WILEY

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

Santosa Sandy Putra[1,2] | Joseph Holden[1] | Andy J. Baird[1]

[1]water@leeds, School of Geography, University of Leeds, Leeds, UK

[2]Research Centre for Water Resources, Ministry of Public Works and Housing, Yogyakarta, Indonesia

Correspondence
Santosa Sandy Putra, Research Centre for Water Resources, Ministry of Public Works and Housing, Balai Sabo, Yogyakarta 55282, Indonesia.
Email: santosa-sp@mail.ugm.ac.id

Funding information
Indonesia Endowment Fund for Education (LPDP), Grant/Award Number: S-103/LPDP.3/2018; University of Leeds, Grant/Award Number: Wiley's Open Access Agreement

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.

KEYWORDS
drainage, hydraulic gradient, restoration, water table, wetland

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 (Harrison et al., 2020; Page et al., 2011; Posa et al., 2011; Warren et al., 2017). 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 (Dargie et al., 2017; Dommain et al., 2010; Hirano et al., 2015; Kelly et al., 2014). Many tropical peatlands have been artificially drained using ditch networks as part of site preparation for plantations or other agricultural activities (Dohong, Aziz, & Dargusch, 2017; Hooijer et al., 2012). 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; Hoyt et al., 2020; Kurnianto et al., 2019; Sinclair 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 (Hoscilo et al., 2011; Page et al., 2009), though land use and land cover are important factors that also contribute to this risk (Cattau et al., 2016; Dohong et al., 2018a; Uda et al., 2017). 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 (Agus et al., 2019; Ballhorn et al., 2009; Glauber et al., 2016; Uda et al., 2019; Wooster et al., 2018). 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).

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., 2018a; 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 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

FIGURE 1 Drain blocking dam in the study site, Sebangau, Indonesia

uses mature peat (Giesen & 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 (Harrison et al., 2020; Puspitaloka 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 (Kelly et al., 2017; Lampela et al., 2016; Roucoux et al., 2017) and ongoing human activity (Dohong et al., 2018a; Harrison et al., 2020; Wijedasa et al., 2017).

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 (Goodbrand et al., 2019; Harris et al., 2020; Holden et al., 2017), 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 (Dohong et al., 2018b; Jaenicke et al., 2010). 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 paper 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 with the Indonesian regulatory requirement of a 40 cm water-table depth limit?

2 | DATA AND METHODS

2.1 | Study sites

Three tropical peatland sites were chosen for this study, located in Sebangau National Park (SNP), Kalimantan, Indonesia (Figure 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 km² 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 (Jaenicke et al., 2008; Page et al., 2004). 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 (Itakura et al., 2016; Page et al., 2004). The mean annual temperature ranges from around 26.2 to 28.1°C (Hirano et al., 2014; Rahajoe et al., 2016).

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). 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 Re-Peat area, Tumbang Nusa Research Forest Zone, Pulang Pisau, Kalimantan, Indonesia. 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 (Lampela et al., 2017; Limin et al., 2007; Morrogh-Bernard et al., 2003). 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). 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.* (Husson et al., 2018; Page et al., 1999). 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, & Dargusch, 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 | Data collection

Hydrological monitoring was conducted at the three sites over a 6-month period between 22 August 2019 and 17 January 2020.
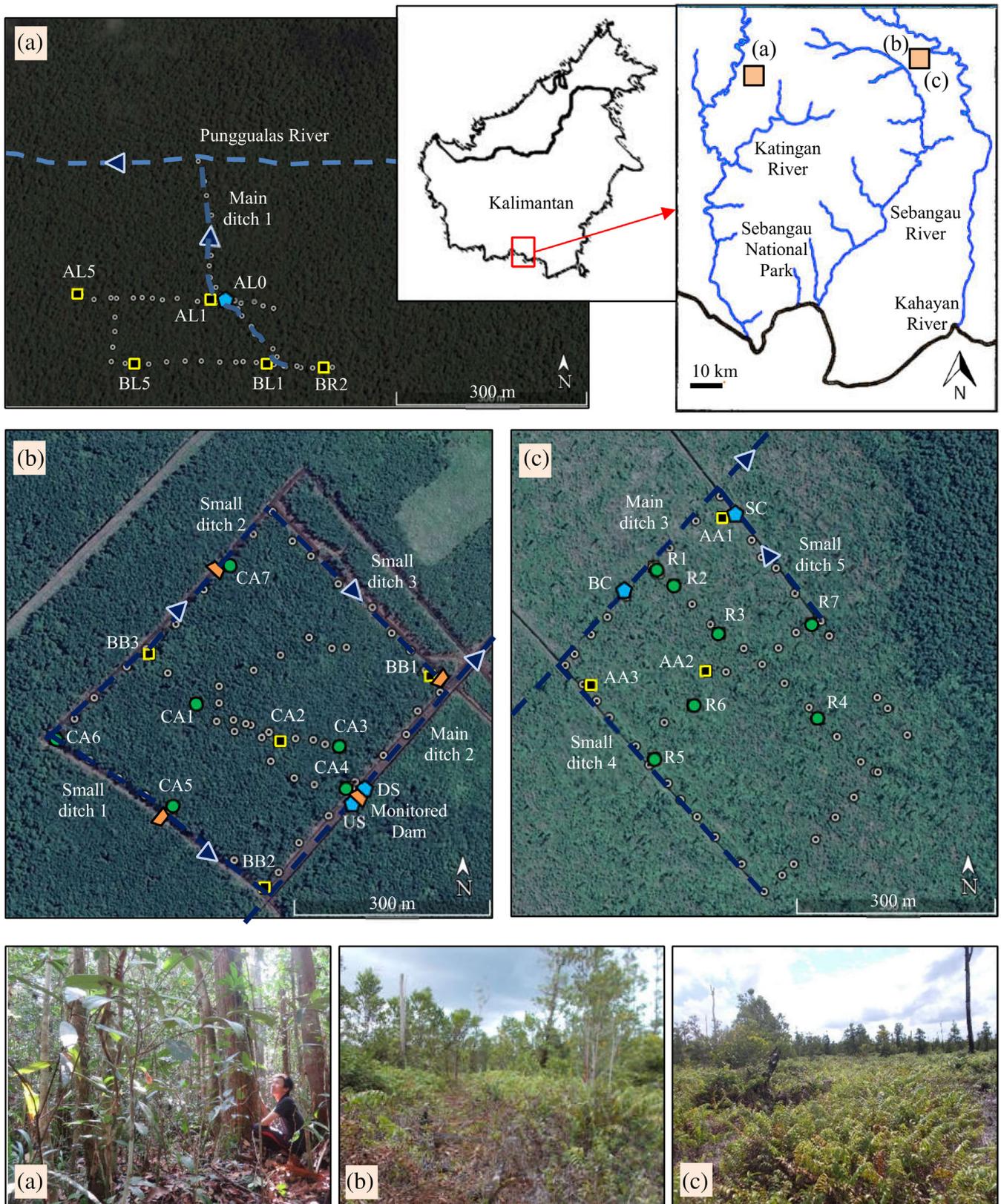
FIGURE 2 Studied sites and instrumentation in Sebangau, Indonesia, showing Forested (a), Blocked (b), and Drained (c). Squares symbolize logger wells, blue pentagons are ditch loggers, green circles are manual wells, smaller circles are levelling points, and trapezoids are dams. The main water flow directions 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

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 & 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). 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). The wells at Forested were arranged in two transects (Figure 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). 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). 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). 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).

2.3 | 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.

3 | RESULTS

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 3). 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 3a) compared to 12–14 November 2019 for Drained/Blocked (Figure 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 3a), the mean dry season PET was 2.9 (SD = 0.8) mm day^{-1}, whereas the average wet season PET was 2.1 (SD = 0.5) mm day^{-1}. At Drained/Blocked (Figure 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.

3.2 | Seasonal water-level dynamics

Figure 4 presents timeseries 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 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 1). In the wet period, sub-surface water-level fluctuations at all sites were mostly triggered by rainfall.
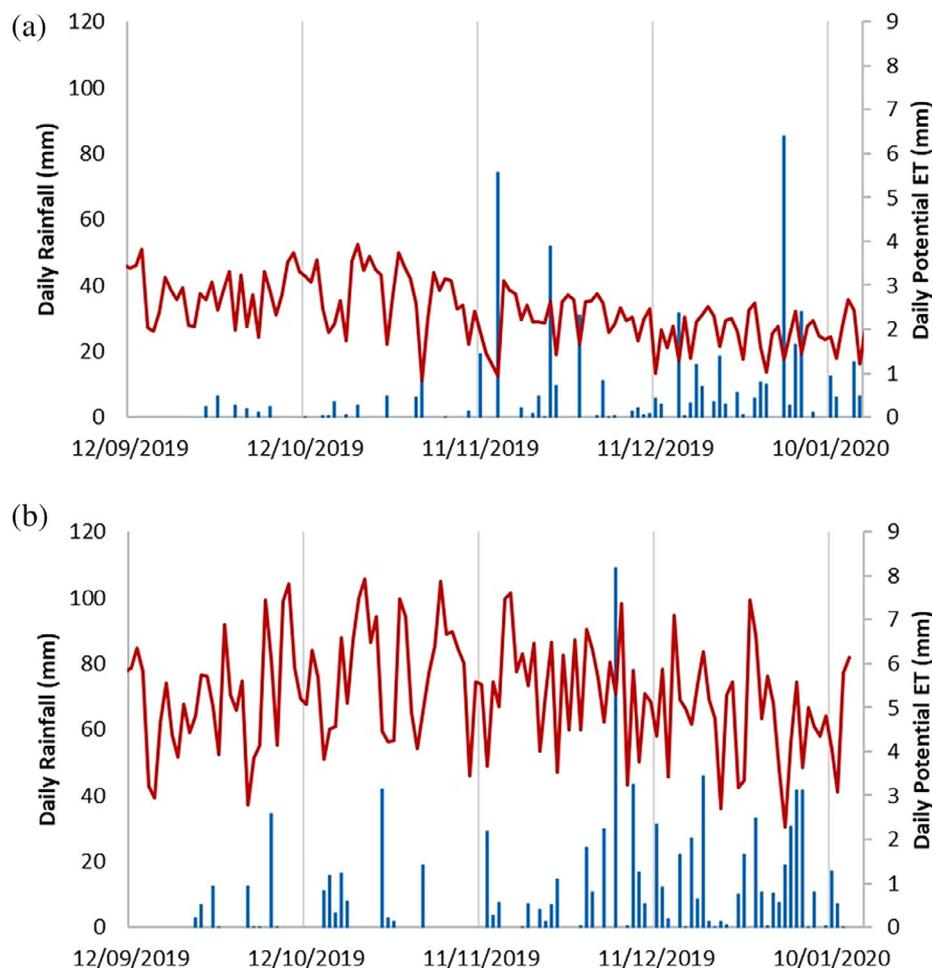
FIGURE 3 Daily rainfall and potential evapotranspiration timeseries in Sebangau Tropical Peatland: (a) Forested (b) Drained/Blocked. Red lines represent the estimated potential evapotranspiration (PET) and bars represent rainfall
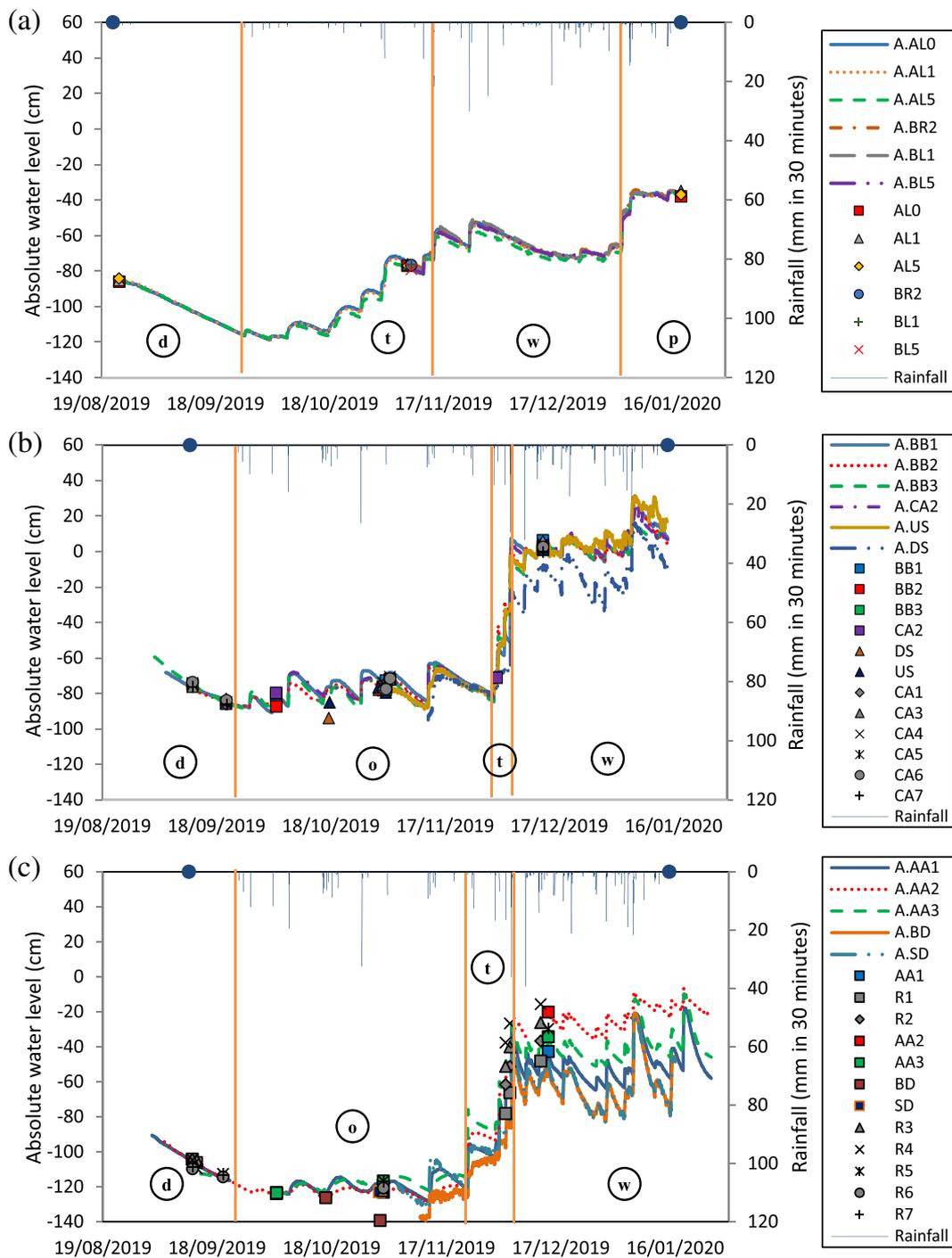
FIGURE 4 Water-level timeseries for late dry to early wet seasons at Forested (a), at Blocked (b), and at Drained (c) sites. Lines represent automatic logger data. Symbols indicate manual measurements. Bars represent rainfall data. Abbreviation 'A.' in legend indicates automatic logger, 'US' is upstream 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)

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 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 8–27 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

TABLE 1 Summary water-level statistics across seasons and spatial locations in the study sites

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

Note: 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).

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.

3.3 | Spatial water-level variations

Figure 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 6 presents water-level differences between the upstream and downstream sides of the dam at Blocked. The right-side graph in Figure 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 (10.9–18.3 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. Specifically, the spatial water-level differences were more affected by the monitoring well's average distance to the nearby side ditches. Figure 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.
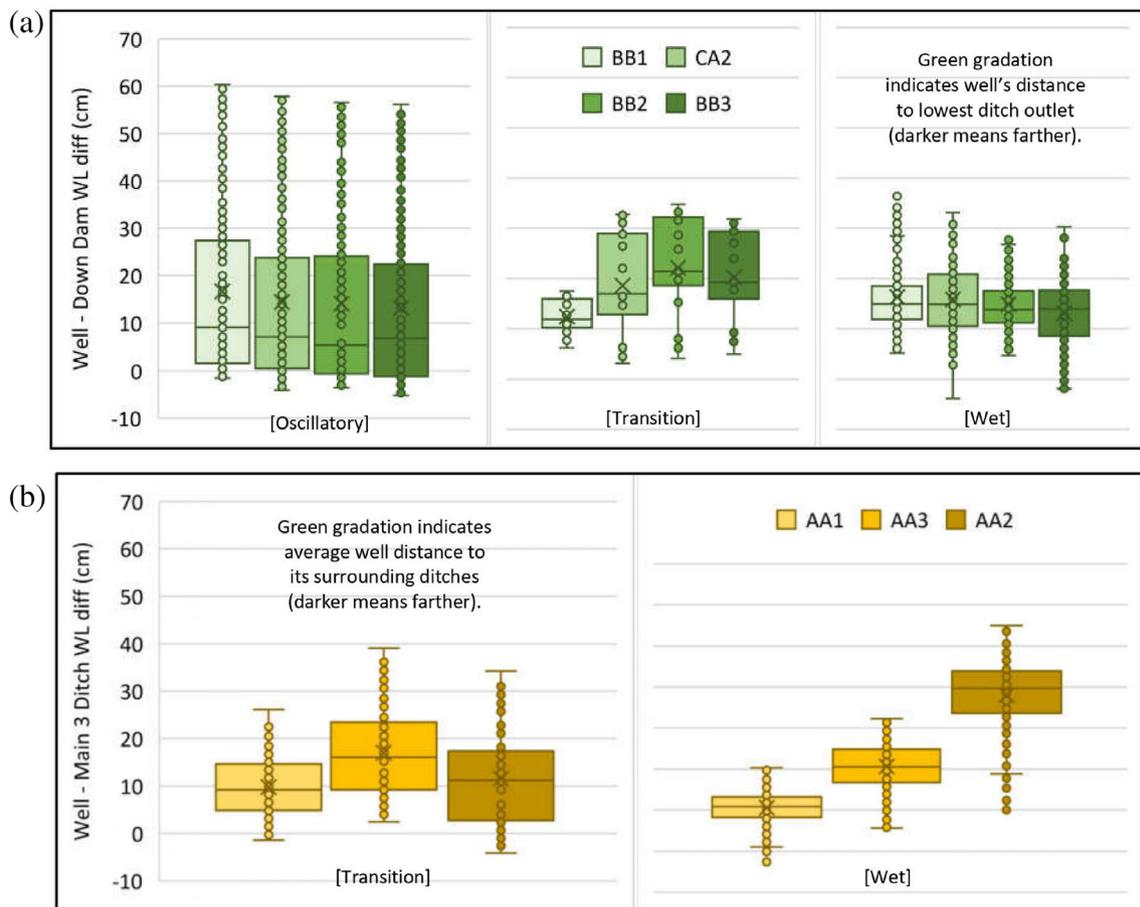
FIGURE 5 Differences between water level in the wells and water level in the lowest ditch on the site. The boxes represent monitoring wells. The lower, middle, and upper lines of the box indicate first quartile (Q1), median (Q2), and third quartile (Q3) of the data. The top and bottom ends of the whisker 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 average 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

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. The spatial differences were greater during the oscillatory period, compared to those in the other periods (Figure 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.

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 mean water-level difference upstream and downstream of the dam was 4 cm during the oscillatory period (Figure 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 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 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.

3.4 | Water-table residence times

Figure 7 presents water-table depth (i.e., relative to the peatland surface) residence times while Figure 8 shows the ditch inundation
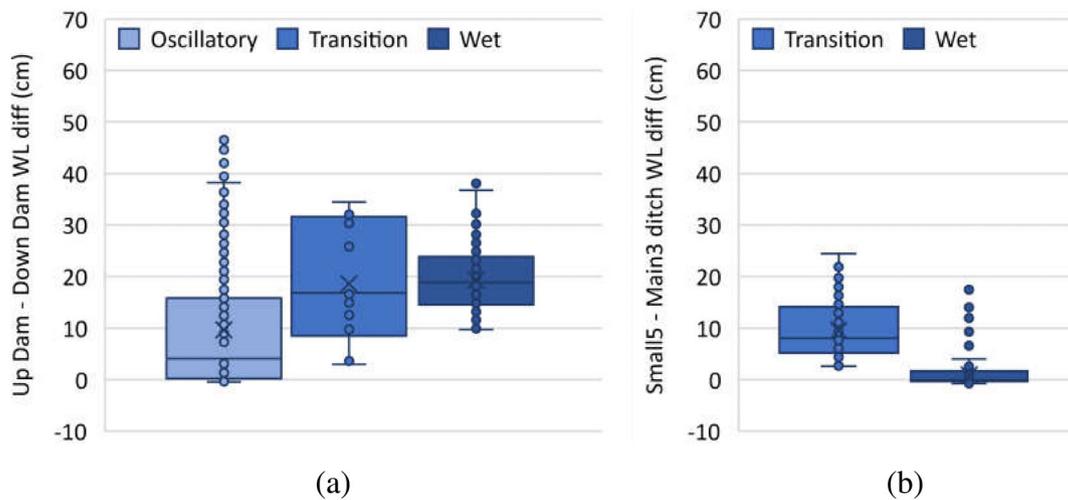
FIGURE 6 Differences in water levels in ditch sections for different time periods. The boxes represent seasonal periods. The lower, middle, and upper lines of the box indicate first quartile (Q1), median (Q2), and third quartile (Q3) of the data. The top and bottom ends of the whisker show Q3 + 0.5×|Q1–Q3| 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

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 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 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 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 average 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 7b).

The ditch water height at Forested fluctuated less than at the other sites (Figure 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 8b).

4 | DISCUSSION

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). However, our study was able to provide additional detail on temporal variability and spatial patterns relative to ditches.

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; Dohong et al., 2018a; Urzainki et al., 2020; Young et al., 2017). 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). 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
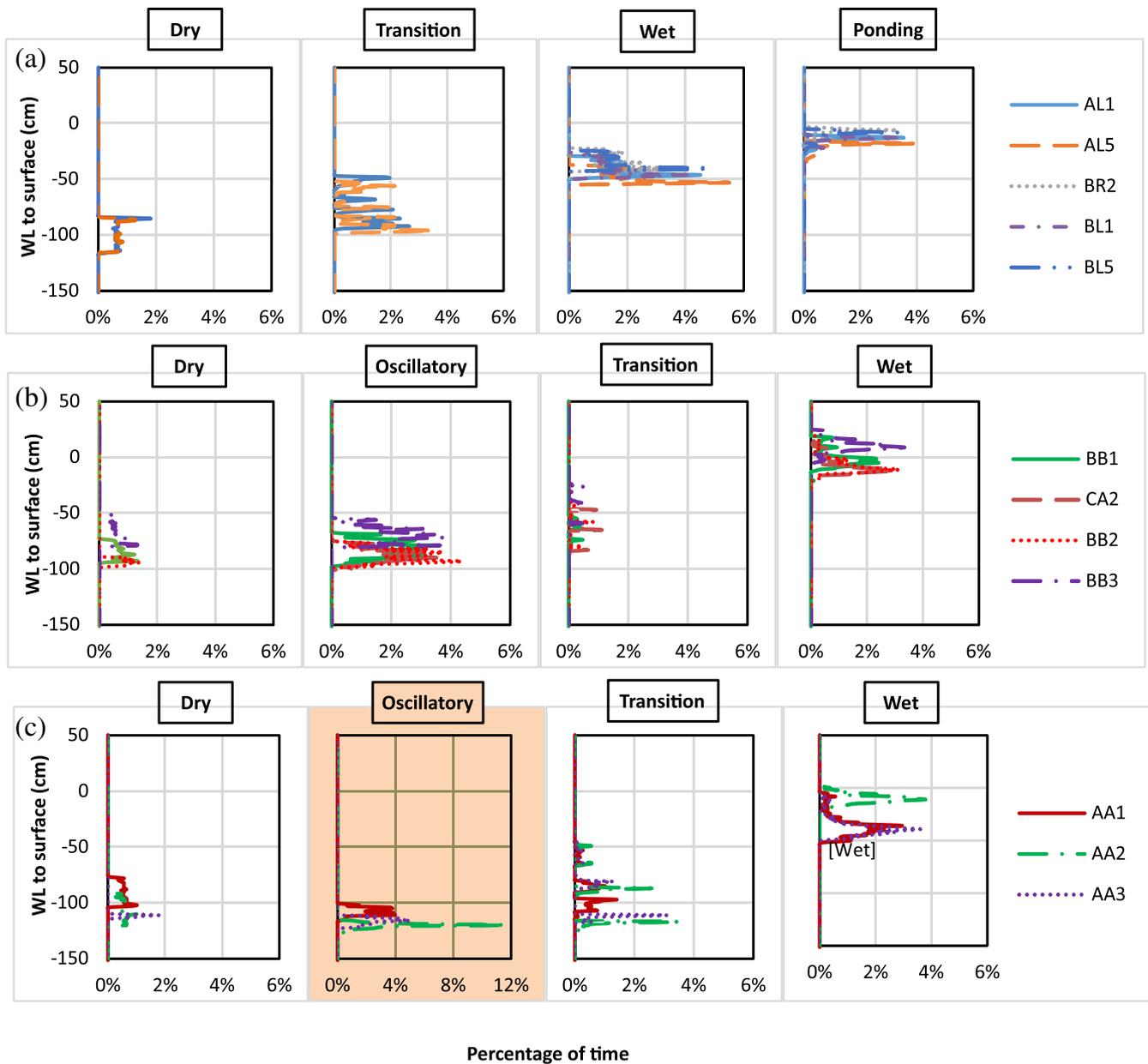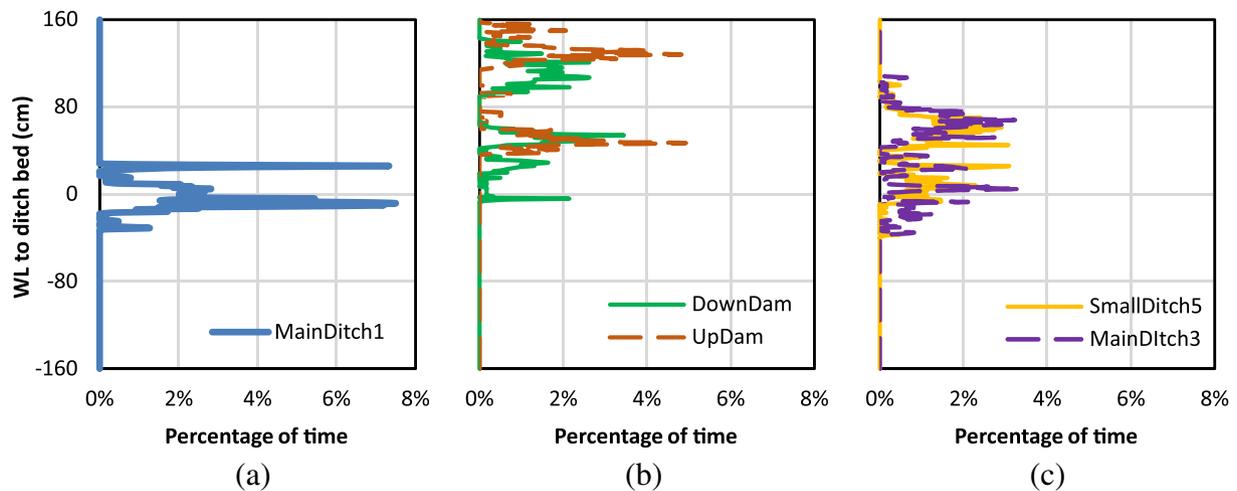
FIGURE 7 Water-level residence time for the whole study period in the three study sites. Forested data (a) are presented for the dry, transition, wet, and ponding periods, from left to right. The data from Blocked (b) and Drained (c) are for the dry, oscillatory, transition, and wet periods, left to 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

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 (Hirano et al., 2015; Kumagai et al., 2005; 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 Kasih et al., 2016; Ritzema

FIGURE 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

et al., 2014; 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 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).

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 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; 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

TABLE 2 Water-table values reported from tropical peatlands

No.	Location	Num	Mean	Min	Q1	Q2	Q3	Max	Period
1	Drained, Kalampangan, Indonesia, 50–150 m from primary canal (Lampela et al., 2017)								November 2012–July 2014
	+ Wet peat	1A	−20	−60	–	–	–	0	
	(2°20′24″S, 114°02′11″E)								
	+ Medium wet peat	1A	−30	−80	–	–	–	0	
	(2°19′18″S, 114°01′05″E)								
	+ Dry peat	1A	−50	−100	–	–	–	−20	
	(2°19′32″S, 114°00′59″E)								
2	Drained, Kalampangan, Indonesia (Ritzema et al., 2014). Transect 3	968 M	−14	−52	–	–	–	9	2006–2009
	(2°21′22.23″S, 114°3′8.30″E)								
3	Drained, Kalampangan, Indonesia (Santoso & Qirom, 2020). Transect 3	75 M	−15	−55	–	–	–	5	March 2013–September 2013
	(2°21′22.23″S, 114°3′8.30″E)								
4	Kalampangan, Palangkaraya, Indonesia (Jauhiainen et al., 2014)								May 2012–September 2012
	+ Agricultural peatland	1A	−50	–	−40	−52	−59	–	
	(Mr. Edi's field)								
	+ Degraded peatland	1A	−52	–	−47	−51	−59	–	
	(2°19′24″S, 114°1′14″E)								
5	Forest, LAHG CIMTROP, floodplain (FP) to forest transect (Lampela et al., 2017)								November 2012–July 2014
	(02°18.843′S, 113°54.159′E)								
	+ 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)	-	−40	−98	–	–	–	20	1993–2000
	(2°18′59.6″S, 113°54′28.9″E)								
7	Block A, Ex-MRP, Mawas (Sinclair et al., 2020)								2012
	(2°15′S, 114°30′E)								
	+ 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, Transect average, Mawas, Large Canal Drainage (Hooijer et al., 2014)								March 2010–December 2012
	(2°15′S, 114°30′E)								
	+ Burned peatland	220 M	−26	−63	−37	–	–	–	
	+ Degraded forest	62 M	−43	−86	−53	–	–	–	
9	Forest, Block E, Ex-MRP, Transect average, Mawas, Small Canal Drainage (Hooijer et al., 2014)	107 M	−34	−89	−49	–	–	–	March 2010–December 2012
	(2°10′S, 114°30′E)								
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 and 1.5 m, respectively								February 2018–March 2018
	(0°07′04.3″N, 109°17′42.8″E)								

TABLE 2 (Continued)

No.	Location	Num	Mean	Min	Q1	Q2	Q3	Max	Period
	+ 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	

Note: 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.

used to help store surface water across the site (Payne et al., 2018; Price et al., 2003; Wichmann et al., 2017).

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 (Putra et al., 2018; Taufik et al., 2015); therefore, it can be suggested that all of the studied sites were prone to fire in the dry season. More data are required to inform 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). 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 (Hoyt et al., 2020; Sinclair 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.

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.

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.

ORCID

Santosa Sandy Putra https://orcid.org/0000-0002-7510-5494
Joseph Holden https://orcid.org/0000-0002-1108-4831
Andy J. Baird https://orcid.org/0000-0001-8198-3229

REFERENCES

Agus, C., Azmi, F. F., Widiyatno, Ilfana, Z. R., Wulandari, D., Rachmanadi, D., Harun, M. K., & Yuwati, T. W. (2019). The impact of Forest fire on the biodiversity and the soil characteristics of tropical Peatland. In L. F. Walter, B. Jelena, & P. Richard (Eds.), *Handbook of climate change and biodiversity. Climate change management* (pp. 287–303). Springer.

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

Ballhorn, U., Siegert, F., Mason, M., & Limin, S. (2009). Derivation of burn scar depths and estimation of carbon emissions with LIDAR in Indonesian peatlands. *Proceedings of the National Academy of Sciences of the United States of America*, 106(50), 21213–21218.

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

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

Cattau, M. E., Husson, S., & 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), 473–480.

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

Davis Instruments (2006). Derived variables in Davis weather products. In *Davis Vantage Pro2 application notes* (p. 1–23). California, USA: Davis Instruments.

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

Dohong, A., Aziz, A. A., & Dargusch, P. (2018a). A review of techniques for effective tropical peatland restoration. *Wetlands*, 38(2).

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

Dohong, A., Cassiophea, L., Sutikno, S., Triadi, B. L., Wirada, F., Rengganis, P., & Sigalingging, L. (2017). Modul pelatihan pembangunan infrastruktur pembasahan gambut sekat kanal berbasis masyarakat (Training Module for community based peatland rewetting infrastructure development). Retrieved from https://brg.go.id/download/558/-Modul-Pelatihan-Pembangunan-Infrastruktur-Pembasahan-Gambut-Sekat-Kanal-Berbasis-Masyarakat.pdf

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

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

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

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

Giesen, W., & Sari, E. N. N. (2018). Tropical Peatland restoration report: The Indonesian case. Retrieved from http://dx.doi.org/10.13140/RG.2.2.30049.40808

Glauber, A. J., Moyer, S., Adriani, M., & Gunawan, I. (2016). The cost of fire: An economic analysis of Indonesia's 2015 fire crisis. Indonesia Sustainable Landscapes Knowledge Note No.1. World Bank, Jakarta. Retrieved from https://openknowledge.worldbank.org/handle/10986/23840

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

Harris, L. I., Roulet, N. T., & Moore, T. R. (2020). Drainage reduces the resilience of a boreal peatland. *Environmental Research Communications*, 2(6), 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., ... Veen, F. J. F. (2020). Tropical forest and peatland conservation in Indonesia: Challenges and directions. *People and Nature*, 2(1), 4–28.

Herawati, H., Ali Akbar, A., Farastika, D., & Azmeri, A. (2018). Water table evaluation post the construction of canal blocks on peatland in West Kalimantan, Indonesia. *MATEC Web of Conferences*, 195, 03016. Retrieved from https://www.matec-conferences.org/articles/matecconf/abs/2018/54/matecconf_icrmce2018_03016/matecconf_icrmce2018_03016.html.

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

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

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

Hooijer, A., Page, S. E., Navratil, P., Vernimmen, R., van der Vat, M., Tansey, K., Konecny, K., Siegert, F., Ballhorn, U., & 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. A summary of KFCP research results for practitioners. Retrieved from https://www.researchgate.net/publication/323616904

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

Hoscilo, A., Page, S., Tansey, K. J., & 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), 578.

Hoscilo, A., Tansey, K. J., & 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), 412–433.

Hoyt, A. M., Chaussard, E., Seppalainen, S. S., & Harvey, C. F. (2020). Widespread subsidence and carbon emissions across Southeast Asian peatlands. *Nature Geoscience*, 13(6), 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., & Schreven, S. (2018). Biodiversity of the Sebangau tropical peat swamp forest, Indonesian Borneo. *Mires and Peat*, 22, 1–50.

Ilek, A., Kucza, J., & 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, 27–34.

Inubushi, K., Furukawa, Y., Hadi, A., Purnomo, E., & 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), 603–608.

Ishii, Y., Koizumi, K., Fukami, H., Yamamoto, K., Takahashi, H., Limin, S. H., Kusin, K., Usup, A., & Susilo, G. E. (2016). Groundwater in Peatland. In M. Osaki & N. Tsuji (Eds.), *Tropical Peatland ecosystems* (1st ed., pp. 265–279) Springer Japan: Springer.

Itakura, T., Nakatsugawa, M., Sugimoto, H., & Watanabe, Y. (2016). Characteristics of watershed in Central Kalimantan. In *Tropical Peatland ecosystems* (pp. 247–263). Springer Japan.

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

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

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

Jensen, M. E., & Allen, R. G. (2016). *Evaporation, evapotranspiration, and irrigation water requirements* (2nd ed.). American Society of Civil Engineers. https://ascelibrary.org/doi/book/10.1061/9780784414057

Kasih, R. C., Simon, O., Ma'mun Ansori, M. P. P., & Wirada, F. (2016). Rewetting on Degraded Tropical Peatland by Canal Blocking Technique in Sebangau National Park, Central Kalimantan, Indonesia. Proceedings on 15th International Peat Congress 2016. Extended Abstract No: A-065. Retrieved from https://peatlands.org/document/rewetting-of-degraded-tropical-peatland-by-canal-blocking-technique-in-sebangau-national-park-central-kalimantan-indonesia/

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

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

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

Kurnianto, S., Selker, J., Boone Kauffman, J., Murdiyarso, D., & 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), 535–555.

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

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

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

Limin, S. H., Jentha, & Ermiasi, Y. (2007). History of the development of tropical Peatland in Central Kalimantan, Indonesia. *Tropics*, 16(3), 291–301.

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

Mackin, F., Barr, A., Rath, P., Eakin, M., Ryan, J., Jeffrey, R., & Fernandez Valverde, F. (2017). Best practice in raised bog restoration in Ireland. Irish Wildlife Manuals, No. 99. National Parks and Wildlife Service, Department of Culture, Heritage and the Gaeltacht, Ireland. Retrieved from https://www.npws.ie/sites/default/files/publications/pdf/IWM99_RB_Restoration_Best%20Practice%20Guidance.pdf

Morrogh-Bernard, H., Husson, S., Page, S., & 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), 141–152.

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

Ochi, N., Okada, M., Hayashi, H., Shimokura, H., & Kamiya, M. (2016). Arrangement and structure of weirs on the Kalampangan Canal. In *Tropical Peatland ecosystems* (pp. 313–325). Springer Japan.

Page, S. E., Hoscilo, A., Langner, A., Tansey, K., Siegert, F., Limin, S., & Rieley, J. (2009). Tropical peatland fires in Southeast Asia. In M. A. Cochrane (Ed.), *Tropical fire ecology* (pp. 263–287). Springer.

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

Page, S. E., Rieley, J. O., Shotyk, Ø. W., & Weiss, D. (1999). Interdependence of peat and vegetation in a tropical peat swamp forest. *Changes and Disturbance in Tropical Rainforest in South-East Asia*, 354(1391), 161–173.

Page, S. E., Wűst, R. A. J., Weiss, D., Rieley, J. O., Shotyk, W., & 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), 625–635.

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

Posa, M. R. C., Wijedasa, L. S., & Corlett, R. T. (2011). Biodiversity and conservation of tropical peat swamp forests. *Bioscience*, 61(1), 49–57.

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

President of the Republic of Indonesia (2020). President of the Republic of Indonesia Regulation No. 120 Year 2020 about Peatland Restoration and Mangrove Agency (BRGM), 1-23.

Price, J. S., Heathwaite, A. L., & Baird, A. J. (2003). Hydrological processes in abandoned and restored peatlands: An overview of management approaches. *Wetlands Ecology and Management*, *11*(1), 65–83.

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

Putra, E. I., Cochrane, M. A., Vetrita, Y., Graham, L., & Saharjo, B. H. (2018). Determining critical groundwater level to prevent degraded peatland from severe peat fire. *IOP Conference Series: Earth and Environmental Science*, *149*, 012027.

Rahajoe, J. S., Alhamd, L., Atikah, T. D., Pratama, B. A., Shiodera, S., & Kohyama, T. S. (2016). Floristic diversity in the Peatland ecosystems of Central Kalimantan. In *Tropical Peatland ecosystems* (pp. 167–196). Springer Japan.

Republic of Indonesia Government (2016). Republic of Indonesia Government Regulation No. 57 Year 2016 about Peatland Ecosystem Protection and Management, 1-16.

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

Roucoux, K. H., Lawson, I. T., Baker, T. R., Del Castillo Torres, D., Draper, F. C., Lähteenoja, O., Gilmore, M. P., Honorio Coronado, E. N., Kelly, T. J., Mitchard, E. T. A., & Vriesendorp, C. F. (2017). Threats to intact tropical peatlands and opportunities for their conservation. *Conservation Biology*, *31*(6), 1283–1292.

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

Sari, A. P., Dohong, A., & Wardhana, B. (2021). Innovative financing for Peatland restoration in Indonesia. In R. Djalante, J. Jupesta, & E. Aldrian (Eds.), *Climate change research, policy and actions in Indonesia* (pp. 247–264) Springer Climate: Springer.

Sinclair, A. L., Graham, L. L. B., Putra, E. I., Saharjo, B. H., Applegate, G., Grover, S. P., & Cochrane, M. A. (2020). Effects of distance from canal and degradation history on peat bulk density in a degraded tropical peatland. *Science of the Total Environment*, *699*, 134199.

Susilo, G. E., Yamamoto, K., Imai, T., Inoue, T., Takahashi, H., Ishii, Y., Fukami, H., Koizumi, K., & 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), 263–274.

Takahashi, H., Ibie, B. F., Usup, A., Shimada, S., & Limin, S. H. (2002). Annual changes of water balance and a drought index in a tropical peat swamp forest of Central Kalimantan, Indonesia. International Symposium on Tropical Peatlands, Jakarta (Indonesia) (pp. 22–23).

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

Taufik, M., Setiawan, B. I., & 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*, 1–10.

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

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

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

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

Warren, M., Hergoualc'h, K., Kauffman, J. B., Murdiyarso, D., & 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*(1), 12.

Wichmann, S., Prager, A., & Gaudig, G. (2017). Establishing sphagnum cultures on bog grassland, cut-over bogs, and floating mats: Procedures, costs and area potential in Germany. *Mires & Peat*, *20*, 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., ... Andersen, R. (2017). Denial of long-term issues with agriculture on tropical peatlands will have devastating consequences. *Global Change Biology*, *23*(3), 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., & Hooijer, A. (2020). Distance to forest, mammal and bird dispersal drive natural regeneration on degraded tropical peatland. *Forest Ecology and Management*, *461*, 117868.

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

Wong, G. X., Hirata, R., Hirano, T., Kiew, F., Aeries, E. B., Musin, K. K., Waili, J. W., Lo, K. S., & Melling, L. (2018). Micrometeorological measurement of methane flux above a tropical peat swamp forest. *Agricultural and Forest Meteorology*, *256–257*, 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., & 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), 495.

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

Wösten, J. H. M., 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., & Wibisono, I. T. (2006). Interrelationships between hydrology and ecology in fire degraded tropical peat swamp forests. *International Journal of Water Resources Development*, *22*(1), 157–174.

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

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