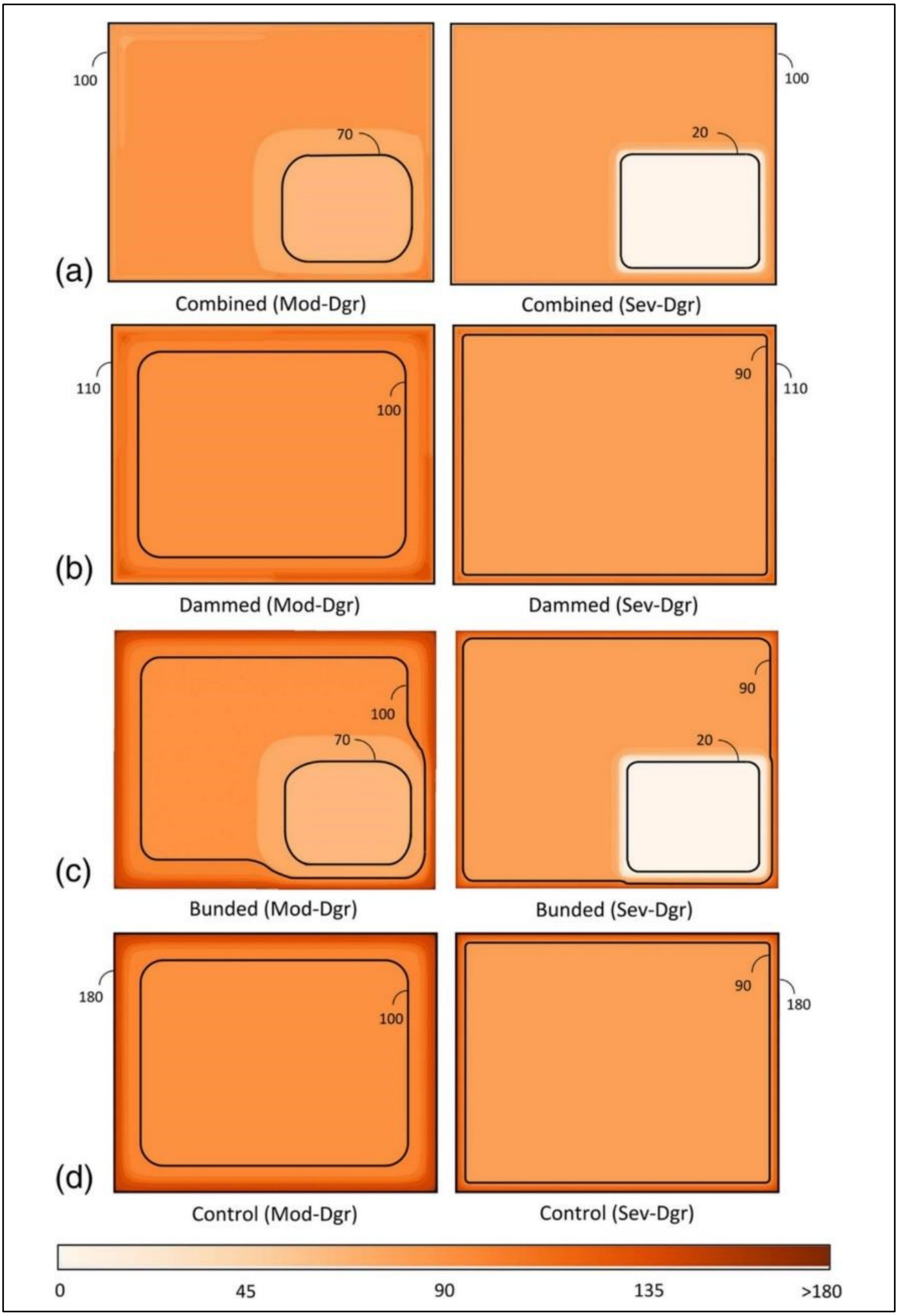

Figure 4.6 Spatial variations in the accumulated number of deep water-table days during the Late Dry and Early Wet periods (total 183 days) in an El Niño year. The spectral scale bar and contours represent the number of days in which the water table at a certain point in the peatland is deeper than 40 cm from the surface.

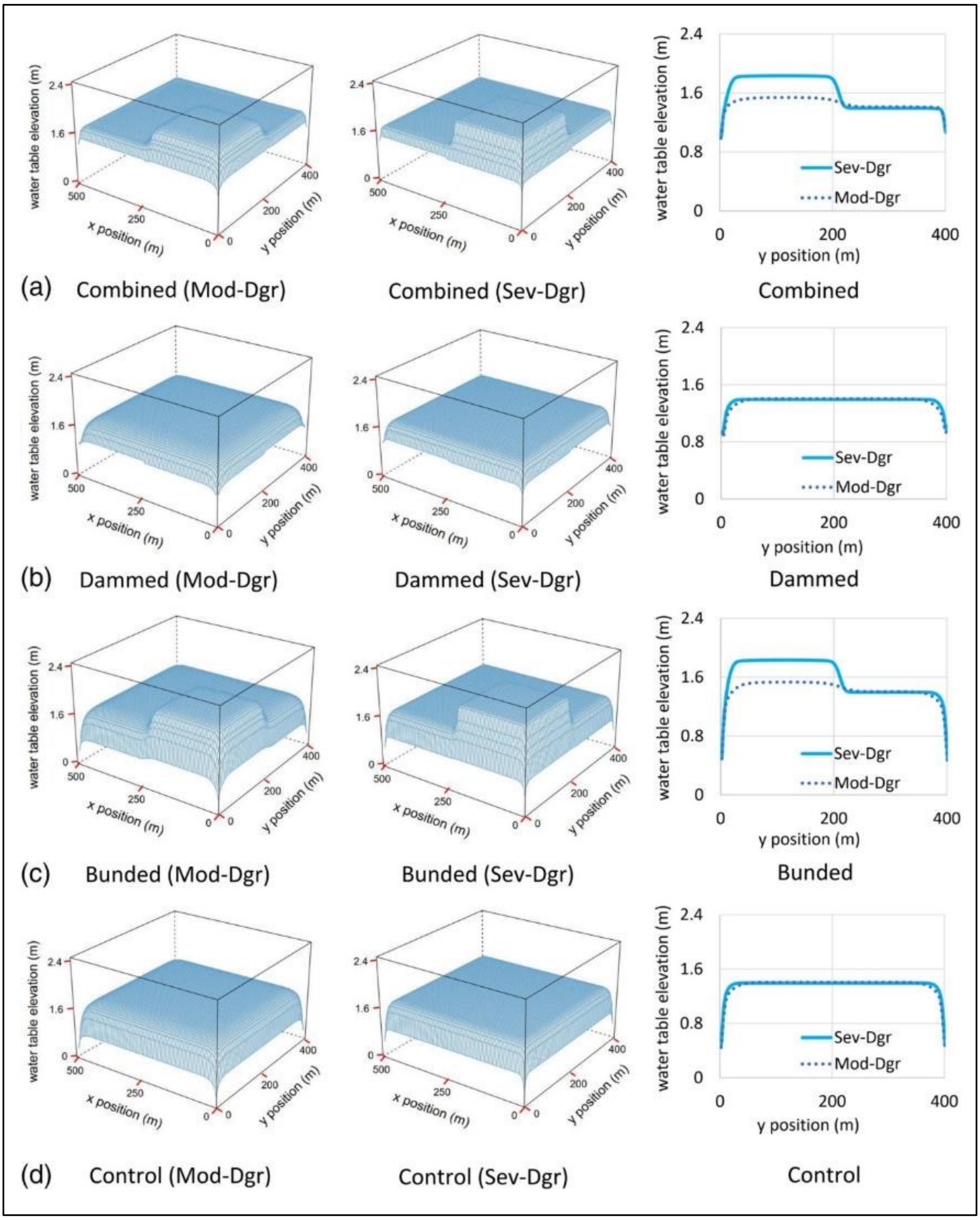

Figure 4.7 Three and two-dimensional spatial water-table profiles on different peatland conditions during the Late Dry period of the El Niño year (simulation day 232). The two-dimensional profiles are taken at cross section C–C′, which is along the line of x = 50 m.

The number of days with a deep water table in the Dammed scenario was fewer than for the Control scenario; this was true for all model cells. For both the Dammed and Control scenarios, there were more deep water-table days near ditches than elsewhere in the peat plot (Figure 4.6). In the Dammed scenario, there were slightly more deep-water table days around ditches near the peat plot outlet (d1) than around ditches farther from the outlet (d3) (the differences were up to 10 days in Sev-Dgr and 20 days in Mod-Dgr). By contrast, in the Control scenario, deep water-table days were similar within the area around all ditches, which were around 130 days in Mod-Dgr and 160 days in Sev-Dgr. On day 232, a typical dry day of the El Niño Year (Figure 4.7), the water level within 26 m of the ditches in the Dammed scenario was higher than that in the Control scenario (e.g., it was 40 cm higher at point $x = 50$ m and $y = 1$ m). However, on that example dry day, the water levels at points farther than 26 m from ditches in both the Dammed and Control scenarios were almost the same (approximately 60 cm below peat surface at the centre of the peat plot).

In the Bunded scenario, the bund reduced deep water-table days in the area behind the bund by 70 days in Sev-Dgr and 30 days in Mod-Dgr compared with the Control scenario (see Table 4.3). The bunded area also supplied water to the surrounding zone, in which the supplied area was wider in Mod-Dgr than in Sev-Dgr (29% compared with 24% of the total area during the dry period of an El Niño year, see Figure 4.6a,c). However, the bund did not reduce deep water-table days in the area near the ditches, as the number of days were similar for the Bunded and Control scenarios. The slope of the water table near bunds in Sev-Dgr was sharper than the one in Mod-Dgr (Figure 4.7c). The Combined scenario had the fewest deep water-table days among the scenarios (see Table 4.3). Locations with deep water-level days did not exist during the Late Wet and Early Dry periods of the El Niño year in the Combined scenario, either in Mod-Dgr or in Sev-Dgr, except in areas near ditches. In the Combined scenario, deep water-table days for the non-bunded area were more than for the bunded area (between 88 and 97 days compared with between 20 and 72 days), during the Late Dry and Early Wet periods (total 183 days) of the 2015 El Niño year.

4.3.3 The performance of the different bund types

Figure 4.8 shows that bund types perform differently in maintaining water level on drained tropical peatland. The subtraction of the water-table depths of the Extended bund scenario from the Surface bund scenario is referred to here as *wt-diff*. As is to be expected, the *wt-diff* values were higher in the bunded area (monitoring point 2) than for any points outside the bunded area (e.g., monitoring point 1). Overall, the time series graphs of *wt-diff* were similar between the Bunded and Combined scenarios. The *wt-diff* in the El Niño year was greater than in the La Niña or Neutral years, which was up to 20 cm in Mod-Dgr or 10 cm in Sev-Dgr during the dry period.

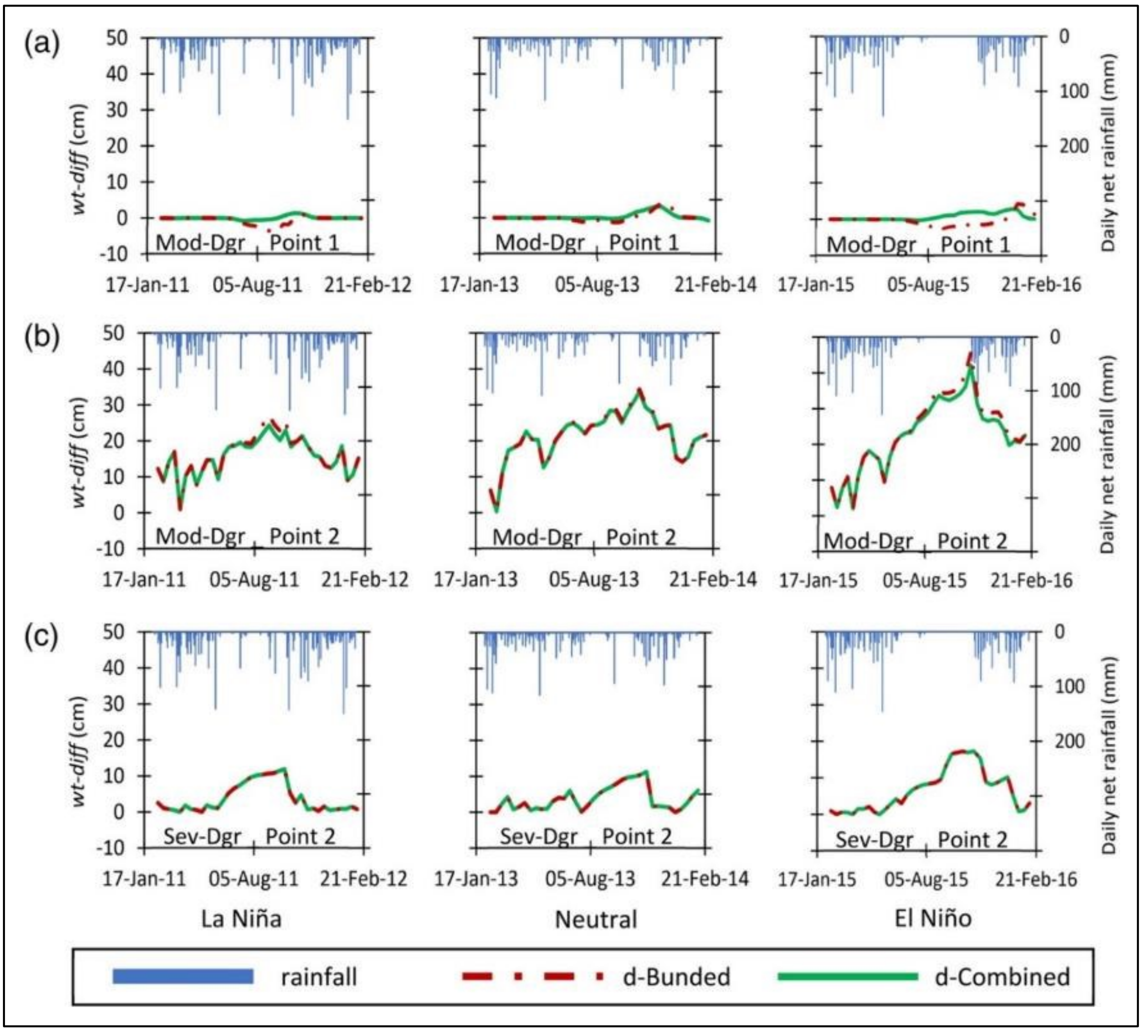

Figure 4.8 Differences in water-table depths between the Extended bund and the Surface bund scenarios (*wt-diff*). The negative values indicate that the water table of the Extended bunds scenario was deeper than that of the Surface bunds scenario. The red dashed lines (d-Bunded) are results from the Bunded scenario. The green lines (d-Combined) are results from the Combined scenario.

In Sev-Dgr, at monitoring point 2, the *wt-diff* values were fewer than 20 cm across seasons. The *wt-diff* became larger towards the end of the Late Dry period, before it receded towards zero when more rainfall occurred during the Early Wet period. Nevertheless, in Sev-Dgr, the *wt-diff* fluctuations were not simply related to rainfall patterns. The maximum *wt-diff* values decreased by about 9 cm in 10–15 days after a series of rainfall events occurred in the Early Wet period (see Figure 4.8c). In Sev-Dgr, during the La Niña or Neutral years, the *wt-diff* recession occurred only during the final phase of the Late Dry period. By contrast, during the El Niño year, the *wt-diff* recession lasted from the Late Dry period to almost the end of the Early Wet period.

In Mod-Dgr, in the bunded area, the first and third quartiles of *wt-diff* were around 10 to 25 cm in the La Niña year, 15 to 30 cm in the Neutral year, and 5 to 35 cm in the El Niño year. The maximum *wt-diff* value was 45 cm, occurring during the Late Dry period of the El Niño year. In Mod-Dgr, the low *wt-diff* values (near to zero) were found in the Late Wet period of all different climatic years, usually when the bunded area was inundated. In Mod-Dgr, the *wt-diff* increased during periods with fewer rainfall events (dry periods — Figure 4.8), but deceased when there were days with a daily net rainfall of at least 50 mm. In Mod-Dgr, the *wt-diff* values outside of the bunded area were close to zero (fewer than 5 cm) (see Figure 4.8a).

4.4 Discussion

4.4.1 The effects of bunds and ditch dams on water levels

Bunds increase the amount of water that can be stored in a peatland during the wet season. They store surface water that would otherwise be lost as overland flow to the ditches. The stored water in the bunded area can then ‘subsidize’ the ET demand and replenish subsurface seepage losses to the ditches during the dry season.

Our modelling results suggest that bunds can be used as a promising restoration solution for maintaining higher water levels in formerly drained tropical peatlands during El Niño years. Considering the moderate to high permeability of peat in tropical peatlands (as reported by Baird et al. (2017), Cobb and Harvey (2019), and Kelly et al. (2014)), extra water storage is necessary in dry periods. Bunds would be best placed in the areas that have the most deep-water-table days, for example near the lowest outlet of the model peat plot. Bunds could be placed at another adjacent location, but the stored water would need to be channelled from the bunded areas to un-bunded parts of a peatland if the latter become too dry (pipes with valves could be fitted across the bunds for this purpose).

Choice of bund design (depth) must be related to peat degradation conditions. In Sev-Dgr, there is no need for extended bunds as the performance of both bund types was nearly the same. The similar performance was expected because the lower hydraulic conductivity of peat in Sev-Dgr reproduces the flow dampening effect of the extended bund. Several studies have confirmed that low hydraulic conductivity peat tends to trigger surface runoff and/or inundation rather than subsurface flow (Holden et al., 2014; Rezanezhad et al., 2016; Crockett et al., 2016). However, in Mod-Dgr, during the dry period in an El Niño year, the maximum *wt-diff* was nearly 50 cm, which suggests extended bunds should be used. The extended bunds also set longer paths for the water to flow from the bunded area to its surroundings, creating a longer water retention time (e.g. the sheet piling effect in peatland, see Armstrong et al. (2009), Schimelpfenig et al. (2014), and Huth et al. (2020)).

Our findings show that ditch dams may boost bund performance, as the Combined scenario have the lowest number of deep water-table days in the non-bunded areas of the model peat plot. First, the ditch dams reduce water-level variations in the areas near the ditches (Kasih et al., 2016; Urzainki et al., 2020; Putra et al., 2021). Second, the dams minimize the hydraulic gradient between ditches and the peatland, as also indicated by several comparable studies in tropical (Susilo, Yamamoto, Imai, Ishii, et al., 2013; Ritzema et al., 2014; Planas-Clarke et al., 2020) or temperate peatlands (Peacock et al., 2015; Holden et al., 2017; Evans et al., 2018). Third, in the Early Wet period, the ditch dams raised the ditch water levels, reduced flow to the margin of the peat, and accelerated water refilling in the model plot. However, ditch dams by themselves will not be of great help in reducing the number of deep water-table days during an El Niño year, as the Dammed and the Control scenarios show (both had a similar low-water-level pattern).

The results indicate that bunds, when combined with ditch dams, perform well for the La Niña and Neutral years, but not so well during part of the dry period of the El Niño year. Enlarging the area which is bunded and perhaps increasing the bund height may enhance water storage and maintain water table during the dry period. For the Mod-Dgr peatland, the extension of the bund depth is an alternative to reduce water-table drawdowns in the bunded area, given that the Extended scenario performed better in maintaining water than the On-Surface one.

4.4.2 Benefits of modelling water-table restoration

Hydrodynamic models, such as DigiBog_Hydro, can be used in advancing our understanding of tropical peatland water-level dynamics for different peat degradation and climatic conditions. We have shown how different ENSO conditions resulted in different water-level dynamics in a typical tropical peatland, a finding that was previously suggested by multi-year field studies from a few dipwells (Ishikura et al., 2017, 2018; Tsuji et al., 2019). Deep water-level conditions in a drained tropical peatland depend strongly on the dry season net rainfall rather than the total yearly net rainfall, in line with findings from other studies (Putra and Hayasaka, 2011; Ritzema et al., 2014; Mezbahuddin et al., 2015; Deshmukh et al., 2021). Our modelling approach could allow assessment of restoration plans under more extreme meteorological conditions, which are expected within future climate-change scenarios (see IPCC (2021)).

DigiBog_Hydro is similar to the model used by Urzainki et al. (2020), who investigated how canal dams affect peat water tables across larger scales than considered here. However, Urzainki et al. (2020) did not consider surface water storage and the role of bunds or dams within a peat-plot domain. Models such as Modflow may also be used to investigate peatland water-table behaviour (see Reeve et al. (2006) and Painter et al. (2008)), but can be difficult to apply to systems with fine-scale variations in near-surface peat properties and where the water-table is highly dynamic.

Our study provides a site-based water-level modelling approach for tropical peatland restoration planning and assessment, as an alternative to the water-level optimization approach (Urzainki et al., 2020) or the canal-slope based approach (Jaenicke et al., 2010). Unlike coarser-scale studies (Jaenicke et al., 2010; Ishii et al., 2016; Cobb and Harvey, 2019; Urzainki et al., 2020), our fine-scale study (1 m × 1 m cell resolution) allows investigation of water-level variations in areas near ditches and bunds, which is important when bunds are usually only 1–2 m in thickness. Our fine-scale approach can accommodate different ditch-dam and bund placements in a typical small peat plot ($0.2\ km^2$) that cannot be set in coarse-scale studies. Our approach can also include variation of microtopography in typical small restoration plots, as demonstrated by the difference in the modelled surface elevation in the area behind and in front of the bunds. Thus, our modelling approach could be adopted by local agencies for tropical peatland restoration to support practical design of restoration features. In doing so, peatland managers would reduce the risk of putting in place ineffective restoration measures, an important concern when trying to maximize benefits from resources allocated to peatland restoration (Hansson and Dargusch, 2018; Ota et al., 2020; Parish et al., 2021; Sari et al., 2021). Nevertheless, our fine-scale study requires shorter modelling time steps (e.g., less than a minute), meaning more computational resources (e.g., for a comparable modelling area) compared with coarse-scale studies. In addition, empirical peat physical properties data, meteorological data and ditch/peatland water-level data are still limited in tropical peatland settings and would add value to fine-scale modelling studies (Hoyt et al., 2019; Tsuji et al., 2021; Nguyen-Thi et al., 2021; Deshmukh et al., 2021).

4.4.3 Modelling limitations and further study

This research has not explored the overall effects of bunding on the ecosystem. The inundation on the bunded area may affect vegetation survival or recruitment of vegetation inside the bund (Jans et al., 2012; Lampela et al., 2016). This ecological constraint needs to be considered when determining the coverage of the bunded area in the peat plot. Inundation may also enhance methane release, and further research is required into such effects. In a place where more than 3 months of inundation on the bunded area is unacceptable, it is suggested to drain the storage of the bunded area in the wet period but allow it to fill fully at the beginning of the Early Dry period. Keeping the peatland sufficiently wet (Evers et al., 2017; Ismail et al., 2021) and maintaining peatland water tables near to the surface should reduce fire risk (Wösten et al., 2006; Page et al., 2009).

Given that the ET is strongly dependent on the water-table depth (Hirano et al., 2015; Deshmukh et al., 2021), the approximation of using fixed ET values for moderately and severely degraded peatlands in this study may not reflect the real condition. The scarcity of onsite ET measurements from the Sebangau tropical peatlands did not allow daily ET data to be included in the simulations for the associated ENSO years. Considering common seasonal ET variations in tropical peatlands (Putra et al., 2021; Ohkubo et al., 2021a, 2021b), our results may slightly overestimate water levels in the wet periods and underestimate them in the dry periods. We expect that daily ET will be higher in the bunded area when inundation occurred, which is perhaps similar to the wet period daily ET determined by Ohkubo et al. (2021a, 2021b).

DigiBog_Hydro model is sensitive to the value of the hydraulic conductivity and drainable porosity parameters (Baird et al., 2017; Young et al., 2017), and more field-based data from tropical peatlands are required on these properties to improve the robustness of hydrodynamic modelling experiments like those conducted in this study. Homogeneous peat layers and the assumption of a flat peat surface may not reflect real site conditions. Lateral variability of peat properties as a result of drainage and disturbances (e.g. fire) (Sinclair et al., 2020; Dhandapani et al., 2022) is to be expected and zones of by-passing flow (perhaps due to soil pipes) may occur in field conditions. Those factors should be evaluated in field studies and accounted for in model simulations.

While our study provides an insight into the potential benefits of bunding on drained tropical peatlands, it is important to understand whether local communities can implement such practices. In Sebangau, the arrangement of agricultural ridges (acting as bunds) could be converted from parallel rows (e.g., with 2 m intervals) to rectangular grids (e.g., with 10 m intervals), while still allowing agricultural practice. In order to minimize disturbances to farming activities on the ridges by higher water levels, ridge height could be increased or plants that are more resilient to high water-table conditions could be planted (Uda et al., 2020; Budiman et al., 2020; Tan et al., 2021). In temperate agricultural peatlands, the same problem exists and trials with waterlogging-tolerant crops and cost-effective paludiculture practices are ongoing (e.g. see Tanneberger et al. (2020)). Thus, further work is required on the physical arrangement of bunding solutions and the related socio-economic requirements for agricultural production.

4.5 Conclusions

This article demonstrates the use of a hydrodynamic model (DigiBog_Hydro) depicting tropical peatland water-level dynamics over fine spatial scales. By incorporating information on peat properties, net rainfall, and ditch water levels, DigiBog_Hydro can be used to plan restoration infrastructure before it is installed in the field, or to assess existing restoration arrangements. The installation of bunds and ditch dams allowed more water storage in a typical drained tropical peat plot during dry periods compared with conditions without restoration. Ditch dams alone reduced hydraulic gradients in the zone up to 26 m from the ditches but had a limited effect on peatland water levels during the dry season. In such dry periods, bunds were not only able to maintain a higher water level for the area enclosed by the bund, but also supplied water to surrounding un-bunded parts of the peatland. The existence of ditch dams and bunds, as well as the type of bunds used, influenced how long water tables took to recover during the early part of the wet season. However, the performance of either ditch dams or bunds depends on the degree of peat degradation (peat hydrological properties) and seasonal weather patterns (net rainfall supply). Our results strongly suggest that the construction of bunds in combination with ditch dams would be beneficial when restoring drained tropical peatlands under different ENSO conditions.

4.6 Data availability statement

The inputs, settings, and outputs of the model that are presented in this study are available via the University of Leeds data repository under Creative Commons BY-NC 4.0 Licence (https://doi.org/10.5518/1053).

4.7 References

Agus, C., Azmi, F.F., Widiyatno, Ilfana, Z.R., Wulandari, D., Rachmanadi, D., Harun, M.K. and Yuwati, T.W. 2019. The Impact of Forest Fire on the Biodiversity and the Soil Characteristics of Tropical Peatland *In*: W. L. Filho, J. Barbir and R. Preziosi, eds. *Handbook of Climate Change and Biodiversity. Climate Change Management*. Springer Cham, pp.287–303.

Armstrong, A., Holden, J., Kay, P., Foulger, M., Gledhill, S., McDonald, A.T. and Walker, A. 2009. Drain-blocking techniques on blanket peat: A framework for best practice. *Journal of Environmental Management*. **90**(11), pp.3512–3519.

Baird, A.J., Gill, P. and Young, D. 2020. *DigiBog Hydro User Manual*. [Online]. Leeds, UK. Available from: https://water.leeds.ac.uk/wp-content/uploads/sites/36/2020/12/DigiBog_Hydro_user_manual_v1_FINAL.pdf.

Baird, A.J., Low, R., Young, D., Swindles, G.T., Lopez, O.R. and Page, S.E. 2017. High permeability explains the vulnerability of the carbon store in drained tropical peatlands. *Geophysical Research Letters*. **44**(3), pp.1333–1339.

Baird, A.J., Morris, P.J. and Belyea, L.R. 2012. The DigiBog peatland development model 1: rationale, conceptual model, and hydrological basis. *Ecohydrology*. **5**(3), pp.242–255.

Ballhorn, U., Siegert, F., Mason, M. and 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*. **106**(50), pp.21213–21218.

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

Boehm, H.-D. V. and Siegert, F. 2001. Ecological impact of the One Million Hectare Rice Project in Central Kalimantan, Indonesia, using Remote Sensing and GIS *In*: *Paper presented at the 22nd Asian Conference on Remote Sensing*. [Online]. Center for Remote Imaging, Sensing and Processing, National University of Singapore, pp.1–6. Available from: http://www.crisp.nus.edu.sg/~acrs2001/pdf/126boehm.pdf.

Budiman, I., Bastoni, Sari, E.N., Hadi, E.E., Asmaliyah, Siahaan, H., Januar, R. and Hapsari, R.D. 2020. Progress of paludiculture projects in supporting peatland ecosystem restoration in Indonesia. *Global Ecology and Conservation*. **23**, article no.: e01084.

Carlson, K.M., Goodman, L.K. and May-Tobin, C.C. 2015. Modeling relationships between water table depth and peat soil carbon loss in Southeast Asian plantations. *Environmental Research Letters*. **10**(7), article no.: 074006.

Cobb, A.R. and Harvey, C.F. 2019. Scalar Simulation and Parameterization of Water Table Dynamics in Tropical Peatlands. *Water Resources Research*. **55**(11), pp.9351–9377.

Cobb, A.R., Hoyt, A.M., Gandois, L., Eri, J., Dommain, R., Abu Salim, K., Kai, F.M., Haji Su'ut, N.S. and Harvey, C.F. 2017. How temporal patterns in rainfall determine the geomorphology and carbon fluxes of tropical peatlands. *Proceedings of the National Academy of Sciences*. **114**(26), pp.E5187–E5196.

Cooper, H. V., Vane, C.H., Evers, S., Aplin, P., Girkin, N.T. and Sjögersten, S. 2019. From peat swamp forest to oil palm plantations: The stability of tropical peatland carbon. *Geoderma*. **342**, pp.109–117.

Couwenberg, J., Dommain, R. and Joosten, H. 2010. Greenhouse gas fluxes from tropical peatlands in south-east Asia. *Global Change Biology*. **16**(6), pp.1715–1732.

Crockett, A.C., Ronayne, M.J. and Cooper, D.J. 2016. Relationships between vegetation type, peat hydraulic conductivity, and water table dynamics in mountain fens. *Ecohydrology*. **9**(6), pp.1028–1038.

Dadap, N.C., Cobb, A.R., Hoyt, A.M., Harvey, C.F. and Konings, A.G. 2019. Satellite soil moisture observations predict burned area in Southeast Asian peatlands. *Environmental Research Letters*. **14**(9), article no.: 094014.

Dargie, G.C., Lewis, S.L., Lawson, I.T., Mitchard, E.T.A., Page, S.E., Bocko, Y.E. and Ifo, S.A. 2017. Age, extent and carbon storage of the central Congo Basin peatland complex. *Nature*. **542**(7639), pp.86–90.

Desai, B.H. 2017. United Nations Environment Program (UNEP). *Yearbook of International Environmental Law*. **28**(14), pp.498–505.

Deshmukh, C.S., Julius, D., Desai, A.R., Asyhari, A., Page, S.E., Nardi, N., Susanto, A.P., Nurholis, N., Hendrizal, M., Kurnianto, S., Suardiwerianto, Y., Salam, Y.W., Agus, F., Astiani, D., Sabiham, S., Gauci, V. and Evans, C.D. 2021. Conservation slows down emission increase from a tropical peatland in Indonesia. *Nature Geoscience*. **14**(7), pp.484–490.

Dhandapani, S., Girkin, N.T. and Evers, S. 2022. Spatial variability of surface peat properties and carbon emissions in a tropical peatland oil palm monoculture during a dry season. *Soil Use and Management*. **38**(1), pp.381–395.

Dohong, A., Aziz, A.A. and Dargusch, P. 2018a. A review of techniques for effective tropical peatland restoration. *Wetlands*. **38**(2), pp.275–292.

Dohong, A., Aziz, A.A. and Dargusch, P. 2017. A review of the drivers of tropical peatland degradation in South-East Asia. *Land Use Policy*. **69**, pp.349–360.

Dohong, A., Aziz, A.A. and Dargusch, P. 2018b. Carbon emissions from oil palm development on deep peat soil in Central Kalimantan Indonesia. *Anthropocene*. **22**, pp.31–39.

Dohong, A., Cassiophea, L., Sutikno, S., Triadi, B.L., Wirada, F., Rengganis, P. and Sigalingging, L. 2017. *Modul pelatihan pembangunan infrastruktur pembasahan* *gambut sekat kanal berbasis masyarakat (Training Module for community based peatland rewetting infrastructure development)*. [Online]. Jakarta, Indonesia. Available from: https://brg.go.id/files/Publikasi/Modul_Sekat_Kanal_Berbasis_Masyarakat_Final_30082017.pdf.

Draper, F.C., Roucoux, K.H., Lawson, I.T., Mitchard, E.T.A., Honorio Coronado, E.N., Lähteenoja, O., Torres Montenegro, L., Valderrama Sandoval, E., Zaráte, R. and Baker, T.R. 2014. The distribution and amount of carbon in the largest peatland complex in Amazonia. *Environmental Research Letters*. **9**(12), article no.: 124017.

Evans, C.D., Peacock, M., Green, S.M., Holden, J., Chapman, P.J., Lebron, I., Callaghan, N., Grayson, R. and Baird, A.J. 2018. The impact of ditch blocking on fluvial carbon export from a UK blanket bog. *Hydrological Processes*. **32**(13), pp.2141–2154.

Evers, S., Yule, C.M., Padfield, R., O'Reilly, P. and Varkkey, H. 2017. Keep wetlands wet: the myth of sustainable development of tropical peatlands - implications for policies and management. *Global Change Biology*. **23**(2), pp.534–549.

Farmer, J., Matthews, R., Smith, P. and Smith, J.U. 2014. The Tropical Peatland Plantation-Carbon Assessment Tool: estimating CO_2 emissions from tropical peat soils under plantations. *Mitigation and Adaptation Strategies for Global Change*. **19**(6), pp.863–885.

Girkin, N.T., Lopes dos Santos, R.A., Vane, C.H., Ostle, N., Turner, B.L. and Sjögersten, S. 2020. Peat Properties, Dominant Vegetation Type and Microbial Community Structure in a Tropical Peatland. *Wetlands*. **40**(5), pp.1367–1377.

Glenk, K., Novo, P., Roberts, M., Sposato, M., Martin-Ortega, J., Shirkhorshidi, M. and Potts, J. 2020. *The costs of peatland restoration: Analysis of an evolving database based on the Peatland Action Programme in Scotland.* [Online]. [Accessed 1 September 2021]. Edinburgh. Available from: https://research-scotland.ac.uk/handle/20.500.12594/9745.

Graham, L.L.B., Giesen, W. and Page, S.E. 2017. A common-sense approach to tropical peat swamp forest restoration in Southeast Asia. *Restoration Ecology*. **25**(2), pp.312–321.

Hansson, A. and Dargusch, P. 2018. An Estimate of the Financial Cost of Peatland Restoration in Indonesia. *Case Studies in the Environment*. **2**(1), pp.1–8.

Hapsari, K.A., Biagioni, S., Jennerjahn, T.C., Reimer, P.M., Saad, A., Achnopha, Y., Sabiham, S. and Behling, H. 2017. Environmental dynamics and carbon accumulation rate of a tropical peatland in Central Sumatra, Indonesia. *Quaternary Science Reviews*. **169**, pp.173–187.

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. and Veen, F.J.F. 2020. Tropical forest and peatland conservation in Indonesia: Challenges and directions. *People and Nature*. **2**(1), pp.4–28.

Hergoualc'h, K. and Verchot, L. V. 2011. Stocks and fluxes of carbon associated with land use change in Southeast Asian tropical peatlands: A review. *Global Biogeochemical Cycles*. **25**(2), article no.: GB2001.

Hirano, T., Kusin, K., Limin, S. and Osaki, M. 2015. Evapotranspiration of tropical peat swamp forests. *Global Change Biology*. **21**(5), pp.1914–1927.

Holden, J., Green, S.M., Baird, A.J., Grayson, R.P., Dooling, G.P., Chapman, P.J., Evans, C.D., Peacock, M. and Swindles, G. 2017. The impact of ditch blocking on the hydrological functioning of blanket peatlands. *Hydrological Processes*. **31**(3), pp.525–539.

Holden, J., Wearing, C., Palmer, S., Jackson, B., Johnston, K. and Brown, L.E. 2014. Fire decreases near-surface hydraulic conductivity and macropore flow in blanket peat. *Hydrological Processes*. **28**(5), pp.2868–2876.

Honorio Coronado, E.N., Hastie, A., Reyna, J., Flores, G., Grández, J., Lähteenoja, O., Draper, F.C., Åkesson, C.M., Baker, T.R., Bhomia, R.K., Cole, L.E.S., Dávila, N., Del Águila, J., Del Águila, M., Del Castillo Torres, D., Lawson, I.T., Martín Brañas, M., Mitchard, E.T.A., Monteagudo, A., Phillips, O.L., Ramírez, E., Ríos, M., Ríos, S., Rodriguez, L., Roucoux, K.H., Tagle Casapia, X., Vasquez, R., Wheeler, C.E. and Montoya, M. 2021. Intensive field sampling increases the known extent of carbon-rich Amazonian peatland pole forests. *Environmental Research Letters*. **16**(7), article no.: 074048.

Hooijer, A., Page, S.E., Jauhiainen, J., Lee, W.A., Lu, X.X., Idris, A. and Anshari, G. 2012. Subsidence and carbon loss in drained tropical peatlands. *Biogeosciences*. **9**(3), pp.1053–1071.

Hoyt, A.M., Chaussard, E., Seppalainen, S.S. and Harvey, C.F. 2020. Widespread subsidence and carbon emissions across Southeast Asian peatlands. *Nature Geoscience*. **13**(6), pp.435–440.

Hoyt, A.M., Gandois, L., Eri, J., Kai, F.M., Harvey, C.F. and Cobb, A.R. 2019. CO_2 emissions from an undrained tropical peatland: Interacting influences of temperature, shading and water table depth. *Global Change Biology*. **25**(9), pp.2885–2899.

Huth, V., Günther, A., Bartel, A., Hofer, B., Jacobs, O., Jantz, N., Meister, M., Rosinski, E., Urich, T., Weil, M., Zak, D. and Jurasinski, G. 2020. Topsoil removal reduced in-situ methane emissions in a temperate rewetted bog grassland by a hundredfold. *Science of The Total Environment*. **721**, article no.: 137763.

International Climate Initiative 2021. The Global Peatlands Initiative: Assessing, Measuring and Preserving Peat Carbon. *Food and Agriculture Organization of the United Nations*. [Online]. [Accessed 1 February 2021]. Available from: https://www.international-climate-initiative.com/en/details/project/the-global-peatlands-initiative-assessing-measuring-and-preserving-peat-carbon-18_III_096-3039.

IPCC 2021. Summary for Policymakers *In*: V. Masson-Delmotte, P. Zhai, A. Pirani, S. L. Connors, C. Péan, S. Berger, N. Caud, Y. Chen, L. Goldfarb, M. I. Gomis, M. Huang, K. Leitzell, E. Lonnoy, J. B. R. Matthews, T. K. Maycock, T. Waterfield, O. Yelekçi, R. Yu and B. Zhou, eds. *Climate Change 2021: The Physical Science Basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change*. [Online]. Cambridge, UK: Cambridge University Press. In Press., pp.1–40. Available from: https://www.ipcc.ch/report/ar6/wg1/.

Ishii, Y., Koizumi, K., Fukami, H., Yamamoto, K., Takahashi, H., Limin, S.H., Kusin, K., Usup, A. and Susilo, G.E. 2016. Groundwater in Peatland *In*: M. Osaki and N. Tsuji, eds. *Tropical Peatland Ecosystems*. Tokyo: Springer Japan, pp.265–279.

Ishikura, K., Hirano, T., Okimoto, Y., Hirata, R., Kiew, F., Melling, L., Aeries, E.B., Lo, K.S., Musin, K.K., Waili, J.W., Wong, G.X. and Ishii, Y. 2018. Soil carbon dioxide emissions due to oxidative peat decomposition in an oil palm plantation on tropical peat. *Agriculture, Ecosystems & Environment*. **254**, pp.202–212.

Ishikura, K., Yamada, H., Toma, Y., Takakai, F., Morishita, T., Darung, U., Limin, A., Limin, S.H. and Hatano, R. 2017. Effect of groundwater level fluctuation on soil respiration rate of tropical peatland in Central Kalimantan, Indonesia. *Soil Science* *and Plant Nutrition*. **63**(1), pp.1–13.

Ismail, I., Haghighi, A.T., Marttila, H., Kurniawan, U., Karyanto, O. and Kløve, B. 2021. Water table variations on different land use units in a drained tropical peatland island of Indonesia. *Hydrology Research*. **52**(6), pp.1372–1388.

Jaenicke, J., Wösten, H., Budiman, A. and Siegert, F. 2010. Planning hydrological restoration of peatlands in Indonesia to mitigate carbon dioxide emissions. *Mitigation and Adaptation Strategies for Global Change*. **15**(3), pp.223–239.

Jans, W.W.P., Dibor, L., Verwer, C., Kruijt, B., Tan, S. and van der Meer, P.J. 2012. Effects of light and soil flooding on the growth and photosynthesis of Ramin (Gonystylus bancanus) seedlings in Malaysia. *Journal of Tropical Forest Science*. **24**(1), pp.54–63.

Kasih, R.C., Simon, O., Ansori, M., Pratama, M.P. and Wirada, F. 2016. Rewetting on Degraded Tropical Peatland by Canal Blocking Technique in Sebangau National Park, Central Kalimantan, Indonesia *In*: *Proceedings on 15th International Peat Congress 2016. Extended Abstract No: A-065*.

Kelly, T.J., Baird, A.J., Roucoux, K.H., Baker, T.R., Honorio Coronado, E.N., Ríos, M. and Lawson, I.T. 2014. The high hydraulic conductivity of three wooded tropical peat swamps in northeast Peru: measurements and implications for hydrological function. *Hydrological Processes*. **28**(9), pp.3373–3387.

Kobayashi, S. 2016. Peatland and Peatland Forest in Brunei Darussalam *In*: M. Osaki and N. Tsuji, eds. *Tropical peatland ecosystems*. Tokyo: Springer Japan, pp.75–89.

Könönen, M., Jauhiainen, J., Straková, P., Heinonsalo, J., Laiho, R., Kusin, K., Limin, S. and Vasander, H. 2018. Deforested and drained tropical peatland sites show poorer peat substrate quality and lower microbial biomass and activity than unmanaged swamp forest. *Soil Biology and Biochemistry*. **123**, pp.229–241.

Kurnianto, S., Selker, J., Boone Kauffman, J., Murdiyarso, D. and Peterson, J.T. 2019. The influence of land-cover changes on the variability of saturated hydraulic conductivity in tropical peatlands. *Mitigation and Adaptation Strategies for Global Change*. **24**(4), pp.535–555.

Lampela, M., Jauhiainen, J., Kämäri, I., Koskinen, M., Tanhuanpää, T., Valkeapää, A. and Vasander, H. 2016. Ground surface microtopography and vegetation patterns in a tropical peat swamp forest. *CATENA*. **139**, pp.127–136.

Land, D. and Brock, A. 2017. *Peatland Restoration Advice for the Norwegian Environment Agency*. [Online]. London, UK. Available from:

https://www.statsforvalteren.no/contentassets/c836cde50e1e448ca6a1897ff76531fe/peatland-restoration-advice-norwegian-ea-17oct17-djl.pdf.

Limin, S.H., Jentha and Ermiasi, Y. 2007. History of the Development of Tropical Peatland in Central Kalimantan, Indonesia. *Tropics*. **16**(3), pp.291–301.

Mawdsley, N., Giesen, W., Haag, A., Hooijer, A., van der Vat, M., Ichsan, N., Ibie, B.F. and Hidayat, T. 2009. *Strategic peatland rehabilitation plan for block A (north-west) in the Ex-Mega Rice Project Area, Central Kalimantan: A report prepared for the Kalimantan Forests and Climate Partnership. Euroconsult Mott MacDonald/Deltares| Delft Hydraulics*. Palangkaraya, Indonesia.

Medrilzam, M., Smith, C., Aziz, A.A., Herbohn, J. and Dargusch, P. 2017. Smallholder Farmers and the Dynamics of Degradation of Peatland Ecosystems in Central Kalimantan, Indonesia. *Ecological Economics*. **136**, pp.101–113.

Mezbahuddin, M., Grant, R.F. and Hirano, T. 2015. How hydrology determines seasonal and interannual variations in water table depth, surface energy exchange, and water stress in a tropical peatland: Modeling versus measurements. *Journal of Geophysical Research: Biogeosciences*. **120**(11), pp.2132–2157.

Miettinen, J., Shi, C. and Liew, S.C. 2017. Fire Distribution in Peninsular Malaysia, Sumatra and Borneo in 2015 with Special Emphasis on Peatland Fires. *Environmental Management*. **60**(4), pp.747–757.

Nguyen-Thi, T., Ngo-Duc, T., Tangang, F.T., Cruz, F., Juneng, L., Santisirisomboon, J., Aldrian, E., Phan-Van, T. and Narisma, G. 2021. Climate analogue and future appearance of novel climate in Southeast Asia. *International Journal of Climatology*. **41**(S1), pp.E392–E409.

Novitasari, N., Sujono, J., Harto, S., Maas, A. and Jayadi, R. 2018. Restoration of peat dome in ex-Mega rice project area in Central Kalimantan *In*: H. Prasetyo, N. Hidayati, E. Setiawan and T. Widayatno, eds. *The 4th International Conference on Engineering, Technology, and Industrial Application (ICETIA) 13–14 December 2017*. Surakarta, Indonesia: AIP Conference Proceedings, article no.: 040008.

Ohkubo, S., Hirano, T. and Kusin, K. 2021a. Assessing the carbon dioxide balance of a degraded tropical peat swamp forest following multiple fire events of different intensities. *Agricultural and Forest Meteorology*. **306**, article no.: 108448.

Ohkubo, S., Hirano, T. and Kusin, K. 2021b. Influence of fire and drainage on evapotranspiration in a degraded peat swamp forest in Central Kalimantan, Indonesia. *Journal of Hydrology*. **603**, article no.: 126906.

Ota, T., Kusin, K., Kilonzi, F.M., Usup, A., Moji, K. and Kobayashi, S. 2020. Sustainable Financing for Payment for Ecosystem Services (PES) to Conserve Peat Swamp Forest Through Enterprises Based on Swiftlets' Nests: An Awareness Survey in Central Kalimantan, Indonesia. *Small-scale Forestry*. **19**(4), pp.521–539.

Page, S.E. and Baird, A.J. 2016. Peatlands and Global Change: Response and Resilience. *Annual Review of Environment and Resources*. **41**, pp.35–57.

Page, S.E., Hosciło, A., Wösten, H., Jauhiainen, J., Silvius, M., Rieley, J., Ritzema, H., Tansey, K., Graham, L., Vasander, H. and Limin, S. 2009. Restoration Ecology of Lowland Tropical Peatlands in Southeast Asia: Current Knowledge and Future Research Directions. *Ecosystems*. **12**(6), pp.888–905.

Page, S.E., Rieley, J. and Banks, C. 2011. Global and regional importance of the tropical peatland carbon pool. *Global Change Biology*. **17**(2), pp.798–818.

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

Painter, S., Başağaoğlu, H. and Liu, A. 2008. Robust Representation of Dry Cells in Single-Layer MODFLOW Models. *Ground Water*. **46**(6), pp.873–881.

Parish, F., Lew, S.Y. (Serena) and Mohd Hassan, A.H. 2021. National Strategies on Responsible Management of Tropical Peatland in Malaysia *In*: M. Osaki, N. Tsuji, N. Foead and J. Rieley, eds. *Tropical Peatland Eco-management*. Springer Singapore, pp.677–723.

Payne, R.J., Anderson, A.R., Sloan, T., Gilbert, P., Newton, A., Ratcliffe, J., Mauquoy, D., Jessop, W. and Andersen, R. 2018. The future of peatland forestry in Scotland: balancing economics, carbon and biodiversity. *Scottish Forestry*. **72**(1), pp.34–40.

Peacock, M., Jones, T.G., Airey, B., Johncock, A., Evans, C.D., Lebron, I., Fenner, N. and Freeman, C. 2015. The effect of peatland drainage and rewetting (ditch blocking) on extracellular enzyme activities and water chemistry. *Soil Use and Management*. **31**(1), pp.67–76.

Planas-Clarke, A.M., Chimner, R.A., Hribljan, J.A., Lilleskov, E.A. and Fuentealba, B. 2020. The effect of water table levels and short-term ditch restoration on mountain peatland carbon cycling in the Cordillera Blanca, Peru. *Wetlands Ecology and Management*. **28**, pp.51–69.

Price, J.S., Heathwaite, A.L. and Baird, A.J. 2003. Hydrological processes in abandoned and restored peatlands: An overview of management approaches. *Wetlands*

Ecology and Management. **11**, pp.65–83.

Putra, E.I. and Hayasaka, H. 2011. The effect of the precipitation pattern of the dry season on peat fire occurrence in the Mega Rice Project area, Central Kalimantan, Indonesia. *Tropics*. **19**(4), pp.145–156.

Putra, S.S., Holden, J. and Baird, A.J. 2021. The effects of ditch dams on water-level dynamics in tropical peatlands. *Hydrological Processes*. **35**(5), article no.: e14174.

Reeve, A.S., Evensen, R., Glaser, P.H., Siegel, D.I. and Rosenberry, D. 2006. Flow path oscillations in transient ground-water simulations of large peatland systems. *Journal of Hydrology*. **316**(1–4), pp.313–324.

Republic of Indonesia Government 2016. *Indonesian Government Regulation No. 57 Year 2016 about Peatland Ecosystem Protection and Management*. Jakarta, Indonesia: Ministry of State Secretariat, Indonesia.

Rezanezhad, F., Price, J.S., Quinton, W.L., Lennartz, B., Milojevic, T. and Van Cappellen, P. 2016. Structure of peat soils and implications for water storage, flow and solute transport: A review update for geochemists. *Chemical Geology*. **429**, pp.75–84.

Ritzema, H., Limin, S., Kusin, K., Jauhiainen, J. and Wösten, H. 2014. Canal blocking strategies for hydrological restoration of degraded tropical peatlands in Central Kalimantan, Indonesia. *CATENA*. **114**, pp.11–20.

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. and Vriesendorp, C.F. 2017. Threats to intact tropical peatlands and opportunities for their conservation. *Conservation Biology*. **31**(6), pp.1283–1292.

Sari, A.P., Dohong, A. and Wardhana, B. 2021. Innovative Financing for Peatland Restoration in Indonesia *In*: R. Djalante, J. Jupesta and E. Aldrian, eds. *Climate Change Research, Policy and Actions in Indonesia*. Springer Cham, pp.247–264.

Schimelpfenig, D.W., Cooper, D.J. and Chimner, R.A. 2014. Effectiveness of Ditch Blockage for Restoring Hydrologic and Soil Processes in Mountain Peatlands. *Restoration Ecology*. **22**(2), pp.257–265.

Shantz, M.A. and Price, J.S. 2006. Characterization of surface storage and runoff patterns following peatland restoration, Quebec, Canada. *Hydrological Processes*. **20**(18), pp.3799–3814.

Sinclair, A.L., Graham, L.L.B., Putra, E.I., Saharjo, B.H., Applegate, G., Grover, S.P. and Cochrane, M.A. 2020. Effects of distance from canal and degradation history on peat bulk density in a degraded tropical peatland. *Science of The Total Environment*. **699**, article no.: 134199.

Supari, Tangang, F., Salimun, E., Aldrian, E., Sopaheluwakan, A. and Juneng, L. 2018. ENSO modulation of seasonal rainfall and extremes in Indonesia. *Climate Dynamics*. **51**(7–8), pp.2559–2580.

Suryadiputra, I.N.N., Dohong, A., Waspodo, R.S.B., Muslihat, L., Lubis, I.R., Hasudungan, F. and Wibisono, I.T.C. 2005. *A guide to the blocking of canals and ditches in conjunction with the community. Climate change, forests and peatlands in Indonesia project*. [Online]. Bogor, Indonesia: Wetlands International – Indonesia Programme and Wildlife Habitat Canada. Available from: https://www.researchgate.net/publication/322525860.

Susilo, G.E., Yamamoto, K., Imai, T., Inoue, T., Takahashi, H., Ishii, Y., Fukami, H., Koizumi, K. and Kusin, K. 2013. Effect of Canal Damming on the Surface Water Level Stability in the Tropical Peatland Area. *Journal of Water and Environment Technology*. **11**(4), pp.263–274.

Susilo, G.E., Yamamoto, K., Imai, T., Ishii, Y., Fukami, H. and Sekine, M. 2013. The effect of ENSO on rainfall characteristics in the tropical peatland areas of Central Kalimantan, Indonesia. *Hydrological Sciences Journal*. **58**(3), pp.539–548.

Tan, Z.D., Lupascu, M. and Wijedasa, L.S. 2021. Paludiculture as a sustainable land use alternative for tropical peatlands: A review. *Science of The Total Environment*. **753**, article no.: 142111.

Tanneberger, F., Schröder, C., Hohlbein, M., Lenschow, † Uwe, Permien, T., Wichmann, S. and Wichtmann, W. 2020. Climate Change Mitigation through Land Use on Rewetted Peatlands – Cross-Sectoral Spatial Planning for Paludiculture in Northeast Germany. *Wetlands*. **40**(6), pp.2309–2320.

Tsuji, N., Kato, T., Osaki, M., Sulaiman, A., Ajie, G.S., Kimura, K., Hamada, Y., Shigenaga, Y., Hirose, K. and Silsigia, S. 2021. Evaluation of Eco-Management of Tropical Peatlands *In*: M. Osaki, N. Tsuji, N. Foead and J. Rieley, eds. *Tropical Peatland Eco-management*. Springer Singapore, pp.163–196.

Tsuji, N., Kimura, K., Sulaiman, A., Hirano, T., Takahashi, H. and Osaki, M. 2019. Carbon and Water Interaction Model in Tropical Peatland. *International Journal of Environmental Sciences & Natural Resources*. **22**(5), article no.: 556099.

Uda, S.K., Hein, L. and Adventa, A. 2020. Towards better use of Indonesian peatlands with paludiculture and low-drainage food crops. *Wetlands Ecology and*

Management. **28**(3), pp.509–526.

Urzainki, I., Laurén, A., Palviainen, M., Haahti, K., Budiman, A., Basuki, I., Netzer, M. and Hökkä, H. 2020. Canal blocking optimization in restoration of drained peatlands. *Biogeosciences*. **17**(19), pp.4769–4784.

Ward, C., Stringer, L.C., Warren-Thomas, E., Agus, F., Crowson, M., Hamer, K., Hariyadi, B., Kartika, W.D., Lucey, J., McClean, C., Nurida, N.L., Petorelli, N., Pratiwi, E., Saad, A., Andriyani, R., Ariani, T., Sriwahyuni, H. and Hill, J.K. 2021. Smallholder perceptions of land restoration activities: rewetting tropical peatland oil palm areas in Sumatra, Indonesia. *Regional Environmental Change*. **21**, article no.: 1.

Warren, M., Hergoualc'h, K., Kauffman, J.B., Murdiyarso, D. and Kolka, R. 2017. An appraisal of Indonesia's immense peat carbon stock using national peatland maps: uncertainties and potential losses from conversion. *Carbon Balance and Management*. **12**, article no.: 12.

Wichmann, S., Prager, A. and Gaudig, G. 2017. Establishing Sphagnum cultures on bog grassland, cut-over bogs, and floating mats: procedures, costs and area potential in Germany. *Mires and Peat*. **20**, pp.1–19.

Wijedasa, L.S., Sloan, S., Page, S.E., Clements, G.R., Lupascu, M. and Evans, T.A. 2018. Carbon emissions from South-East Asian peatlands will increase despite emission-reduction schemes. *Global Change Biology*. **24**(10), pp.4598–4613.

Wijedasa, L.S., Vernimmen, R., Page, S.E., Mulyadi, D., Bahri, S., Randi, A., Evans, T.A., Lasmito, Priatna, D., Jensen, R.M. and Hooijer, A. 2020. Distance to forest, mammal and bird dispersal drive natural regeneration on degraded tropical peatland. *Forest Ecology and Management*. **461**, article no.: 117868.

WMO 2014a. *El Niño/Southern Oscillation, WMO-No. 1145*. [Online]. Geneva, Switzerland. Available from: https://library.wmo.int/index.php?lvl=notice_display&id=16747#.YNJmLEwo9hE.

WMO 2012. *The WMO El Niño/La Niña Update 10 February 2012*. [Online]. Geneva, Switzerland. Available from: https://library.wmo.int/doc_num.php?explnum_id=5008.

WMO 2014b. *The WMO El Niño/La Niña Update 15 April 2014*. [Online]. Geneva, Switzerland. Available from: https://reliefweb.int/report/world/wmo-el-ni-ola-ni-update-15-april-2014.

WMO 2016. *The WMO El Niño/La Niña Update 18 February 2016*. [Online]. Geneva, Switzerland. Available from: https://reliefweb.int/report/world/wmo-el-ni-ola-ni-update-18-february-2016.

Wösten, H., Van Den Berg, J., Van Eijk, P., Gevers, G.J.M., Giesen, W.B.J.T., Hooijer, A., Idris, A., Leenman, P.H., Rais, D.S., Siderius, C., Silvius, M.J., Suryadiputra, N. and Wibisono, I.T. 2006. Interrelationships between Hydrology and Ecology in Fire Degraded Tropical Peat Swamp Forests. *International Journal of Water Resources Development*. **22**(1), pp.157–174.

Wösten, H., Clymans, E., Page, S.E., Rieley, J.O. and Limin, S.H. 2008. Peat–water interrelationships in a tropical peatland ecosystem in Southeast Asia. *CATENA*. **73**(2), pp.212–224.

Xu, J., Morris, P.J., Liu, J. and Holden, J. 2018. PEATMAP: Refining estimates of global peatland distribution based on a meta-analysis. *CATENA*. **160**, pp.134–140.

Young, D.M., Baird, A.J., Morris, P.J. and Holden, J. 2017. Simulating the long-term impacts of drainage and restoration on the ecohydrology of peatlands. *Water Resources Research*. **53**(8), pp.6510–6522.

Zhou, Z., Li, Z., Waldron, S. and Tanaka, A. 2019. InSAR Time Series Analysis of L-Band Data for Understanding Tropical Peatland Degradation and Restoration. *Remote Sensing*. **11**(21), article no.: 2592.

Chapter V

Synthesis and conclusions

This chapter synthesises the findings of the three results chapters. It then examines the key factors in hydrological restoration, followed by a discussion on the plausibility of keeping the restored peatland wet, before outlining the implications for peatland restoration. In this chapter, I also note limitations of this study and suggest areas for future study.

5.1 Overview of the research findings

Using a field monitoring campaign at three nearby peatland sites in Sebangau, Chapter II showed that hydrological processes differ across tropical peatlands with different restoration conditions. In the dry season, water tables at all sites were deeper than 40 cm from the peat surface, while the ditches were dry and had no flow. The dry season water-table drawdown rates in response to evapotranspiration were 9.3 mm day^{-1} at Forested, 9.6 mm day^{-1} at Blocked, but 12.7 mm day^{-1} at Drained. In the wet periods, ditches kept imposing hydraulic gradients, in which the interquartile range of hydraulic gradients between the lowest ditch outlet and the farthest well from ditches was from 3.7×10^{-4} to 7.8×10^{-4} m m^{-1} at Blocked, and from 1.9×10^{-3} to 2.6×10^{-3} m m^{-1} at Drained. In the wet season, water tables at wells in the Forested site were deeper than 40 cm from surface for 45% of the time. Considering these results, I recommended a water-table depth limit policy that reflects natural seasonal and spatial dynamics in tropical peatlands rather than the current static 40-cm depth limit.

In Chapter III, a detailed analysis of the water-table responses to individual rainfall events at the Sebangau sites provided evidence that drainage and ditch dam can affect peatland water-table responses to storms. Water-table responses to storms had strong dependency on the initial water-table condition, the depth-duration patterns of the storm, and the position with respect to ditches, which explain why the responses were very different between intact, drained and ditch-dammed systems. In the Forested site, the mean of the post-storm water-level drawdown speed (DSpeed) when the water table was deep (deeper than 50 cm below surface) was 0.039 (SD = 0.024) cm $hour^{-1}$ but it was 0.047 (SD = 0.039) cm $hour^{-1}$ when the water table was shallow (within the upper 50 cm of the peat). In the Drained/Blocked sites, DSpeed variations depended on depth, distance to ditches, and distance to the main outlet of ditches. Mean dry bulk density values were between 0.055 and 0.121 g cm^{-3} at Forested, 0.116 and 0.154 g cm^{-3} at Blocked, and between 0.106 and 0.175 at Drained. The surface K values measured at Forested (10.05 and 16.87 m day^{-1}) were the highest among all, whereas the surface K values at Blocked (average = 6.0 m day^{-1}, SD = 2.7 m day^{-1}) were generally lower than at Drained (average = 9.2 m day^{-1}, SD = 3.2 m day^{-1}). It was concluded that hydrological functioning at the restored site (with ditch dams installed five years before the study commenced) was quite different to that at the Forested site.

Numerical modelling with DigiBog_Hydro presented in Chapter IV highlighted the potential for bunds to support ditch dams in maintaining the water table near to the peatland surface in different climate conditions. The fine-scale (1 m × 1 m spatial resolution) modelling of a typical peat plot (~0.2 km^2) in Sebangau revealed that in the dry season of any ENSO scenario, the ditch-dams alone substantially helped in maintaining peatland water level up to lateral distances of 26 m (in the moderately degraded peatland, Mod-Dgr) and 12 m (in the severely degraded peatland, Sev-Dgr) from the ditch. In the dry season of an El Niño year, the bunding arrangement supported ditch dams, and the water levels were maintained up to 72 cm (in Mod-Dgr) and 69 cm (in Sev-Dgr) higher than in the no-restoration (control) condition. During the extreme-dry period of an El Niño year, the bunds reduced the number of days when the water table was deeper than 40 cm in Mod-Dgr by 50% compared to the control condition, and 73% in Sev-Dgr. The combinations of ditch dams and bunds may provide additional water storage, buffering the water table from drainage and evapotranspiration demands in dry periods. The presence of ditch dams and bunds, the type of bunds used, as well as the degree of peat degradation (peat hydrological properties) and seasonal weather patterns (net rainfall supply) affect the speed of water table recovery during the early part of the wet season.

5.2 Key factors in hydrological restoration

This study indicates three key factors to consider in the hydrological restoration of tropical peatlands: i) the impact of drainage on water-table dynamics; ii) the functioning of restoration structures in retaining water, and iii) the seasonal and climatic variations of water tables in tropical peatlands.

This study found that ditch drainage altered the water-table dynamics compared to more intact forested tropical peat. The ditches were associated with spatial variation in water tables whereby water tables near to ditches were deeper than those further away from ditches. The ditches promoted hydraulic gradients that reduced the capacity of the peatland plot to retain water, resulting in faster water-table drawdowns after storm events. The effect of drainage will be minimal if the difference between the water-level in ditches and in the peat plot is also minimal. During the transition period from the dry to wet seasons, drainage slowed down the rise of water table in the studied peatlands. In the dry season, the ditches were dry, the effect of ditches in draining the peatland was at a minimum, and the water-table drawdowns were mainly induced by evapotranspiration.

Ditch dams did act to block the flow of water in ditches, so that the water level at upstream of the dams can be maintained up to some point, and the hydraulic gradient between the peatland and ditches can be minimized. The ditch dam also diverted some of the flow so that some water may flow through the surrounding peat layers instead of over the crest of the dam. Nevertheless, the effectiveness of ditch dams was spatially limited, and also depended on rainfall inputs and flows at the outlet of the peatland plot. This finding is consistent with studies presented by Ritzema et al. (2014) and Urzainki et al. (2020) as they recognized that canal dam performance in maintaining peatland water levels varies with distance to the dam and between seasons. However, I also found that ditches were empty in the dry season even when blocked. Therefore, the dams were ineffective at this time.

Bunding is designed to pool rainfall water and retain water on the peatland surface by limiting overland flow in the bounded peatland area. As the bund has low permeability, the water stored behind the bund will leave the system either through evaporation or by infiltrating into the peat before flowing to the ditches. There is also a possibility to create controlled outlets on the bund so that the stored surface water can be used to supply other areas with deeper water tables. Field reports from the UK, continental Europe, and North America on the performance of bunds in providing extra water storage in temperate peatlands have been promising (Shantz and Price, 2006; Land and Brock, 2017; Payne et al., 2018). I conclude that bunds could be developed for use in tropical peat restoration projects.

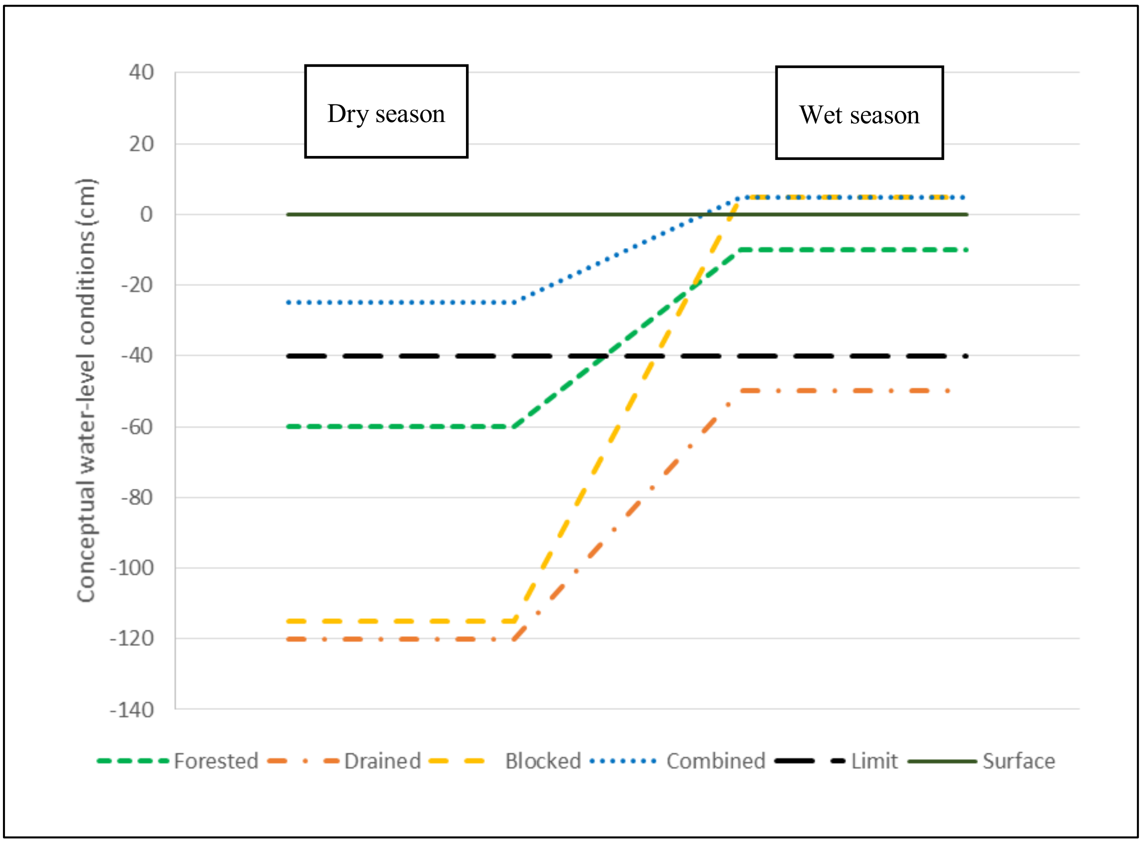

Figure 5.1 Conceptual water-table dynamics of the study peat plots in an ENSO neutral year, including the possible alterations caused by ditch dams and bunding. Blocked represents the condition of peatlands with ditch dams, whereas Combined is for peatlands with ditch dams and bunds. Limit is the 40-cm depth limit used to determine the hydrological condition of peatlands in Indonesia (President of the Republic of Indonesia, 2016).

I presented evidence that precipitation played important roles in governing the seasonal water-table dynamics in the studied peat plots. Figure 5.1 presents a conceptual summary showing the effect of rainfall conditions (dry and wet seasons) on the water tables for peatlands with different restoration conditions. In the wet periods, this study shows that the net water supply to the ditch dammed peatlands was generally enough to overcome evapotranspirative demands and drainage effects while maintaining water tables near to the peat surface (except for the area near ditches). However, a similar condition cannot be achieved in the dry periods, when the ditches were dry and rainfall supplies were limited, indicating that extra water storage is needed to keep the water table near the surface. Figure 5.1 also highlights that water tables at all sites were generally below the 40-cm limit, including those at Forested, during the dry season of an ENSO neutral year. Either the Indonesian legal limits need to be altered or extra water is needed in the dry season to ensure the water table is above the 40-cm limit. In the dry period of a year with plenty of rainfall (e.g., a La Niña year), the modelling results presented in Chapter IV showed that there were still areas with deep water tables (e.g., those near ditches). Meanwhile, the modelling results showed that ditch dams supported by bunding helped maintain dry season water tables between 69 cm and 72 cm higher than in the no-restoration condition.

5.3 Plausibility of keeping the restored peatland wet

The evidence from my study suggests that additional interventions are needed to keep the water table of the peatlands in Sebangau near to the surface. Given the uncertainty of rainfall availability and climatic conditions (Feng et al., 2013; Supari et al., 2017; IPCC, 2021; Muhammad et al., 2021), and considering the dry conditions reported by Ballhorn et al. (2009), Field et al. (2016), and Islam et al. (2018), the existence of supplemental water storage in tropical peatlands is important to avoid critical water-level conditions such as in El Niño years. Nevertheless, the aim of storing sufficient additional water for the dry season could be challenging. Therefore, this section suggests some other sorts of restoration work that might be required to support peatland restoration in the region.

Another possible approach for controlling peatland water levels is ditch infilling (Giesen and Sari, 2018; Dohong et al., 2018). Ditch infilling involves adding mature peat along the ditches, but in the field local people also add plant debris and grow hardwood vegetation on the edge of the ditches. The permeability of the infilled material may mean that ditch infilling is not as effective as ditch dams at reducing the hydraulic gradient across a peatland site as flow will preferentially move along the infilled ditch and further research into this practice is required.

In the ditch-dammed site, during wet periods, inundation of areas near to ditches needs to be minimized by slowing surface runoff. In the ditch-dammed site, in the wet period, I noted some areas in the middle of the peat plot that were not inundated, meaning that water tended to flow to the ditches rather than stay longer in the peat plot. To increase the retention time of water in tropical peatlands, revegetation efforts could be accelerated so that surface runoff can be slowed (Hoekman, 2007; Dommain et al., 2010; Evers et al., 2017; Adi et al., 2021; Apers et al., 2022), as also shown by studies from temperate peatlands (Grayson et al., 2010; Shuttleworth et al., 2019; Goudarzi et al., 2021). The bunding solution that is proposed by my study could also help retain water on peat plots, delaying water flow to the outlet of the drained peatland complex. If the water stored by bunds is not needed due to wet conditions, the surplus water can be conveyed to the drawdown zone near to ditches (e.g., gravitationally through pipes with valves) to keep water tables there nearer the surface.

The modelling work covering different El Niño conditions provided insights that bunding can support the performance of ditch dams in retaining water in the peatlands. The bunds retained rainwater in the wet period and provided extra water supply to the peatland in the dry period. The dimensions and the capacity of bunds in drained tropical peatlands can be adjusted to the drainage outflow context and the targeted water-table depth in the dry period. This study also showed that the degradation condition of the peatlands needs to be considered in designing the structures too. Moreover, the variation of evapotranspiration demands needs to be accommodated in designing the bunding area, because Hirano et al. (2015) and Ohkubo et al. (2021) recognised that higher water tables and denser vegetation in degraded tropical peatlands may lead to a higher evapotranspiration demand. Chapter IV showed that only a portion of the peat plot was required to be inundated by bunding (acts as a buffer area), and the rest of the area could be prioritized for intensive revegetation using native species. The buffer area could form as a portion of the peatland complex and consist of a series of distributed small reservoirs.

5.4 Implications of this study to the management of tropical peatlands

I suggest that the assessment of the hydrological condition of tropical peatlands needs to consider the spatial and seasonal variations of the water table, rather than just using a static benchmark (e.g., the 40-cm water-table depth limit) that has been deemed as suitable by policy makers. As the 40 cm limit does not appear to be a natural feature of undisturbed and less disturbed peatlands such as the forested site, then perhaps it is not a reasonable static rule to apply to restored peatlands. Therefore, aligned with other literature (Marttila and Kløve, 2010; Haahti et al., 2016; Tuukkanen et al., 2016; Holden et al., 2017; Vernimmen et al., 2020), this study highlighted the benefit of ditch and peat water-level monitoring for improving the quality of hydrological condition assessment in drained tropical peatlands.

The establishment of new drainage in tropical peat systems should be avoided where possible noting that even after restoration efforts, the hydrological dynamics cannot be restored at least in the short-term (several years) following ditch blocking. The drainage imposed hydraulic gradients that modified the response of the water table to storms compared to the intact peatland. The implementation of ditch dams provided benefits that were limited in time and space, and were unlikely to have altered the physical properties (e.g., bulk density and surface hydraulic conductivity) of the peat toward that of the less disturbed site.

In particular, the national climate policy in Indonesia should consider the need for extra water storage to support the sustainability of tropical peatlands. The results in Chapter III showed that peatlands may experience water deficit in the dry period of different ENSO conditions, which is hard to be remedy without additional water storage. New ways of retaining water in degraded tropical peatlands are required, but this study examined how ditch dams and bunds might work in combination. Bunding is a promising solution because it has the ability to harvest rainfall and the local people are already familiar with ridge structures.

Hydrodynamic computer models such as DigiBog_Hydro can be used to inform management and infrastructure designs for tropical peatlands. Such modelling has the potential to provide faster predictions of restoration performance than field trials. As stated in Chapter III, more peat properties data are needed to minimize the uncertainty of modelling results, and more hydrological monitoring is necessary to validate, develop, or adjust models. Modelling should be used in the future for a range of tropical peatland sites and scenarios that managers are interested in, so that the selection of restoration plans can be supported by modelling evidence.

The results of this study have relevance in other tropical peatland systems elsewhere in the world, as long as there are similarities in peat properties, drainage conditions, and seasonal characteristics. Understanding how deep dry season water tables fall in other tropical peatlands, particularly intact systems, would be very useful for designing restoration schemes and managing expectations about water-table management. The dry bulk density and hydraulic conductivity values of less disturbed peatlands in Sebangau, Kalimantan, Indonesia, were comparable to those in Panama, Brunei Darussalam, the

Congo, and Peruvian Amazonia as discussed in Chapter III and IV (Baird et al., 2017; Cobb and Harvey, 2019; Dargie et al., 2017; Kelly et al., 2014). The peat organic matter data in Chapter III indicated that the peat layers at the study sites were thicker than 2 m, which may not be the case in other forested peatlands (e.g., peatlands in the Congo, see Dargie et al. (2017)). The current and the predicted future regional climate conditions of Southeast Asia are not the same as those in Equatorial Africa or South America (Li et al., 2007; Leng et al., 2019; IPCC, 2021). Nevertheless, our modelling approaches can be implemented in other tropical peatlands and will help in understanding future water-table dynamics in response to changing environmental conditions.

I hope my research can be further disseminated and used by government and international organizations, such as Global Peatlands Initiative and International Tropical Peatlands Centre. Those organizations have direct communication networks to policy makers and legislators worldwide. By using empirical field data and numerical model results, I hope many environmental managers and law makers can undertake more informed decisions in planning and maintaining peat carbon storage.

5.5 Limitations of this study and the opportunities for further research

This study involved one drained, one blocked and one forested site as comparisons. Ideally, I would have included several replicates of each management type to enhance the representativeness and understand more about variability between sites. In addition, if resources and time had permitted it would have been more powerful to conduct a before-after-control-impact study whereby monitoring occurred before and after management interventions compared to control sites where management was not changed. In such cases revegetation, ditch blocking, and bunding programs would need to be implemented well after the hydrological monitoring has commenced. However, I had limited time and resources for my study and there is a lack of long-term monitoring of suitable resolution in tropical peatland systems. The current water-table monitoring records in the Drained/Blocked sites are temporary and therefore can not be used to do long-term water-table analysis. Nevertheless, for future research which can cover periods that capture ENSO variability, some permanent regional water-level monitoring stations have been initiated by the Indonesian government, mostly since 2018, and have been dedicated for long-term operation (Widodo et al., 2019; Yananto et al., 2021; Turmudi et al., 2022; Umarhadi et al., 2022). Further work may be required to install spatial networks of monitoring points across some individual sites to capture both the spatial and temporal variations across ENSO cycles. Site access was also a limiting factor especially for the forested site which had dense vegetation and as local people and visitors use boats and walk long distances to approach the forested site, frequent visits were challenging. Therefore, I had less spatial data on water tables from that site and relied more on automatic dipwells.

The field campaign in this study covered an ENSO neutral period. The timing and duration of the PhD study, also the challenges of field access, did not allow me to investigate water-table dynamics in more extreme conditions. However, in Chapter IV the water-table dynamics in La Niña or El Niño conditions were simulated, which was particularly useful for examining water-table transition patterns from wet to dry periods and vice versa. Further high-resolution field data on water tables would be beneficial that cover the full ENSO cycle. Mezbahuddin et al. (2015) and Apers et al. (2022) also indicated the need for more long-term hydrological data collection in tropical peatlands to enhance the validity of water-table modelling results. The forested site had somewhat different rainfall conditions to the drained site, perhaps because both sites were located 70 km from each other, within the Sebangau Peat Dome area. However, my PCA analyses controlled for these effects by considering water-table responses to individual local storm events.

Climate change may alter total rainfall, rainfall depth–duration patterns, and temperature in the study region (Li et al., 2007; Leng et al., 2019; IPCC, 2021). These potential changes may need to be considered in designing the dimensions of the restoration bunds or other restoration features. The change in the depth–duration pattern of storms may be important in the operation of bunds, for example, including determining the time when the stored water needs to be released or distributed around the area outside of the bunds. My research indicates that restoration might help in delivering peatland adaptation and mitigation against climate change. The latest IPCC report indicated that there might be a possibility of more days without rainfall and higher temperatures in dry seasons than in the current situation in the Southeast Asia region (IPCC, 2021). In that context, the combination of ditch dams and bunds will be important to maintain water storage for the dry seasons, but ditch dams alone might not be so helpful. Some studies indicate that paludiculture and revegetation might also be important to support maintain peatland wetness (Budiman et al., 2020; Mishra et al., 2021; Yuwati et al., 2021; Tata et al., 2022). However, keeping the peatland pristine might be one of the best options we can choose to minimize the effects of climate change on peatland functions (Tanneberger et al., 2020; Deshmukh et al., 2021; Glenk et al., 2021; Merten et al., 2021). Higher temperatures may result in higher evapotranspiration in peatlands (Limpens et al., 2014; Deshmukh et al., 2021). Hirano et al. (2015) and Ohkubo et al. (2021) noted that the evapotranspiration varied with water-table depth, yet my modelling study used a uniform evapotranspiration rate for each simulated condition. However, evapotranspiration rates can be made variable in the model and the modelling approach I have developed enables future users to test different climate change and land management scenarios and their interactions for tropical peatlands. Such testing could be of great benefit and is an area I propose for future research.

The modelling approaches implemented in this study did not cover the variation of peat properties in depth, overland flow, and flow in ditches. Non-uniformity of peat layers and also the variations of peat degradation condition with depth are known to occur (Anshari et al., 2010; Kurnianto et al., 2019; Kurnain, 2019; Sinclair et al., 2020); therefore, using uniform or low-detail peat properties assumptions in modelling the 2D – 3D flow across peat layers may not be representative of real-world conditions. As more field data are collected from tropical systems, it may be possible to represent differences in peat properties with depth or distance to ditches within the model. While the modelling outputs were very useful, further field data would be welcome for a wider range of tropical sites to help parameterise models of tropical peatlands and validate outputs under a range of scenarios. If possible, more water-table monitoring wells could be added at some strategic monitoring sites too, so that an enriched understanding of spatial water-table dynamics in disturbed and intact tropical peatland sites can be achieved.

Overall, I was able to develop a modelling approach that can be applied in future work in tropical peatlands (potentially also in cold and temperate peatlands) before the field implementation of restoration measures. This approach could be extremely valuable in helping to refine bunding restoration designs for maximum effectiveness. Such modelling could be further developed to help site managers and policy makers in enhancing the future design of tropical canal/ditch blocking and revegetation programmes (Dohong et al., 2017, 2018; Harrison et al., 2020). The barriers to modelling research being embedded in planning future restoration initiatives in tropical peatlands include inadequate transfer of knowledge (peatland science and best practices) to site managers, legislators, and government. Modelling results need to be translated and related to daily life experiences so that practitioners can easily understand it. However, in many countries flood models are successfully used in planning. So it should be possible to overcome barriers to use peatland models to inform restoration practice. Training and focus group discussions with local people and practitioners, allocating dedicated areas for raising public awareness to peatland conservation (e.g., RePeat area in Sebangau peatland, Kalimantan, Indonesia), and also enhancing international cooperation (e.g., through Global Peatlands Initiative (2022)) are plausible solutions to aid model integration into peatland restoration plans.

5.6 Conclusions

The water-table dynamics in three tropical peat plots varied with different restoration conditions, seasonal conditions, and spatial location within the plots. In the intact forested site, water tables fell below 40 cm depth during the dry season suggesting that the current Indonesian legislation about ensuring water tables do not fall below this value at restoration sites may need to be revised to account for natural peatland functioning. The construction of ditch dams was insufficient to restore hydrological functions in drained tropical peatlands including the responses to individual rainfall events. In the dry period, the ditches at the study sites were dry, including those at the restoration site, and peatland water tables were deep, indicating that the ditch dams did not help the peatlands to store enough water. The ditch dams appeared to accelerate water-table rise in response to rainfall events during the transition from the dry to wet periods, in comparison to the drained site, and maintained water tables near to the surface during the wet season. However, extra water storage is needed to subsidize the evapotranspirative demands in the dry period, and to slow the water-table drawdown in the drained and blocked sites. Modelling showed that surface reservoirs (e.g., an inundated bunded area) have the potential to be used to retain water in the peatlands and replenish the surrounding area in the dry periods. Therefore, rather than rely on ditch dams alone for restoration water management, I recommend managers of tropical peatland systems apply bunding and explore other methods to retain more water for the dry season in formerly drained peatlands, particularly for those more severe dry seasons that occur as part of ENSO cycles.

5.7 References

Adi, R.N., Savitri, E., Putra, P.B. and Indrajaya, Y. 2021. Water Balance of Various Peatland Typologies in Central Kalimantan. *IOP Conference Series: Earth and Environmental Science*. **874**, article no.: 012002.

Anshari, G.Z., Afifudin, M., Nuriman, M., Gusmayanti, E., Arianie, L., Susana, R., Nusantara, R.W., Sugardjito, J. and Rafiastanto, A. 2010. Drainage and land use impacts on changes in selected peat properties and peat degradation in West Kalimantan Province, Indonesia. *Biogeosciences*. **7**(11), pp.3403–3419.

Apers, S., De Lannoy, G.J.M., Baird, A.J., Cobb, A.R., Dargie, G.C., Pasquel, J., Gruber, A., Hastie, A., Hidayat, H., Hirano, T., Hoyt, A.M., Jovani-Sancho, A.J., Katimon, A., Kurnain, A., Koster, R.D., Lampela, M., Mahanama, S.P.P., Melling, L., Page, S.E., Reichle, R.H., Taufik, M., Vanderborght, J. and Bechtold, M. 2022. Tropical Peatland Hydrology Simulated With a Global Land Surface Model. *Journal of Advances in Modeling Earth Systems*. **14**(3), article no.: e2021MS002784.

Baird, A.J., Low, R., Young, D., Swindles, G.T., Lopez, O.R. and Page, S.E. 2017. High permeability explains the vulnerability of the carbon store in drained tropical peatlands. *Geophysical Research Letters*. **44**(3), pp.1333–1339.

Ballhorn, U., Siegert, F., Mason, M. and 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*. **106**(50), pp.21213–21218.

Budiman, I., Sari, E. N., Hadi, E. E., Siahaan, H., Januar, R. and Hapsari, R. D. 2020. Progress of paludiculture projects in supporting peatland ecosystem restoration in Indonesia. *Global Ecology and Conservation*. **23**, article no.: e01084.

Cobb, A.R. and Harvey, C.F. 2019. Scalar Simulation and Parameterization of Water Table Dynamics in Tropical Peatlands. *Water Resources Research*. **55**(11), pp.9351–9377.

Dargie, G.C., Lewis, S.L., Lawson, I.T., Mitchard, E.T.A., Page, S.E., Bocko, Y.E. and Ifo, S.A. 2017. Age, extent and carbon storage of the central Congo Basin peatland complex. *Nature*. **542**(7639), pp.86–90.

Deshmukh, C.S., Julius, D., Desai, A.R., Asyhari, A., Page, S.E., Nardi, N., Susanto, A.P., Nurholis, N., Hendrizal, M., Kurnianto, S., Suardiwerianto, Y., Salam, Y.W., Agus, F., Astiani, D., Sabiham, S., Gauci, V. and Evans, C.D. 2021. Conservation slows down emission increase from a tropical peatland in Indonesia. *Nature Geoscience*. **14**(7), pp.484–490.

Dohong, A., Aziz, A.A. and Dargusch, P. 2018. A review of techniques for effective tropical peatland restoration. *Wetlands*. **38**(2), pp.275–292.

Dohong, A., Cassiophea, L., Sutikno, S., Triadi, B.L., Wirada, F., Rengganis, P. and Sigalingging, L. 2017. *Modul pelatihan pembangunan infrastruktur pembasahan gambut sekat kanal berbasis masyarakat (Training Module for community based peatland rewetting infrastructure development)*. [Online]. Jakarta, Indonesia. Available from: https://brg.go.id/files/Publikasi/Modul_Sekat_Kanal_Berbasis_Masyarakat_Final_30082017.pdf.

Dommain, R., Couwenberg, J. and Joosten, H. 2010. Hydrological self-regulation of domed peatlands in south-east Asia and consequences for conservation and restoration. *Mires and Peat*. **6**, pp.1–17.

Evers, S., Yule, C.M., Padfield, R., O'Reilly, P. and Varkkey, H. 2017. Keep wetlands wet: the myth of sustainable development of tropical peatlands - implications for policies and management. *Global Change Biology*. **23**(2), pp.534–549.

Feng, X., Porporato, A. and Rodriguez-Iturbe, I. 2013. Changes in rainfall seasonality in the tropics. *Nature Climate Change*. **3**(9), pp.811–815.

Field, R.D., van der Werf, G.R., Fanin, T., Fetzer, E.J., Fuller, R., Jethva, H., Levy, R., Livesey, N.J., Luo, M., Torres, O. and Worden, H.M. 2016. Indonesian fire activity and smoke pollution in 2015 show persistent nonlinear sensitivity to El Niño-induced drought. *Proceedings of the National Academy of Sciences*. **113**(33), pp.9204–9209.

Giesen, W. and Sari, E.N.N. 2018. *Tropical Peatland Restoration Report: the Indonesian Case*. [Online]. Jambi, Indonesia. Available from: https://www.researchgate.net/publication/323676663_Tropical_Peatland_Restoration_Report_the_Indonesian_case.

Glenk, K., Faccioli, M., Martin-Ortega, J., Schulze, C. and Potts, J. 2021. The opportunity cost of delaying climate action: Peatland restoration and resilience to climate change. *Global Environmental Change*. **70**, article no.: 102323.

Global Peatlands Initiative 2022. UNFCCC COP26 Global Peatlands Pavilion Summary

Report. [Online]. [Accessed 20 June 2022]. Glasgow, UK. Available from: https://www.iucn-uk-peatlandprogramme.org/events/peatland-pavilion-cop26.

Goudarzi, S., Milledge, D.G., Holden, J., Evans, M.G., Allott, T.E.H., Shuttleworth, E.L., Pilkington, M. and Walker, J. 2021. Blanket Peat Restoration: Numerical Study of the Underlying Processes Delivering Natural Flood Management Benefits. *Water Resources Research*. **57**(4), article no.: e2020WR029209.

Grayson, R., Holden, J. and Rose, R. 2010. Long-term change in storm hydrographs in response to peatland vegetation change. *Journal of Hydrology*. **389**(3–4), pp.336–343.

Haahti, K., Warsta, L., Kokkonen, T., Younis, B.A. and Koivusalo, H. 2016. Distributed hydrological modeling with channel network flow of a forestry drained peatland site. *Water Resources Research*. **52**(1), pp.246–263.

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. and Veen, F.J.F. 2020. Tropical forest and peatland conservation in Indonesia: Challenges and directions. *People and Nature*. **2**(1), pp.4–28.

Hirano, T., Kusin, K., Limin, S. and Osaki, M. 2015. Evapotranspiration of tropical peat swamp forests. *Global Change Biology*. **21**(5), pp.1914–1927.

Hoekman, D.H. 2007. Satellite radar observation of tropical peat swamp forest as a tool for hydrological modelling and environmental protection. *Aquatic Conservation: Marine and Freshwater Ecosystems*. **17**(3), pp.265–275.

Holden, J., Green, S.M., Baird, A.J., Grayson, R.P., Dooling, G.P., Chapman, P.J., Evans, C.D., Peacock, M. and Swindles, G. 2017. The impact of ditch blocking on the hydrological functioning of blanket peatlands. *Hydrological Processes*. **31**(3), pp.525–539.

IPCC 2021. Summary for Policymakers *In*: V. Masson-Delmotte, P. Zhai, A. Pirani, S. L. Connors, C. Péan, S. Berger, N. Caud, Y. Chen, L. Goldfarb, M. I. Gomis, M. Huang, K. Leitzell, E. Lonnoy, J. B. R. Matthews, T. K. Maycock, T. Waterfield, O. Yelekçi, R. Yu and B. Zhou, eds. *Climate Change 2021: The Physical Science Basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change*. [Online]. Cambridge, UK:

Cambridge University Press. In Press., pp.1–40. Available from: https://www.ipcc.ch/report/ar6/wg1/.

Islam, M., Chan, A., Ashfold, M., Ooi, C. and Azari, M. 2018. Effects of El-Niño, Indian Ocean Dipole, and Madden-Julian Oscillation on Surface Air Temperature and Rainfall Anomalies over Southeast Asia in 2015. *Atmosphere*. **9**(9), article no.: 352.

Kelly, T.J., Baird, A.J., Roucoux, K.H., Baker, T.R., Honorio Coronado, E.N., Ríos, M. and Lawson, I.T. 2014. The high hydraulic conductivity of three wooded tropical peat swamps in northeast Peru: measurements and implications for hydrological function. *Hydrological Processes*. **28**(9), pp.3373–3387.

Kurnain, A. 2019. Hydrophysical properties of ombrotrophic peat under drained peatlands. *International Agrophysics*. **33**(3), pp.277–283.

Kurnianto, S., Selker, J., Boone Kauffman, J., Murdiyarso, D. and Peterson, J.T. 2019. The influence of land-cover changes on the variability of saturated hydraulic conductivity in tropical peatlands. *Mitigation and Adaptation Strategies for Global Change*. **24**(4), pp.535–555.

Land, D. and Brock, A. 2017. *Peatland Restoration Advice for the Norwegian Environment Agency*. [Online]. London, UK. Available from: https://www.statsforvalteren.no/contentassets/c836cde50e1e448ca6a1897ff76531fe/peatland-restoration-advice-norwegian-ea-17oct17-djl.pdf.

Leng, L.Y., Ahmed, O.H. and Jalloh, M.B. 2019. Brief review on climate change and tropical peatlands. *Geoscience Frontiers*. **10**(2), pp.373–380.

Li, W., Dickinson, R.E., Fu, R., Niu, G.-Y., Yang, Z.-L. and Canadell, J.G. 2007. Future precipitation changes and their implications for tropical peatlands. *Geophysical Research Letters*. **34**(1), article no.: L01403.

Limpens, J., Holmgren, M., Jacobs, C.M.J., Van der Zee, S.E.A.T.M., Karofeld, E. and Berendse, F. 2014. How Does Tree Density Affect Water Loss of Peatlands? A Mesocosm Experiment. *PLoS ONE*. **9**(3), article no.: e91748.

Marttila, H. and Kløve, B. 2010. Managing runoff, water quality and erosion in peatland forestry by peak runoff control. *Ecological Engineering*. **36**(7), pp.900–911.

Merten, J., Nielsen, J. Ø. and Faust, H. 2021. Climate change mitigation on tropical peatlands: a triple burden for smallholder farmers in Indonesia. *Global Environmental Change*. **71**, article no.: 102388.

Mishra, S., Page, S. E., Cobb, A. R., Lee, J. S. H., Jovani-Sancho, A. J., Sjögersten, S., Jaya, A., Aswandi, A. and Wardle, D. A. 2021. Degradation of Southeast Asian tropical peatlands and integrated strategies for their better management and restoration. *Journal of Applied Ecology*. **58**(7), pp.1370-1387.

Mezbahuddin, M., Grant, R.F. and Hirano, T. 2015. How hydrology determines seasonal and interannual variations in water table depth, surface energy exchange, and water stress in a tropical peatland: Modeling versus measurements. *Journal of Geophysical Research: Biogeosciences*. **120**(11), pp.2132–2157.

Muhammad, F.R., Lubis, S.W. and Setiawan, S. 2021. Impacts of the Madden–Julian oscillation on precipitation extremes in Indonesia. *International Journal of Climatology*. **41**(3), pp.1970–1984.

Ohkubo, S., Hirano, T. and Kusin, K. 2021. Influence of fire and drainage on evapotranspiration in a degraded peat swamp forest in Central Kalimantan, Indonesia. *Journal of Hydrology*. **603**, article no.: 126906.

Payne, R.J., Anderson, A.R., Sloan, T., Gilbert, P., Newton, A., Ratcliffe, J., Mauquoy, D., Jessop, W. and Andersen, R. 2018. The future of peatland forestry in Scotland: balancing economics, carbon and biodiversity. *Scottish Forestry*. **72**(1), pp.34–40.

President of the Republic of Indonesia 2016. *President Regulation No.1 Year 2016 on Peatland Restoration Agency (Badan Restorasi Gambut)*. Indonesia.

Ritzema, H., Limin, S., Kusin, K., Jauhiainen, J. and Wösten, H. 2014. Canal blocking strategies for hydrological restoration of degraded tropical peatlands in Central Kalimantan, Indonesia. *CATENA*. **114**, pp.11–20.

Shantz, M.A. and Price, J.S. 2006. Characterization of surface storage and runoff patterns following peatland restoration, Quebec, Canada. *Hydrological Processes*. **20**(18), pp.3799–3814.

Shuttleworth, E.L., Evans, M.G., Pilkington, M., Spencer, T., Walker, J., Milledge, D. and Allott, T.E.H. 2019. Restoration of blanket peat moorland delays stormflow from hillslopes and reduces peak discharge. *Journal of Hydrology X*. **2**, article no.: 100006.

Sinclair, A.L., Graham, L.L.B., Putra, E.I., Saharjo, B.H., Applegate, G., Grover, S.P. and Cochrane, M.A. 2020. Effects of distance from canal and degradation history on peat bulk density in a degraded tropical peatland. *Science of The Total Environment*. **699**, article no.: 134199.

Supari, Tangang, F., Juneng, L. and Aldrian, E. 2017. Observed changes in extreme temperature and precipitation over Indonesia. *International Journal of*

Climatology. **37**(4), pp.1979–1997.

Tanneberger, F., Schröder, C., Hohlbein, M., Lenschow, † Uwe, Permien, T., Wichmann, S. and Wichtmann, W. 2020. Climate Change Mitigation through Land Use on Rewetted Peatlands – Cross-Sectoral Spatial Planning for Paludiculture in Northeast Germany. *Wetlands*. **40**(6), pp.2309–2320.

Tata, H. L., Nuroniah, H. S., Ahsania, D. A., Anggunira, H., Hidayati, S. N., Pratama, M., Istomo, I., Chimner, R. A., van Noordwijk, M. and Kolka, R. 2022. Flooding tolerance of four tropical peatland tree species in a nursery trial. *PloS one*. **17**(4), article no.: e0262375.

Turmudi, T., Suwarno, Y., Munajati, S. L. and Suryanta, J. 2021. Spatial modeling in sustainable peat land management based on Peat Hydrological Units (KHG). *Jurnal Pengelolaan Sumberdaya Alam dan Lingkungan (Journal of Natural Resources and Environmental Management)*. **11**(4), pp.613-620.

Tuukkanen, T., Stenberg, L., Marttila, H., Finér, L., Piirainen, S., Koivusalo, H. and Kløve, B. 2016. Erosion mechanisms and sediment sources in a peatland forest after ditch cleaning. *Earth Surface Processes and Landforms*. **41**(13), pp.1841–1853.

Umarhadi, D. A., Widyatmanti, W., Kumar, P., Yunus, A. P., Khedher, K. M., Kharrazi, A. and Avtar, R. 2022. Tropical peat subsidence rates are related to decadal LULC changes: Insights from InSAR analysis. *Science of The Total Environment*. **816**, article no.: 151561.

Urzainki, I., Laurén, A., Palviainen, M., Haahti, K., Budiman, A., Basuki, I., Netzer, M. and Hökkä, H. 2020. Canal blocking optimization in restoration of drained peatlands. *Biogeosciences*. **17**(19), pp.4769–4784.

Vernimmen, R., Hooijer, A., Mulyadi, D., Setiawan, I., Pronk, M. and Yuherdha, A.T. 2020. A New Method for Rapid Measurement of Canal Water Table Depth Using Airborne LiDAR, with Application to Drained Peatlands in Indonesia. *Water*. **12**(5), article no.: 1486.

Widodo, J., Sulaiman, A., Awaluddin, A., Riyadi, A., Nasucha, M., Perissin, D. and Sri Sumantyo, J. T. 2019. Application of SAR Interferometry Using ALOS-2 PALSAR-2 Data as Precise Method to Identify Degraded Peatland Areas Related to Forest Fire. *Geosciences*. **9**(11), article no.: 484.

Yananto, A., Sartohadi, J. and Marhaento, H. 2022. Groundwater level estimation model on peatlands using SAR Sentinel-1 data in part of Riau, Indonesia.

International Journal of Remote Sensing and Earth Sciences (IJReSES). **18**(2), pp.203-216.

Yuwati, T. W., Rachmanadi, D., Turjaman, M., Indrajaya, Y., Nugroho, H. Y. S. H., Qirom, M. A., Narendra, B.H., Winarno, B., Lestari, S., Santosa, P.B., Adi, R.N., Savitri, E., Putra, P.B., Wahyuningtyas, R.S., Prayudyaningsih, R., Halwany, W., Nasrul, B., Bastoni, B. and Mendham, D. 2021. Restoration of degraded tropical peatland in Indonesia: A review. *Land*. **10**(11), article no.: 1170.