RESEARCH ARTICLE

Increased fire hazard in human-modified wetlands in Southeast Asia

Muh Taufik, Budi I. Setiawan, Henny A. J. Van Lanen

Received: 25 July 2017 / Revised: 24 April 2018 / Accepted: 20 July 2018 / Published online: 3 August 2018

Abstract Vast areas of wetlands in Southeast Asia are undergoing a transformation process to human-modified ecosystems. Expansion of agricultural cropland and forest plantations changes the landscape of wetlands. Here we present observation-based modelling evidence of increased fire hazard due to canalization in tropical wetland ecosystems. Two wetland conditions were tested in South Sumatra, Indonesia, natural drainage and canal drainage, using a hydrological model and a drought-fire index (modified Keetch–Byram index). Our results show that canalization has amplified fire susceptibility by 4.5 times. Canal drainage triggers the fire season to start earlier than under natural wetland conditions, indicating that the canal water level regime is a key variable controlling fire hazard. Furthermore, the findings derived from the modelling experiment have practical relevance for public and private sectors, as well as for water managers and policy makers, who deal with canalization of tropical wetlands, and suggest that improved water management can reduce fire susceptibility.

Keywords Canalization · Canal water level · Fire hazard · SWAP · Water management

INTRODUCTION

Tropical wetland ecosystems typically have shallow groundwater tables throughout the year that support a wealth of endemic species because of their anaerobic, and frequently extreme acidic and nutrient poor conditions (Yule 2010; Evers et al. 2017). Recent literature explains that tropical wetlands in Southeast Asia, including peat swamps, are biodiversity hotspots (Kier et al. 2005; Sodhi et al. 2010; Yule 2010; Evers et al. 2017) for endemic flora

and fauna. Additionally, these ecosystems store large amounts of carbon compared to other tropical wetlands in Africa and America (Page et al. 2011), especially in belowground biomass. In addition, wetland ecosystems store large volumes of water in their soils (up to 80–90%) because of the nature of the soil (organic), high rainfall, and the restricted drainage under natural conditions, which creates an environment that is little prone to fire.

Recent human activities in Southeast Asian tropical wetland ecosystems are increasingly changing the landscape from natural forest into export-oriented agricultural cropland and forestry. In many cases, these changes can be attributed to clearance of natural forest. Land-use change in the last two decades has reduced wetland forest cover by 36% in Indonesia and Malaysia (Miettinen et al. 2012). This reduction included various measures to create favourable conditions for agronomy and forestry practices in wetlands, for instance drainage through canalization. Canalization is a common practice to drain excessive water, making wetland suitable for agriculture and forestry purposes. By draining water, the groundwater table depth declines (Hirano et al. 2012; Ishii et al. 2016), depending on the water level in the canal and subsurface characteristics.

Furthermore, climate variability mainly induced by El Niño has increased the pressure on the often drier wetland ecosystems. During El Niño, prolonged low precipitation led to lower groundwater tables in wetland ecosystems, which created a drier environment that favours extensive fires to burn in tropical forests. More than 2.6 million hectares of crops and forests in Indonesia were burnt by the 2015 fires (Tacconi 2016), the second worst fires of the past two decades in Southeast Asia. The extent and severity of wildfires has increased during El Niño events (Wooster et al. 2012; Taufik et al. 2017). Studies reported that fire

susceptibility increased in human-modified wetlands with canalization (Hoscilo et al. 2011; Konecny et al. 2016).

Although there has been growing interest in wetland ecosystem studies, fire-drought research that integrates groundwater as a key variable for fire hazard in wetlands is limited. Current research mostly uses soil moisture deficiency derived from only climate information, such as the Keetch–Byram Drought Index-KBDI (Petros et al. 2011) and the Fire Weather Index-FWI (Amiro et al. 2005), as a proxy for flammability. In wetlands, however, groundwater tables greatly influence soil moisture dynamics, hence the fire hazard (Wösten et al. 2008; Taufik et al. 2015; Takeuchi et al. 2016; Taufik et al. 2017). The closer the groundwater table to the surface, the lower the fire hazard is, as reported by Wösten et al. (2008) and Takeuchi et al. (2016). The position of the groundwater table varies in response to wet and dry spells. Occasionally, the groundwater table substantially declines during a prolonged drought, which coincides with strong El Niño events (e.g. 1997/1998 and 2015). As expansion of land for product-oriented export (oil palm, fibre) takes place, humans now also play a key role in groundwater dynamics in wetland ecosystems through building canal networks (Ritzema et al. 2014). This requires research that explores the interplay of humans and fires occurring in the changing landscapes of tropical wetland ecosystems. The aims of this study are (i) to quantify the impact of human interference on increased fire hazard, and (ii) to assess the amplification of fire hazard caused by human interference. We used an observation-based modelling approach to connect surface water regimes (including canalization) through a soil water model that considers groundwater table depth to fire hazard.

MATERIALS AND METHODS

Study area and data collection

Our study area is a wetland ecosystem in South Sumatra, Indonesia (Fig. 1), within the evergreen humid climate zone, that generally experiences some water shortage during the dry season. The area receives mean precipitation of 2540 mm annually, whereof 1700 mm during the wet season. The peak dry season occurs from August to September, when the median monthly rainfall is below 100 mm, whereas the typical wet season from November to March. The study region was burnt during the 1997/1998 El Niño due to wildfires. In the late 2000s, the region has been planted with a fast-growing monoculture forest plantation (i.e. *Acacia crassiparpa*). To create favourable conditions for root development, water and nutrient uptake, excessive water in the wetlands is drained through

canalization (Fig. 1c). Additionally, the canals are of importance for transportation, logging, and logistics.

The area is characterized by mineral soils, which mainly consist of silty clay, and sand at very few places. In the study site of Baung (105.3°E and 2.74°S, Fig. 1), we built a mini-AWS (automatic weather station) to monitor daily weather (rainfall and air temperature) and soil moisture. In this station, we built a piezometer to monitor the groundwater depth. In the piezometer, we installed a pressure transducer to measure the water depth every hour. Simultaneously, we measured atmospheric pressure with a Baro Diver (Van Essen Instruments) in the station. Once every 3 months, we manually monitored the groundwater tables in the field to check the pressure-transducer measurements. In this station, soils were sampled in two different depths (0–30 cm and 30–60 cm) to represent the upper and the deep layers using ring sample of 100 cm³. The soils were then analysed in a soil laboratory to obtain physical characteristics, namely water retention and soil hydraulic conductivity. For details on the soil characteristics, readers are referred to Taufik et al. (2015). The data monitoring at the site lasted for 2 years, covering two contrasting ENSO conditions, that is, in 2009 (El Niño, dry) and in 2010 (La Niña, wet).

SWAP model and data

Soil Water Atmosphere Plant (SWAP) is a one-dimensional, vertically oriented model to simulate transport of water, solutes, and heat in the vadose zone in interaction with vegetation development and subsurface hydrology (Kroes et al. 2008, Fig. 2). Its model domain covers the zone between the groundwater table and the soil surface on which vegetation can grow and water ponding (i.e. water on land) may occur. Water balance components and groundwater levels are simulated by numerically solving soil moisture flow using the Richards equation (Eq. 1, Van Dam and Feddes 2000). SWAP requires weather data, soil-hydrological properties, and vegetation characteristics.

$$\frac{\partial \theta}{\partial t} = \frac{\partial}{\partial z} \left[K(\theta) \left(\frac{\partial h}{\partial z} + 1 \right) \right] - S, \quad (1)$$

where θ is volumetric soil moisture content (m³ m⁻³), t time (days), z the elevation (m, positive upwards), h soil water pressure (m), S a sink of the system (m days⁻¹), which accounts for external losses like evapotranspiration and drainage, and $K(\theta)$ the conductivity (m days⁻¹) as a function of water content (θ). The SWAP model uses an implicit, backward, finite difference scheme to solve the Richards equation. The soil water content, θ , (m³ m⁻³) is calculated applying the van Genuchten approach (Eq. 2):

Fig. 1 Wetland study site in South Sumatra, Indonesia: **a** locations of the Baung station and Kenten station in the region. Data from the Baung station were used for model calibration, while data from the Kenten station were applied for drought/fire modelling, **b** location of the Baung station in the area of the nine grids of GFED4 at 0.25° resolution, **c** photo of canal surrounded by the 4-year-old *Acacia* plantation, and **d** schematic diagram of the canal with dimensions

$$\theta(h) = \theta_r + \frac{(\theta_s - \theta_r)}{[1 + |\alpha h|^n]^{1-1/n}}, \quad (2)$$

where θ_r , θ_s are the residual and saturated soil moisture content ($\text{m}^3 \text{m}^{-3}$), respectively, and α (–) and n (–) are the shape parameters (Van Genuchten 1980).

The model runs with a daily time step and is composed of 36 vertical layers representing the soil up to a depth of 300 cm below the soil surface (Fig. 2). The first 10 cm was simulated using 10 layers of 1 cm, followed by four layers of 5 cm, followed by seven layers of 10 cm, and followed by 15 layers of 20 cm. For drainage, we applied a Cauchy type of boundary condition. Soil water uptake by roots is assumed to be evenly distributed up to a depth of 70 cm (rooting depth *Acacia*).

The top boundary condition of SWAP consists of precipitation and evapotranspiration, which is driven by

weather and the interplay between weather, vegetation, and soil conditions. We calculated reference evapotranspiration based on the Penman–Monteith FAO method (Allen et al. 1998). SWAP simulates actual daily soil evaporation and transpiration by combining reference evapotranspiration with information on vegetation development stage, crop coefficient, soil/vegetation cover, and soil moisture. We used a crop coefficient of 1.05 for the wetland conditions, which is about the median of the range in the coefficient previously reported (Allen et al. 1998; Sun et al. 2011). Further, we assumed no change in vegetation development stage of the mature monoculture forestry, i.e. we applied a constant soil cover fraction of 80% by the *acacia* canopy.

As a lower boundary condition, different drainage conditions can be included in the SWAP model. For example, natural drainage or human-modified drainage systems can

be implemented in the model through the lateral drainage option. In this study, SWAP calculating the drainage flux (Q_{drain}) was selected, which is driven by the difference between the groundwater elevations and the canal/stream level (Fig. 2a). We argue that the variation of the phreatic groundwater in wetlands of Southeast Asia is less than 2 m, as shown in our data and in the literature (Wösten et al. 2008; Hirano et al. 2015), even during El Niño years. This led us to use a 3-m soil profile (Fig. 2a) for the modelling. In the end, the daily outputs of the model are time series of soil moisture profile and groundwater levels. Figure 2 presents the schematization of the SWAP model.

Moisture in the vertical soil column is controlled by surface water levels of drainage system (canal) through the groundwater system (saturated flow in the shallow aquifer). Two types of drainage system were defined, i.e. natural drainage and human-modified drainage through wetland canalization. The canalization creates deep open channels in wetlands that reach depths up to 6 m below surface depending on operational purposes, such as water transport and water management. During the wet season, the canal water level is close to the surface, whereas it may substantially decline during the dry season. In the study site, controlled drainage is applied through canal blocking (e.g. Ritzema et al. 2014) to ideally maintain groundwater tables in the range of 40–90 cm (see Evers et al. 2017). When we use canalization in this paper, we refer to controlled drainage (i.e. reference situation), unless it is stated clearly otherwise. For natural drainage, we used a surface water depth of 50 and 100 cm during the wet and dry season, respectively (130 cm during the dry season for strong El Niño years), which drives water losses via the natural drainage system that usually can be found in this kind of regions. These estimates are based upon field experiences in Sumatra and Kalimantan.

Model calibration

For the calibration process, we used the available daily weather and groundwater table data from almost 2 years of observations (Taufik et al. 2015) from 1 April 2009 to 15 March 2011. These observed data are important, as it covers a period with climate extremes. Both the 2009 El Niño and the 2010 La Niña are included. We applied two approaches for assessing model performance. First, we used a visual inspection of its performance. This approach aims to detect model behaviour, and to obtain an overview of the overall performance (Bennett et al. 2013). Then we applied five statistical criteria of goodness of fit (see Bennett et al. 2013) including percent bias (*pbias*), RMSE-observations standard deviation ratio (*rsr*), index agreement (*Id*) coefficient of correlation (*r*), and Kling Gupta Efficiency (*kge*).

Percent bias (PBIAS) measures the average tendency of the simulated data to be larger or smaller than their observed counterparts. Ratio of the Root Mean Square Error (*rsr*) between simulated and observed values to the standard deviation of the observations, index of agreement (*Id*) compares the sum of squared error to the potential error. The Pearson's correlation coefficient (*r*), which ranges from -1 to 1 , is an index of the degree of linear relationship between observed and simulated data. The *kge* provides a diagnostically interesting decomposition of the Nash–Sutcliffe efficiency, which facilitates the analysis of the relative importance of its different components (correlation, bias, and variability) in the context of hydrological modelling.

Our calibration showed that the SWAP model performed well in simulating groundwater tables (Fig. 3). The onset of the groundwater table drop and its recovery are well simulated during the El Niño in 2009, whereas there is some overestimation of the groundwater table during the La Niña year in 2010. The well model performance is reflected in the high *kge*, *Id*, and *r*, and low *pbias* and *rsr* (Fig. 3). By means of this calibration, we determined the values of the key model input data for SWAP, namely canal water level (CWL) regime and drainage resistance (DR). The CWL should be not deeper than 130 cm below the surface during the dry season. For the other seasons, we applied a canal water level closer to the surface, namely 50 and 80 cm below the surface during the wet and intermediate season (May/June), respectively. The calibrated DR was 200 days for the study site with canal drainage. The drainage resistance determines how easy water flows through the groundwater system to the drainage system. The lower the resistance, the easier water flows. The natural situation is characterized by a higher drainage resistance than human-modified drainage, because of the longer distance between natural channels compared to the drainage canals. We anticipated that the drainage resistance for natural drainage is around three times higher (600 days). This estimate is based on drainage theory (Ritzema 1994) and field observations of differences between canal and stream distances.

Fire hazard assessment

We used the modified Keetch–Byram Drought Index (mKBDI) to assess daily fire hazard and the associated fire hazard class (i.e. low, moderate, and high). The hazard class follows previous research in Southeast Asia (Ainuddin and Ampun 2008; Taufik et al. 2015). The daily time-step calculation of the mKBDI is as follows (Eqs. 3–6):

$$\text{mKBDI}^t = \text{mKBDI}^{t-1} + \text{DF}^t + \text{RF}^t - \text{WTF}^t. \quad (3)$$

Here mKBDI is the moisture deficiency, DF is the drought factor, RF is the rainfall factor, and WTF is the water

Fig. 3 Hydrograph of observed (obs) and simulated (sim) groundwater tables with the SWAP model during the calibration period. This graph shows that the model performed well in simulating groundwater table in particular for the dry season, but it slightly overestimates the shallow levels in the wet season, which are controlled by local site conditions. Goodness-of-fit measures are provided: *pbias* (percent of bias), RMSE-observations standard deviation ratio (*rsr*), index of agreement (*Id*), coefficient of correlation (*r*), and Kling Gupta Efficiency (*kge*). The smaller the *pbias* and *rsr*, the better model performs (Moriassi et al. 2007). On other hand, the model performs well if the *Id*, *r*, and *kge* are high (Bennett et al. 2013)

table factor on day t . The drought factor (DF^t , Taufik et al. 2015) on a given day in the metric system is

$$DF^t = (203 - KBDI^{t-1}) \frac{(0.4982e^{(0.0905 \times T_m + 1.6096)} - 4.268) \times 10^{-3}}{1 + 10.88e^{(-0.001736 \times R_0)}}, \quad (4)$$

where T_m is daily maximum air temperature, and R_0 is average annual rainfall. We used 2500 mm/year for the rainfall.

Rainfall is considered to reduce the drought index, if it is more than 5.1 mm/day (Eq. 5):

$$RF^t = \begin{cases} (R^t - 5.1), & R^t \geq 5.1 \text{ mm/day, 1st rainy day} \\ R^t, & R^{t-1} \geq 5.1 \text{ mm/day, 2nd and the next rainy days} \\ 0, & R^t < 5.1 \text{ mm/day} \end{cases} \quad (5)$$

Then, the water table factor (WTF) takes the following form:

$$WTF^t = 10.64 - 0.283x[(1 - \theta(h)^t) * 203]. \quad (6)$$

The mKBDI is scaled from zero to 203 as maximum value. Prolonged extreme wet spells increase soil moisture to eventually reach saturation, and therefore mKBDI is at minimum (0), whereas long-lasting hot and dry spells create favourable conditions for mKBDI to reach its maximum value. The groundwater level also affects the fire hazard, i.e. shallow groundwater tables reduce fire hazard through capillary rise. When the groundwater level drops below a critical threshold (e.g. 85 cm below surface in the study site, Taufik et al. 2015), it does not reduce fire hazard because capillary rise cannot sufficiently feed the topsoil anymore.

The emphasis in this study is to explore the probabilities of groundwater table depths and associated fire hazards for the current state of plantation rather than to reconstruct historic fire hazards also considering the plantation development. The long time series of weather data (1980–2015) for simulation of groundwater levels, and hence, the fire hazard was obtained from the nearby climate station Kenten (latitude/longitude are 2.93°S/104.77°E) in South Sumatra, Indonesia. There is no difference in input data for the calculation of fire hazard (Eqs. 3–6) for the two scenarios (natural and canal drainages), except for the water table factor. WTF was computed using different simulated groundwater table series based on the two scenarios used in this study, i.e. natural and canal drainage. Then, one of the three fire hazard classes was assigned to each day, namely low, moderate, or high fire hazard.

To analyse the robustness of the predicted fire hazard, a bootstrapping experiment was designed. From the 36-year period covered, x random years (ranging from 3 to 36) are selected to generate the class of fire hazard. Then we only focus on the high fire class to analyse the uncertainty of the prediction.

Verification of fire hazard

To test the performance of the mKBDI, we used monthly fire area burnt derived from the Global Fire Emission Dataset-GFED (Giglio et al. 2013). The GFED is at 0.25° resolution, which is available from 1996 onward. The drought index (i.e. mKBDI) reflects dryness over a large area and not specifically the Baung station site. Therefore, we used nine GFED grid cells (Fig. 1b) surrounding the site for verification. We verified performance of mKBDI (mean

monthly) for the canal drainage situation in 1996–2015, as this represented the weather situation that actually has driven the fires. The canalization in the region started well before our fire hazard verification period, i.e. in the 1990s. Totally, there are 240 data points, that is, months for verification.

RESULTS

Simulated groundwater tables

The time the simulated daily water table is deeper than the critical threshold (85 cm) is 1.9% under natural drainage and 34% under canal drainage (not shown). Furthermore, the SWAP simulations also show that ponding is normal for wetland ecosystems under natural drainage, whereas the ponding period becomes eight times shorter under canal drainage. Ponding in excess of 0.10 m depth does not occur under canal drainage at the investigated site.

On a monthly scale, the groundwater level is always below the surface during the period August–October under natural drainage. In this period, the minimum groundwater level reaches the critical threshold (-0.85 m, Fig. 4) only a few occasions (9 times in 1980–2015). Canal drainage causes lower groundwater levels below the surface throughout the whole year compared to natural drainage. Groundwater gradually starts to decline in May, and will reach its deepest levels in October. In the period July–October, the median groundwater level is below the critical threshold and in November it is still very close. This is clear evidence that canalization prolongs periods with low

Fig. 4 Monthly groundwater tables for two drainage conditions during 1980–2015. The boxplot indicates the median, and the 25 and 75% quantiles. The whisker represents 10 and 90% of quantiles. The dots indicate outliers. We use depth of -0.85 cm as a critical threshold (a dash-blue line), which below this depth groundwater levels do not influence fire hazard anymore. Y-axis is negative to present depth below the surface

groundwater, hence it increases dry conditions that favour fire.

Verification of fire hazard

Our analysis showed that fire events (expressed as area burnt) were reported for all three fire hazard classes derived from the mKBDI (Table 1, canal drainage). In total, 172 out of 240 months were assigned to the low hazard class, 23 months were in the moderate class, and the remaining (45) were categorized as having a high probability of fire. Of the 172 months that were reported to have a low level, 68% (117) of the months had zero burnt area, whereas only 1 month had an area burnt larger than 5000 ha (Table 1). On the other hand, months with high fire hazard probability were characterized by large area burnt (> 5000 ha, 22 months).

Few small areas burnt (< 100 ha, 3 months) were reported when the mKBDI of a certain month is in the high class. For the moderate probability on fire, most of the months had an area burnt of 100–5000 ha. This verification demonstrates that mKBDI is able to classify fire events in three different hazard levels based on the size of area burnt. In summary, the mKBDI is useful to identify large area burnt that is characterized by the high fire class.

Fire hazard behaviour

Wetland ecosystems store large volumes of water in their soils because of high organic matter contents, high rainfall both monthly and annually, and the restricted drainage under natural conditions, which creates an environment that is little prone to fire. About 90% of the time, fire hazard under natural drainage is at a low level at the investigated site, and 3.8% of the time at a high level, which is during a prolonged dry season (Table 2). With drainage by canalization, low fire hazard level conditions drop to 72.2% of the time. Canalization causes lower groundwater levels during the dry season, leading to an increased presence of the high fire hazard level from 3.8%

Table 1 The size distribution of observed monthly area burnt (total of nine grid cells, in ha) for the three different predicted fire hazard classes, 1996–2015. The class is derived from the mKBDI under canal drainage

Predicted hazard class	Observed area burnt (ha)				Total
	0	0–100	100–5000	> 5000	
Low	117	22	32	1	172
Moderate	6	2	15	0	23
High	7	3	13	22	45

Table 2 Fire hazard (% of time) using weather data from 1980–2015. Two different types of climatic years are introduced: normal and El Niño years. Fire hazard is clustered in three classes. Column 2 should be compared with column 5, column 3 with column 6, and column 4 with column 7

Hazard class	Drainage conditions					
	Natural			Controlled canal		
	Normal	El Niño	Total	Normal	El Niño	Total
Low	61.2	28.9	90.1	49.1	23.1	72.2
Moderate	2.1	4.0	6.0	7.0	3.7	10.7
High	0.6	3.2	3.8	7.9	9.2	17.1

under natural drainage to 17.1% of the time. This implies an amplification of fire susceptibility by 4.5 times when natural wetland forest is drained.

High fire hazard is reported to be low for normal years and natural drainage conditions (0.6%), whereas this is 7.9% of the time under canal drainage (Table 2). Most high fire hazard conditions under controlled canal drainage events (9.2%) coincide with El Niño. This provides evidence that El Niño strongly influences fire hazard, which is induced by drainage.

Fire behaviour is strongly controlled by groundwater level. As anticipated, we found a strong correlation of the groundwater level and KBDI with $r = -0.85$ and -0.88 for natural and canal drainage, respectively. Other variables, such as daily rainfall, were only weakly correlated to fire hazard level (not shown). This correlation means that during prolonged periods with low groundwater levels, high fire level is expected to occur.

Under natural drainage conditions, monthly averaged groundwater levels decline during the period July–October, but they rarely exceed the critical threshold (Fig. 2, natural). This little drawdown does not create high fire hazard. Totally only 3.8% (503 days) of the days are in the high fire hazard class (Table 2) that occurs only in 11 out of 36 years. The remaining years have predominantly a low fire hazard level. During warm ENSO years, the fire susceptibility increases more than five-fold (3.2 vs. 0.6%). Table 3 provides the distribution of the average number of days with a high fire hazard level for each month (15–94% of days for natural drainage, dependent on the month) in those years when fires occur (3–31% of the years). If we exclude August and December, because of the low number of years when fires occur in these months ($<10\%$), the fraction of days with a high fire hazard level is 40–64% in the years with fires (22–31%, Table 3). On natural drainage, high fire hazard level in December occurred only in 1 year (3%) during the very strong 2015 El Niño; almost the entire month (94%) was at a high fire hazard level.

Table 3 Distribution of high fire risk days (average number of days per month, in %) over the months, and the number of years (%) that high fire risk happens for two drainage conditions using weather data from 1980–2015

Month	Drainage conditions			
	Natural		Controlled canal	
	High fire level days (%)	Number of years (%)	High fire level days (%)	Number of years (%)
June	–	–	3	3
July	–	–	41	25
August	15	8	66	64
September	40	25	73	89
October	64	31	71	83
November	56	22	70	36
December	94 ^a	3	55	6

^a During El Niño 2015

Canalization, however, leads to low groundwater levels during the dry season, which can go well below the critical threshold (Fig. 2, controlled canal). About 33% of the time, the groundwater level is below the threshold, which leads to an increased frequency of days with a high fire hazard level (Table 2). High fire hazard level is recorded for 32 out of 36 years in the period 1980–2015. This implies that about 88% of the years are at a high hazard level, which is about 2.4 times higher than under natural drainage conditions. Seasonally, high fire hazard levels frequently occur during the period July–November, but occasionally they also occur in June and December (Table 3). The mean number of high fire hazard level days in the period August–October under canal drainage conditions is 21.4 days per month (66–73% of the days) for those years that fires occur. Moreover, under canalization, the fire season comes at least 1 month earlier, starting in July (June not considered because of the low number of years), whereas the fire season under natural drainage conditions starts in August (Table 3). This is evidence that canalization lengthens the period with a high fire hazard level in addition to the substantially larger number of fire years. The higher percentage of high fire level days in December under natural conditions than under controlled drainage (94 vs. 55%) is caused by the higher number of years that show high fire hazard under controlled drainage (6% of the years relative to 3% under natural conditions).

DISCUSSION AND CONCLUSIONS

This research contributes to the emerging literature that attempts to quantify the interplay of humans on

hydrological processes in tropical forest wetlands, including fire hazard, using an observation-based modelling experiment. This includes the interplay between El Niño and associated lower surface water levels, deeper groundwater tables, consequently higher fire hazard. We applied the SWAP model, which showed good agreement with 2 year of observations, to simulate time series of daily groundwater tables for a wetland ecosystem in South Sumatra with input of 36 years of daily weather data. Two wetland drainage conditions were introduced for the SWAP simulation, i.e. natural and canal drainage conditions for a mature acacia plantation. Of the two wetland drainage types tested, canal drainage generates longer periods with low groundwater levels (Fig. 2, controlled canal). Next, we calculated the daily modified KBDI using the simulated water tables to assess daily fire hazard for each drainage condition. Our verification of the simulated fire hazard level by using 20 years of observed monthly area burnt revealed that fire may occur on each day irrespective of the estimated fire hazard level. A condition of large area burnt (> 5000 ha) occurs only when the fire hazard level is high, and the opposite, no or a small area burnt occurs when the fire hazard level is low.

Hydrological studies in tropical wetlands found very low groundwater levels during the dry season due to canalization, for instance, in the ex-Mega Rice Project, Central Kalimantan, Indonesia. This is consistent with our substantial difference (between natural and canal drainage) in number of days with groundwater levels below the critical threshold (i.e. 24%). Prolonged low groundwater levels create conditions that favour wildfire. For a peat-swamp forest in Kalimantan, Wösten et al. (2008) showed that the probability of high fire hazard increases when the groundwater level drops below 40 cm. Other studies under similar hydroclimatological conditions used a groundwater table depth of 100 cm as critical threshold for fire (Wösten et al. 2006; Takeuchi et al. 2016). Here, we apply 85 cm as critical groundwater level for identifying high fire hazard. The critical groundwater depth is not constant for all wet tropical ecosystems but depends among others on soil properties (Taufik et al. 2015).

Although there is substantial understanding on how low precipitation enhances fire hazard, our results highlight the fact that efforts to understand fire hazard without examining the role of soil moisture and groundwater table depth miss a critical contributor to fire hazard in wetland ecosystems (Wösten et al. 2006, 2008; Takeuchi et al. 2016; Taufik et al. 2017). In other studies, like on hydroclimatic change, the connection of soil moisture and groundwater depth is well recognized (e.g. Leung et al. 2011; Destouni and Verrot 2014; Verrot and Destouni 2016). Our finding supports this claim by the following: (i) drainage conditions clearly affect fire hazard given the

same rainfall input, i.e. canal drainage has lower groundwater levels, and hence higher fire hazard than natural drainage, and (ii) controlled drainage through canal blocking causes high fire hazard to come at least 1 month earlier. This outcome confirms a recent study, which reported increased fire susceptibility in drained wetlands of Central Kalimantan (Konecny et al. 2016), which points at ecosystem degradation. The finding is in line with similar experiences in mangrove wetlands where canalization also leads to degraded ecosystems (Jaramillo et al. 2018a, b).

Our findings show that there is substantial amplification of fire hazard when natural forest is drained for agronomy. First, the frequency of high fire levels increases by 4.5 times (Table 2). The increase means that fire susceptibility under drained conditions is higher and occurs in the dry season for 9 out of 10 years. In addition, the number of days experiencing high fire hazard levels has risen by 70% in the period August–October (Table 3). The number of years having a high fire hazard level rose 3.6 times. Further, our modelling experiment reveals that drainage lengthens the period with high fire hazard level through an earlier start of the fire season. By applying a bootstrap approach, we found that our prediction of fire hazard (under canal drainage) was robust, as shown by low variation in the prediction (0.7%). The amplification applies for controlled drainage conditions, therefore the question remains how large the amplification will be in case of uncontrolled drainage in this type of ecosystem.

Our modelling experiment shows that, if the drainage is uncontrolled (with canal water levels reaching to 3 m depth during the dry season), prolonged low groundwater tables are inevitable. Hence, days with high fire hazard will increase substantially, from 17% (controlled drainage) to almost 40% (Fig. 5). In other words, without improved water management, the expected frequency of high fire hazard levels will be more than twice as high (relative to the reference) and will occur each year. The very low groundwater levels will likely influence the endemic flora and fauna (Kier et al. 2005; Sodhi et al. 2010) that can only survive under un-impacted hydrological conditions (Evers et al. 2017). A lot of effort has been proposed to restore degraded wetland, including canal blocking (Ritzema et al. 2014; Ishii et al. 2016), with the purpose to maintain groundwater levels as close as possible to natural conditions. Improved water management is anticipated to provide an adequate basis for endemic biodiversity and additionally to control wildland fires. However, maintaining groundwater levels as high as possible needs to be negotiated with the agronomy sector, because it likely has detrimental effects. The application of the proposed analysis tools (SWAP and mKBDI) provides guidance for best practice water management, including a critical canal water level during the dry season to avoid a substantial increase

Fig. 5 Impact of water management on high fire hazard frequency (%) of the time). The deeper canal water levels during the dry season reflect uncontrolled drainage condition (reference represents controlled drainage). x-axis values indicate maximum depth of the surface canal water level (CWL) below soil surface in the dry season

of days with a high fire hazard. The canal water levels should be kept as high as possible during the dry and intermediate season to avoid high fire hazard levels. For instance, if the levels are not allowed to go deeper than 150 cm below the surface during the peak dry season (August–October), the expected occurrence frequency of high fire hazard levels would be kept below 20%, which might be an acceptable compromise, although it is still 6 times larger than under natural conditions.

A number of caveats should be considered. For example, our study uses an observation-driven modelling approach that analyses groundwater level dynamics under current climate variability, but it did not explicitly attempt to reconstruct actual historic conditions (e.g. acacia plantation development), nor did it explore future climates that may favour fire susceptibility. Likewise, we used a one-dimensional vertical water flow model and a drainage boundary conditions (Cauchy type) to simulate groundwater tables (Van Dam and Feddes 2000; Kroes et al. 2008; Van Dam et al. 2008) instead of a (pseudo) three-dimensional model (e.g. Wösten et al. 2006; Ishii et al. 2016), which may introduce some uncertainty. Although our calibration period captures both climate extreme conditions (El Niño and La Niña) our calibration results may disregard the influence of decadal climate variability on groundwater table dynamics. Long-term monitoring of groundwater levels could help reduce the uncertainty in calibrated parameters. However, we believe that given the observed data used from years with climate extremes, this modelling-monitoring study contributes towards better understanding on the interplay between human and natural disturbances in the fragile ecosystems of tropical wetlands in Southeast Asia. Moreover, our findings offer valuable

science-based evidence to policymakers in the region to regulate and promote sustainable wetland utilization.

Acknowledgements This present study was completed with support of the DIKTI Scholarship (Contract No: 4115/E4.4/K/2013) and the SPIN-JRP-29 project Granted by the Royal Netherlands Academy of Arts and Sciences (KNAW). It contributes to WIMEK-SENSE and the UNESCO IHP-VIII programme FRIEND-Water.

REFERENCES

- Ainuddin, N.A., and J. Ampun. 2008. Temporal Analysis of the Keetch-Byram Drought Index in Malaysia: Implications for Forest Fire Management. *Journal of Applied Sciences* 8: 3991–3994. <https://doi.org/10.3923/jas.2008.3991.3994>.
- Allen, R.G., L.S. Pereira, D. Raes, and M. Smith. 1998. *Crop Evapotranspiration-Guidelines for Computing Crop Water Requirements*. Rome: FAO.
- Amiro, B.D., K.A. Logan, B.M. Wotton, M.D. Flannigan, J.B. Todd, B.J. Stocks, and D.L. Martell. 2005. Fire Weather Index System Components for Large Fires in the Canadian Boreal Forest. *International Journal of Wildland Fire* 13: 391–400.
- Bennett, N.D., B.F.W. Croke, G. Guariso, J.H.A. Guillaume, S.H. Hamilton, A.J. Jakeman, S. Marsili-Libelli, L.T.H. Newham, et al. 2013. Characterising performance of environmental models. *Environmental Modelling & Software* 40: 1–20. <https://doi.org/10.1016/j.envsoft.2012.09.011>.
- Destouni, G., and L. Verrot. 2014. Screening Long-term Variability and Change of Soil Moisture in a Changing Climate. *Journal of Hydrology* 516: 131–139. <https://doi.org/10.1016/j.jhydrol.2014.01.059>.
- Evers, S., C.M. Yule, R. Padfield, P. O'Reilly, and H. Varkkey. 2017. Keep Wetlands Wet: The Myth of Sustainable Development of Tropical Peatlands—Implications for Policies and Management. *Global Change Biology* 23: 534–549. <https://doi.org/10.1111/gcb.13422>.
- Giglio, L., J.T. Randerson, and G.R. Van Der Werf. 2013. Analysis of Daily, Monthly, and Annual Burned Area Using the Fourth-Generation Global Fire Emissions Database (GFED4). *Journal of Geophysical Research: Biogeosciences* 118: 317–328. <https://doi.org/10.1002/jgrg.20042>.
- Hirano, T., K. Kusin, S. Limin, and M. Osaki. 2015. Evapotranspiration of Tropical Peat Swamp Forests. *Global Change Biology* 21: 1914–1927. <https://doi.org/10.1111/gcb.12653>.
- Hirano, T., H. Segah, K. Kusin, S. Limin, H. Takahashi, and M. Osaki. 2012. Effects of Disturbances on the Carbon Balance of Tropical Peat Swamp Forests. *Global Change Biology* 18: 3410–3422. <https://doi.org/10.1111/j.1365-2486.2012.02793.x>.
- Hoschilo, A., S.E. Page, K.J. Tansey, and J.O. Rieley. 2011. Effect of Repeated Fires on Land-Cover Change on Peatland in Southern Central Kalimantan, Indonesia, from 1973 to 2005. *International Journal of Wildland Fire* 20: 578. <https://doi.org/10.1071/WF10029>.
- Ishii, Y., K. Koizumi, H. Fukami, K. Yamamoto, H. Takahashi, S.H. Limin, K. Kusin, A. Usup, et al. 2016. Groundwater in Peatland. In *Tropical Peatland Ecosystems*, 15th ed, ed. M. Osaki and N. Tsuji. Tokyo, Heidelberg, New York, Dordrecht, London: Springer.
- Jaramillo, F., I. Brown, P. Castellazzi, L. Espinosa, A. Guittard, S.-H. Hong, V.H. Rivera-Monroy, and S. Wdowinski. 2018a. Assessment of Hydrologic Connectivity in an Ungauged Wetland with InSAR Observations. *Environmental Research Letters* 13: 024003. <https://doi.org/10.1088/1748-9326/aa9d23>.

- Jaramillo, F., L. Licero, I. Åhlen, S. Manzoni, J.A. Rodríguez-Rodríguez, A. Guittard, A. Hylin, J. Bolaños, et al. 2018b. Effects of Hydroclimatic Change and Rehabilitation Activities on Salinity and Mangroves in the Ciénaga Grande de Santa Marta, Colombia. *Wetlands*. <https://doi.org/10.1007/s13157-018-1024-7>.
- Kier, G., J. Mutke, E. Dinerstein, T.H. Ricketts, W. Küper, H. Kreft, and W. Barthlott. 2005. Global Patterns of Plant Diversity and Floristic Knowledge. *Journal of Biogeography* 32: 1107–1116. <https://doi.org/10.1111/j.1365-2699.2005.01272.x>.
- Konecny, K., U. Ballhorn, P. Navratil, J. Jubanski, S.E. Page, K. Tansey, A. Hooijer, R. Vernimmen, et al. 2016. Variable Carbon Losses from Recurrent Fires in Drained Tropical Peatlands. *Global Change Biology* 22: 1469–1480. <https://doi.org/10.1111/gcb.13186>.
- Kroes, J.G., J.C. Van Dam, P. Groenendijk, R.F.A. Hendriks, and C.M.J. Jacobs. 2008. *SWAP version 3.2. Theory Description and User Manual*. Alterra: Wageningen.
- Leung, L.R., M. Huang, Y. Qian, and X. Liang. 2011. Climate–Soil–Vegetation Control on Groundwater Table Dynamics and its Feedbacks in a Climate Model. *Climate Dynamics* 36: 57–81. <https://doi.org/10.1007/s00382-010-0746-x>.
- Miettinen, J., C. Shi, and S.C. Liew. 2012. Two Decades of Destruction in Southeast Asia's Peat Swamp Forests. *Frontiers in Ecology and the Environment* 10: 124–128. <https://doi.org/10.1890/100236>.
- Moriassi, D.N., J.G. Arnold, M.W. Van Liew, R.L. Binger, R.D. Harmel, and T.L. Veith. 2007. Model Evaluation Guidelines for Systematic Quantification of Accuracy in Watershed Simulations. *Transactions of the ASABE* 50: 885–900. <https://doi.org/10.13031/2013.23153>.
- Page, S.E., J.O. Rieley, and C.J. Banks. 2011. Global and Regional Importance of The Tropical Peatland Carbon Pool. *Global Change Biology* 17: 798–818. <https://doi.org/10.1111/j.1365-2486.2010.02279.x>.
- Petros, G., M. Antonis, and T. Marianthi. 2011. Development of an Adapted Empirical Drought Index to the Mediterranean Conditions for Use in Forestry. *Agricultural and Forest Meteorology* 151: 241–250. <https://doi.org/10.1016/j.agrformet.2010.10.011>.
- Ritzema, H.P. (ed.). 1994. *Subsurface Flow to Drains*. *Subsurface Flow to Drains*, 236–304. Wageningen: International Institute for Land Reclamation and Improvement.
- Ritzema, H., S. Limin, K. Kusin, J. Jauhainen, and H. Wösten. 2014. Canal Blocking Strategies For Hydrological Restoration of Degraded Tropical Peatlands in Central Kalimantan, Indonesia. *CATENA* 114: 11–20. <https://doi.org/10.1016/j.catena.2013.10.009>.
- Sodhi, N.S., M.R.C. Posa, T.M. Lee, D. Bickford, L.P. Koh, and B.W. Brook. 2010. The State and Conservation of Southeast Asian Biodiversity. *Biodiversity and Conservation* 19: 317–328. <https://doi.org/10.1007/s10531-009-9607-5>.
- Sun, G., K. Alstad, J. Chen, S. Chen, C.R. Ford, G. Lin, C. Liu, N. Lu, et al. 2011. A General Predictive Model for Estimating Monthly Ecosystem Evapotranspiration. *Ecohydrology* 4: 245–255. <https://doi.org/10.1002/eco.194>.
- Tacconi, L. 2016. Preventing Fires and Haze in Southeast Asia. *Nature Climate Change* 6: 640–643. <https://doi.org/10.1038/nclimate3008>.
- Takeuchi, W., T. Hirano, and O. Roswintarti. 2016. Estimation Model of GroundWater Table at Peatland in Central Kalimantan, Indonesia. In *Tropical Peatland Ecosystems*, 9th ed, ed. M. Osaki and N. Tsuji. Tokyo, Heidelberg, New York, Dordrecht, London: Springer.
- Taufik, M., B.I. Setiawan, and H.A.J. van Lanen. 2015. Modification of a Fire Drought Index for Tropical Wetland Ecosystems by Including Water Table Depth. *Agricultural and Forest Meteorology* 203: 1–10. <https://doi.org/10.1016/j.agrformet.2014.12.006>.
- Taufik, M., P.J.J.F. Torfs, R. Uijlenhoet, P.D. Jones, D. Murdiyarso, and H.A.J. Van Lanen. 2017. Amplification of Wildfire Area Burnt by Hydrological Drought in the Humid Tropics. *Nature Climate Change* 7: 428–431. <https://doi.org/10.1038/nclimate3280>.
- Van Dam, J.C., and R.A. Feddes. 2000. Numerical Simulation of Infiltration, Evaporation and Shallow Groundwater Levels with the Richards Equation. *Journal of Hydrology* 233: 72–85. [https://doi.org/10.1016/S0022-1694\(00\)00227-4](https://doi.org/10.1016/S0022-1694(00)00227-4).
- Van Dam, J.C., P. Groenendijk, R.F.A. Hendriks, and J.G. Kroes. 2008. Advances of Modeling Water Flow in Variably Saturated Soils with SWAP. *Vadose Zone Journal* 7: 640. <https://doi.org/10.2136/vzj2007.0060>.
- Van Genuchten, M.T. 1980. A Closed-Form Equation for Predicting the Hydraulic Conductivity of Unsaturated Soils. *Soil Science Society of America Journal* 44: 892–898.
- Verrot, L., and G. Destouni. 2016. Data-Model Comparison of Temporal Variability in Long-term Time Series of Large-Scale soil Moisture: Results from an Analytical Framework. *Journal of Geophysical Research: Atmospheres* 121: 10056–10073. <https://doi.org/10.1002/2016JD025209>.
- Wooster, M.J., G.L.W. Perry, and A. Zoumas. 2012. Fire, Drought and El Niño Relationships on Borneo (Southeast Asia) in the pre-MODIS Era (1980–2000). *Biogeosciences* 9: 317–340. <https://doi.org/10.5194/bg-9-317-2012>.
- Wösten, J.H.M., E. Clymans, S.E. Page, J.O. Rieley, and S.H. Limin. 2008. Peat–Water Interrelationships in a Tropical Peatland Ecosystem in Southeast Asia. *Hydropedology*. <https://doi.org/10.1016/j.catena.2007.07.010>.
- Wösten, H., A. Hooijer, C. Siderius, D.S. Rais, A. Idris, and J. Rieley. 2006. Tropical Peatland Water Management Modelling of the Air Hitam Laut Catchment in Indonesia. *International Journal of River Basin Management* 4: 233–244. <https://doi.org/10.1080/15715124.2006.9635293>.
- Yule, C.M. 2010. Loss of Biodiversity and Ecosystem Functioning in Indo-Malayan Peat Swamp Forests. *Biodiversity and Conservation* 19: 393–409. <https://doi.org/10.1007/s10531-008-9510-5>.

AUTHOR BIOGRAPHIES

Muh Taufik (✉) is a Lecturer at Bogor Agricultural University Indonesia. His research interests include Drought and Associate Impacts, Forest Fire, and Ecophysiology of Humid Tropics. Address: Department of Geophysics and Meteorology, Bogor Agricultural University, Bogor 16680, Indonesia. e-mail: mtaufik@ipb.ac.id

Budi I. Setiawan is a Professor of Soil Physics and Hydrology at Bogor Agricultural University. His research interests include Irrigation, Drainage, Stormwater Harvesting, and Water Resource Management. Address: Department of Civil and Environmental Engineering, Bogor Agricultural University, Bogor 16680, Indonesia. e-mail: budindra@ipb.ac.id

Henny A. J. Van Lanen is an Associate Professor of Hydrogeology at Wageningen University and Research Centre. His research interest includes hydrological drought and associated impacts. He is a Coordinator of the European Drought Centre and other International Projects and Networks. Address: Hydrology and Quantitative Water Management Group, Wageningen University, 6708PB Wageningen, The Netherlands. e-mail: henny.vanlanen@wur.nl