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WETLAND RESTORATION

Application of Palaeoecological and Geochemical Proxies in the Context of Tropical Peatland Degradation and Restoration: A Review for Southeast Asia

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Abstract

Tropical peatlands in Southeast Asia (SEA) have undergone large-scale degradation in recent times due to extensive land use changes and drainage associated with their conversion for economic gains, and resulting fires during dry periods. This has had detrimental impacts on key peatland ecosystem processes and services such as hydrology, peat formation, carbon storage, fire prevention and biodiversity. Palaeoecological and geochemical proxies have been increasingly used in tropical peatland studies to extend contemporary instrumental records of peat conditions. Despite not yet being used to actively inform tropical peatland degradation and restoration interventions, these proxies are able to provide long-term trends in responses, resilience (threshold) and feedback processes of vegetation dynamics, groundwater level, peat pH, peat decomposition and accumulation rates, and degradation history. In this review, through the assessment of relevant tropical peatland studies in SEA, the palaeoecological and geochemical proxies were evaluated for their potential to reconstruct long-term peatland responses to climatically and anthropogenically-driven degradation. This information can potentially be utilised to provide better understanding of the extent of degradation and assist with the development of restoration management plans in SEA through its application in peat-hydrology restoration models.

Keywords Tropical peatlands · Restoration · Drivers of degradation · Palaeoecology · Palynomorphs · Testate Amoeba · Charcoal · Geochemistry

1. Introduction

Rapid large-scale degradation of tropical peatlands in Southeast Asia (SEA) has occurred in recent times due to economic pressures to convert these ecosystems to plantation, agriculture and human settlement, logging disturbance, and associated drainage and fire (Murdiyarso et al. 2019; Green and Page 2017; Page et al. 2011; Dommain et al. 2011). In Malaysia, Sumatra and Borneo, land use changes, which have involved deforestation and drainage, have resulted in swamp forest cover on peatlands (15.7 Mha) dropping from 76 to 29% between 1990 and 2015 (Miettinen et al. 2016). This anthropogenically-driven degradation is exacerbated by drought episodes and wildfires associated with El Niño events over this period (Page and Hooijer 2016). The culmination of this degradation has resulted in only 6% of remaining SEA tropical peatland area being considered to be “pristine peat-swamp forest” as of 2015 (Miettinen et al. 2016). The impacts of this degradation include carbon

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storage loss (Page et al. 2004, 2011), carbon emissions (Rieley et al. 2008; Hirano et al. 2007; Hooijer et al. 2006), modifications in hydrology and increased drainage (Ritzema et al. 2014; Rieley et al.,19961996) and subsequent forest fires (Page and Hooijer 2016; Osaki et al. 2016), biodiversity losses (Posa et al.,2011; Yule 2010) and public health risks (Marlier et al. 2019). To address these impacts, SEA governments alongside non-governmental organisations (NGOs) have developed restoration initiatives such as rewetting, revegetation and revitalisation to raise the groundwater level, replant native species and develop more peat-friendly local economies in degraded peatlands to limit peat decomposition and subsidence, as well as decrease fire risks (Giesen and Nirmala 2018; Dohong et al. 2018; Ritzema et al. 2014).

Past studies of tropical peatland degradation and restoration efforts in SEA have showed success in quantifying and mitigating negative impacts of degradation (e.g. Ritzema et al. 2014; Husen et al. 2014; Dohong and Lilia 2008; Limin et al. 2008). However, such studies rarely collect and assess data on peatland ecosystem condition prior to degradation as they were conducted in response to the 'crisis' of the disturbance. Further, as the peat, hydrological and biological components of a peatland ecosystem are interconnected (e.g. Harrison 2013), degradation such as deforestation will have a direct impact on vegetation dynamics as well as potential (lagged) impacts on runoff/groundwater level and peat decomposition over time (Price et al. 2016; Quinton and Hayashi 2005). As such, estimates of carbon losses from deforestation may be improved by including losses from the interconnected processes of oxidative decomposition of exposed peat and fluvial carbon leached from degraded peat (Jauhiainen et al. 2016; Hirano et al. 2014; Moore et al. 2013). Furthermore, assessment of the effectiveness of peatland restoration efforts requires definition of, and comparison to, appropriate pre-degradation and target reference conditions, which are not always available (Harrison et al. 2020; Graham et al. 2017). More studies are therefore needed on the functioning of relatively 'pristine' tropical peatland ecosystems prior to the impacts of degradation in order to establish these baselines.

To obtain pre-degradation baselines of peat, hydrological and vegetation systems, palaeoecological and geochemical analysis have been applied in some temperate/boreal peatland studies (e.g. Wingard et al. 2017; Chambers et al. 2012). Palaeoecological proxies in peat, specifically palynomorphs (Gałka et al. 2017; Lavoie et al. 2001) and testate amoeba (Kurina and Golovatskaya 2018; Booth et al. 2010) are identified to infer past vegetation dynamics and hydrological conditions such as changes in groundwater level and sea level. Stable carbon isotope ($\delta^{13}C$) is also used to ascertain the abundance of C3 plants (i.e. tree species) and C4 plants (i.e. grassland species), while organic carbon concentration is used to infer peat (carbon) accumulation rates (Rao et al. 2019; Tareq et al. 2007). In contrast, nitrogen isotope ($\delta^{15}N$) is left behind during organic matter decomposition and enriched levels in peat substrate have been used to postulate peat degradation (Krüger et al. 2015; Andersson et al. 2012). In addition, regional and local fire history is revealed using micro- (particles < 125 µm) and macro-charcoal (particles > 125 µm) analyses, respectively (Norström et al. 2009; Lavoie et al. 2001). The degree of peat disturbance has been inferred from the concentration and ratios of magnetic susceptibility (k) and elemental Silicon, Titanium, Aluminum, Potassium, Iron and Magnesium (Andersson et al. 2012; Kuhry and Vitt 1996). Peat dating techniques using radiocarbon and radionuclide (Lead-210 and Cesium-137) dating are also applied together to provide ages for alterations in peatland condition that may be associated with degradation (Appleby 2008; Turetsky et al. 2004).

Although these proxies have been applied to studies reconstructing long-term hydroclimate and environmental characteristics in tropical peatlands, the explicit goal of using proxies to inform assessments of the impacts of degradation, and testing and proposing restoration interventions have not been achieved (Hapsari et al. 2017; Cole et al. 2015; Biagioni et al. 2015; Tareq et al. 2007; Yulianto and Hirakawa 2006; Morley 1981; Anderson and Muller 1975). Hence, this study aims to assess the application of palaeoecological and geochemical proxies in tropical peatlands to reveal long-term hydrological, peat and vegetation responses to anthropogenic and climate-driven degradation that can help inform contemporary restoration efforts. In particular, we ask: (i) What can palaeoecological and geochemical proxies tell us about changes in peatland conditions in response to past drivers of disturbance? (ii) How can long-term hydrological, peat and vegetation conditions inferred from these proxies help inform contemporary peatland degradation assessment and restoration efforts? and (iii) How can such long-term data be incorporated into peat-hydrological models (e.g. HPMTrop and DigiBog) to project the responses and pathways of planned tropical peatland restoration activities? To answer these questions, we review peer-reviewed studies describing the application of relevant proxies to infer peatland conditions, and the assessment of recent degradation and restoration efforts in tropical peatlands in SEA.

Table 1 Peatland variable indicators and challenges of key proxies in inferring tropical peatland ecosystem responses to disturbance. Further details of the proxies and discussion of their challenges are provided in Sect. 2

Proxy type	Peatland variable indicated by proxy	Challenges of proxy in indicating degree of ecosystem degradation
Palynomorphs	Vegetation dynamics, moisture conditions and peat types.	Biological traits of some vegetation species can promote or inhibit pollen accumulation in peat.
Testate amoeba (TA)	Hydrological conditions (e.g. groundwater level and pH of peat).	Limited tropical peatland literature on TA genera/morphospecies and their habitat preferences (e.g. hydrology and peat conditions).
Micro- and macro- charcoal	Regional and localised forest fires.	Charcoal concentration may not be fully indicative of the impacts of fires on vegetation as peat swamp forest species can vary in their fire resistance.
Stable isotope ratios, and concentrations of carbon and nitrogen	Degree of peat decomposition and accumulation, and moisture conditions.	Other processes of ^{13}C enrichment are methanogenesis and biodegradation, while TN and ^{15}N can be enriched by rainfall and decomposition in standing water with large composition of leaf litter.
Magnetic susceptibility and elemental concentration	Peat disturbances through differentiating between authigenic and diagenetic deposition in peat.	Possibility of remobilisation and precipitation of these elements through redox-related diagenesis.
Dating techniques (^{14}C, ^{210}Pb and ^{137}Cs)	Peat accumulation ages to ascertain when environmental changes occurred.	Possible errors to ages when peat and plant matter are remobilised due to bioturbation, reservoir effects and/or contamination.

Assessment of the Application of Palaeoecological and Geochemical Proxies

The palaeoecological and geochemical proxies described in this review are well-preserved in waterlogged and anaerobic settings, and have been applied in studies attempting to reconstruct a variety of tropical peatland characteristics over time (e.g. Hapsari et al. 2021; Cheng et al. 2020; Hapsari et al. 2017; Cole et al. 2015; Biagioni et al. 2015; Tareq et al. 2007; Yulianto and Hirakawa 2006; Morley 1981; Anderson and Muller 1975). These proxies are grouped into six types, for which the key indicators and challenges associated with their use as indicators of peatland characteristics and ecosystem degradation are summarised in Table 1 .

Pollen and Spores (Palynomorphs)

Pollen grains and pteridophyte spores (collectively described as palynomorphs) are produced in large quantities as part of the natural reproductive cycles of plants and are released into the environment through their dispersal methods such as wind, animals and water (Birks and Birks 1980). The palynomorphs that are not utilised in plant reproduction can be preserved in peat sediments as microfossils due to the decay-resistant properties of the palynomorph surface (known as exine) (van der Kaars 1991; Birks and Birks 1980). These microfossils reflect the local vegetation present as the closed (dense) canopies of tropical peatlands will limit the deposition of palynomorphs beyond the local environment (Morley 1981; Anderson and Muller 1975). However, palynomorphs can be deposited more widely via wind and water in deforested peatlands (open canopies) and through flooding events (fluvial transport) (Anderson and Muller 1975). By identifying and counting fossilised palynomorphs in the peat cores (van der Kaars 1991; Moss 2013), palynological studies in tropical peatlands are able to infer long-term vegetation dynamics, changes in groundwater level and disturbances to peat swamp forest over time (i.e. peatlands cleared for agriculture and forestry) (Hapsari et al. 2017; Biagioni et al. 2015; Biagioni 2015; Cole et al. 2015; Hunt et al. 2012; Yulianto and Hirakawa 2006; Anshari et al. 2001; Morley 1981; Haseldonckx 1977; Anderson and Muller 1975).

The taxonomy of fossilised palynomorphs identified is used to classify the peatland into ecological groups relating to the information the palynomorphs provide regarding the forest vegetation, and the prevalent hydrological and peat conditions in which the vegetation was growing (Biagioni et al. 2015; Cole et al. 2015; van der Kaars 1991). The past hydrological and peat conditions can be inferred as different vegetation species have different tolerances to groundwater level and peat types, in particular where nutrient influx to the system occurs via water from the river catchment (minerotrophic peat) or exclusively via rainfall (ombrotrophic peat) (Rieley et al. 2008; Page et al. 1999; Cameron et al. 1989). This review identifies eight main ecological groups based on palynomorph taxonomy identified in SEA peatland studies (Supplementary data): coastal vegetation, upland forest (dry), lowland forest (dry), freshwater swamp forest (minerotrophic, wet), lowland vegetation mixed with swamp forest (fluctuating wet and dry), peat swamp forest (ombrotrophic, wet), pioneer swamp forest (early succession plants), and open canopied vegetation (degraded environment) (Cheng et al. 2020; Cole et al. 2015; Biagioni et al. 2015; Hunt et al. 2012; Yulianto and Hirawaka, 2006). The transitions between ecological groups in a peat record are identified through statistical clustering analysis of taxonomy data to detail when vegetation dynamics shifted significantly in response to changes in environmental conditions (Biagioni et al. 2015; Grimm et al. 2013; Grimm 1987).

Environmental conditions such as hydrological and sea level changes in tropical peatlands can be detected from shifts in the dominant ecological groups (linked to palynomorph diversity). Ecological group transitions from coastal forest, to peat swamp, to upland/ lowland forest over time are used to postulate sea-level fall and change to drier conditions (Biagioni et al. 2015; Yulianto and Hirawaka, 2006; Caratini and Tissot 1988; Anderson and Muller 1975). For example, studies conducted in inland peatlands showed decreasing levels of coastal vegetation when sea-level fell (Morley 1981; Anderson and Muller 1975) and decreasing trends of peat/freshwater swamp during strong El Niño events (droughts) (Biagioni et al. 2015). However, in coastal peatlands, Dommain et al. (2011) and Cole et al. (2015) detected minimal changes in the local peat swamp forest vegetation during sea-level fall and El Niño events due to permanent humid rainfall regimes with low seasonality in their study sites.

To infer peat types, the shift in vegetation from freshwater swamp to peat swamp forest taxa is used as an indicator of the transition from minerotrophic to ombrotrophic peat due to change in the source of nutrients (Biagioni et al. 2015; Hunt et al. 2012). Furthermore, disturbances to the peatlands such as fire events and land clearance are interpreted from the dominant presence of open-canopied palynomorphs in peat such as *Poaceae, Cyperaceae*, ferns and shrubs (van der Kaars 1991; Caratini and Tissot 1988; Haseldonckx 1977). There may also be an increase in aeolian derived palynomorphs (i.e. from taxa such as *Podocarpus*, *Agathis*, *Oncosperma*, *Casuarina* and *Stenochlaena*) in the record, as palynomorphs derived from plants growing within the region can be more readily distributed in the peat sediments in open-canopied compared to close-canopied forest conditions (Biagioni et al. 2015; Morley 1981; Anderson and Muller 1975).

Despite the vast potential of palynomorphs, the biological traits of some vegetation species and peat disturbances to the peat substrate, can promote or inhibit pollen accumulation in peat, which can distort the palaeoenvironmental reconstruction (Cole et al. 2015; Anshari et al. 2001; Anderson and Muller 1975) reiterated that some vegetation species will contribute to high localised pollen accumulation in peat due to their abundant pollen production (e.g. *Campnosperma* and *Rhizophora*) and/or being predominantly insect-pollinated (e.g. *Ficus* and *Dipterocarpaceae*) in close-canopied forest, which limits horizontal transport. The use of pollen taxa to interpret groundwater level therefore has to be conducted with caution as there are gaps in the literature concerning the relationships between pollen production and moisture availability (Anderson and Muller 1975). In addition, Anshari et al. (2001) found that low palynomorph concentration could be due to discontinuous peat layers caused by erosion or non-deposition at specific sites. Hence, it is important that the interpretations made using palynomorphs are verified by other proxies, sediment context, the site's contemporary ecological characteristics and documented historical archives (i.e. human arrivals and anthropogenic degradation) (Hapsari et al. 2021; van der Kaars 1991).

Testate Amoeba (TA)

Testate amoeba (TA) are a group of amoeboid protozoan that occur naturally in a range of environments including lakes and peatlands (Liu et al. 2019; Swindles et al. 2014). TA are made up from either proteinaceous, calcareous or siliceous material, and are small in size (approximately 20 to 200 μm) (Mitchell et al. 2008). Modern TA live in abundance on peat surfaces and comprise up to 30% of the microbial biomass (Swindles et al. 2014), while subfossil TA are preserved in peat through gluing to organic and mineral particles (Mitchell et al. 2008). As the taxonomy (genus), and morphology and size (morphospecies) of TA are primarily influenced by hydrological conditions (Mitchell et al. 2008; Charman 1999), modern and subfossil TA are extracted from peat in tropical/sub-tropical peatland to reconstruct past groundwater level and peat pH (Krashevska et al. 2020; Liu et al. 2019; Hapsari et al. 2017; Swindles et al. 2014, 2016; Biagioni et al. 2015; Song et al. 2014; Qin et al. 2012).

Modern TA in surface peat samples provides information on the relationships between TA assemblages (genus and morphospecies), and their preferred environmental variables (Liu et al. 2019). To establish these relationships, modern TA are sampled from different micro-habitats with known groundwater levels and peat pH, such as waterbodies, regularly flooded areas, hollows, hummocks and soil (Liu et al. 2019; Song et al. 2014; Swindles et al. 2014; Qin et al. 2012). Transfer function models using weighted average partial least square regression between modern TA assemblages, groundwater level and peat pH are developed to reconstruct past conditions from the subfossil TA assemblages in peat cores (Liu et al. 2019; Swindles et al. 2018; Qin et al. 2012; Birks 1995). Based on the available tropical and sub-tropical peatland studies, these transfer function models are statistically significant between TA assemblages and groundwater levels in ombrotrophic peatlands, and between TA assemblages and peat pH in minerotrophic peatlands (Biagioni 2015; Swindles et al. 2014, 2016; Qin et al. 2012). The differences in the relationships between TA and environmental variables could be due to the characteristics of minerotrophic peat as nutrient-rich as it receives water input from catchment waterways while ombrotrophic peat receives water input only from rainfall (Booth et al. 2010; Markel et al. 2010). Thus, this leads to changes in TA assemblages in response to fluctuations of pH in minerotrophic peat and groundwater level in ombrotrophic peat.

Although some studies showed poor transfer function relationships, the diversity of the TA assemblages in peat was still used to explain hydrological changes using the classification to ecological groups method (Krashevska et al. 2020; Biagioni 2015; Mitchell et al. 2008). Based on the findings of tropical/sub-tropical peatlands studies on the preference of TA assemblages to certain hydrological conditions and habitats, this review identifies four common ecological groups of TA taxa, characterising wet, dry, hydrologically variable (fluctuating groundwater level) and other conditions (no direct hydrological trends) (Supplementary data) (Krashevska et al. 2020; Liu et al. 2019; Swindles et al. 2016, 2018; Biagioni 2015). TA assemblages are classified to these ecological groups to illustrate (changes in) the groundwater level conditions over time (Biagioni 2015; Mitchell et al. 2008). Furthermore, size variations of TA, such as *Hyalosphenia subflava* (small: <75 µm, medium: 75 to 105 µm and large: >105 µm), have been found to have an indirect relationship to moisture availability in peat, with larger sizes representing higher groundwater levels (Krashevska et al. 2020; Mitchell et al. 2008).

Despite the potential of TA to infer hydrological conditions, this proxy has not been widely applied to tropical peatlands in SEA due to limited literature on the TA present and its classification into ecological groups (Biagioni 2015; Swindles et al. 2014; Mitchell et al. 2008; Charman 1999). Furthermore, the use of transfer function models includes assumptions regarding the contribution of other environmental variables to the relationship between TA and groundwater levels/peat pH through time (Liu et al. 2019; Song et al. 2014; Qin et al. 2012). Studies in temperate peatlands have also suggested that some TA species responded to extreme hydrological changes caused by anthropogenic and climatic factors rather than seasonal variations (i.e. wet and dry season) (Beyens et al. 2008; Mitchell et al. 2008). As a result, larger spatial sampling of modern TA is needed to establish robust transfer function models between TA and environmental variables for this approach to be applied readily and widely in tropical peatlands (Swindles et al. 2014; Song et al. 2014).

Micro- and Macro-Charcoal Concentration

Charcoal in sediments are inorganic carbon particles that are produced during the incomplete combustion and pyrolysis of plant materials when oxygen becomes scarce (Braadbaart and Poole 2008; Patterson et al. 1987). The resulting charred product consists of mainly lignin from plant matter and is well preserved in peat after fire events due to its resistance to oxidation, microbial activity and further thermal degradation (Scott 2000; Pyne et al. 1996). Mathematical models based on modern fires indicate that charcoal particles of more than 125 µm in diameter are deposited close to fire margins (i.e. derived from localised fires), while charcoal sizes of less than 125 µm are transported by aeolian processes and deposited away from the fire margins to more than 30 km, depending on fire intensity (i.e. derived from regional fires) (Clark et al. 1998; Clark and Patterson 1997; Whitlock and Millspaugh 1996). Hence, palaeofire studies use micro-charcoal concentration (< 125 µm) to indicate regional fires and macro-charcoal concentration (> 125 µm) to indicate localised fires in tropical peatlands (Biagioni et al. 2015; Cole et al. 2015; Norström et al. 2009; Hope et al. 2005; Yulianto et al. 2004; Anshari et al. 2001; Lavoie et al. 2001).

Micro- and macro-charcoal particles in peat cores can be counted in order to calculate the charcoal concentration, which can then be multiplied by the sedimentation rate to obtain charcoal accumulation rate (Biagioni et al. 2015; Moss 2013; Stevenson and Haberle 2005). Characteristics of fire regimes, such as fire return intervals and peaks in fire magnitude, can be determined using algorithms developed by Higuera et al. (2009), which can be applied to fossil charcoal data using the statistical software, CharAnalyis. Furthermore, the transitions of fire magnitude over time can be obtained using clustering analysis to identify periods of high and low fire incidence (Grimm 1987). Fire frequencies can also be calculated using spectral analysis to obtain the number of peak fire events through removing background fire levels (Higuera et al. 2009; Biagioni et al. 2015). In addition, ratios of charcoal and palynomorph concentration are used to infer the reasons behind fire events (i.e. natural fires or land clearance associated with anthropogenic activities). This can be observed by peaks in charcoal coinciding with low production of palynomorphs, followed by peaks in open-canopied and agricultural palynomorphs that could indicate slash-and-burn clearings for plantation or other land use changes (Cole et al. 2015; van Eijk et al. 2009; Page et al. 2009).

The majority of studies using charcoal proxies reiterated that fire has been a common occurrence in tropical peatlands over the past 30,000 years (Biagioni et al. 2015; Yulianto et al. 2004; Anshari et al. 2001). However, the reconstructed fire record indicates fire frequency and intensity in SEA tropical peatlands increased rapidly in the late Holocene likely due to anthropogenic factors (Hapsari et al. 2017; Biagioni et al. 2015; Cole et al. 2015). In addition to human activities, climate-induced fires can be observed from strong positive relationships between charcoal concentration and hydroclimate conditions, especially during drought periods caused by El Niño events (Biagioni et al. 2015). The relationships between charcoal concentration, and recent carbon emissions and biomass loss were also used in temperate peatlands to reconstruct long-term estimates of carbon loss during fire events (Hawthorne et al. 2018; Carcaillet et al.,2002).

However, long-term fire reconstruction from charcoal in peat cores can be influenced by other factors (Cole et al. 2015; Hope et al. 2005). For example, Hope et al. (2005) observed from fossil charcoal records in Kutai Peatlands, Indonesia that anthropogenically-influenced palaeofire events were rare in forests remote from rivers in older peat (> 3000 years BP) and were more frequent to sites accessible to waterways. Forest fires could also spread unevenly based on the localised hydrological conditions (i.e. groundwater level variation), which may result in some palaeofires not being captured in sampled peat cores (Giesen and Nirmala 2018). Lastly, the use of palynomorphs and charcoal concentration ratios to ascertain the impacts of fires, and distinguish between anthropogenic and climatic-driven fires, should consider the caveat that some peat swamp forest species can be resilient to fire events (Cole et al. 2015; Jiang et al. 2008).

Geochemical Analyses

Organic and inorganic geochemical constituents enter peat deposits through allogenic (externally derived through wet/dry deposits) and authigenic (internally derived through chemical precipitation) processes, which are dependent on the prevailing environmental conditions (Weiss et al. 2002; Neuzil et al. 1993). Allogenic deposits include solid particles from detrital influx (plant matter), airborne terrestrial dusts and volcanic ash (Neuzil et al. 1993), and dissolved minerals from marine aerosols, seawater, rainfall, river discharge and groundwater (Siegel and Glaser 1987). In contrast, the authigenic deposits are the result of mobilisation, chemical dissolution and precipitation of mineral matter, cation exchange, chelation, and the decay/growth of plants and microbes (Neuzil et al. 1993; Hill and Siegel 1991). Both allogenic and authigenic geochemical deposits are preserved in low pH and organic-rich peat, and can be used to infer peat accumulation/decomposition, disturbances and ages (Lampela et al. 2014; Sjögersten et al. 2011; Muller et al. 2008; Appleby 2008; Turetsky et al. 2004; Yulianto et al. 2004; Weiss et al. 2002; Hong et al. 2001; Neuzil et al. 1993). The geochemical proxies reviewed here are carbon and nitrogen stable isotopes and absolute concentrations, magnetic susceptibility and elemental concentration, as well as chronological dating techniques.

Carbon and Nitrogen Stable Isotopes and Absolute Concentration

Carbon and Nitrogen minerals enter peat samples from both allogenic and authigenic processes. Stable isotope ratios ($\delta^{13}C$ and $\delta^{15}N$) and absolute concentrations of carbon and nitrogen (TC and TN) have been applied in many tropical peatland studies to infer past vegetation composition, moisture conditions and degree of peat decomposition/accumulation (Fig. 1) (Krüger et al. 2015; McClymont et al. 2010; Hong et al. 2001; Kuhry and Vitt 1996). The TC, TN, $\delta^{13}C$ and $\delta^{15}N$ are usually analysed in bulk peat samples, although some studies recommended the use of peat biomarkers such as lignin for stable isotopic analysis as it is resistant to biodegradation and methanogenesis (Killops and Killops 2013; Steinbeiss et al. 2006; Glaser 2005).

The $\delta^{13}C$ has been used successfully in tropical peatland studies to distinguish between C3 and C4 vegetation, as C3 plants (i.e. tree species) exhibit values ranging from – 30 to -20‰, while C4 plants (i.e. grassland species) have values from – 16 to -9‰ (Tareq et al. 2007; Hong et al. 2001). Furthermore, some studies utilised the higher values of $\delta^{13}C$ in bulk peat samples to postulate lower moisture conditions, as pathways for photosynthesis in C3 plants during water-stress conditions produced ^{13}C-sugar (^{12}C-sugar is produced during well-watered conditions) in plant matter that was ultimately accumulated into peat (Fig. 1) (Fiorentino et al. 2015; Hong et al. 2001). However, the use of $\delta^{13}C$ to represent dry conditions has limitations, as it can be produced during methanogenesis of peat in anaerobic conditions or biodegradation in charred/burnt peat samples, and the $\delta^{13}C$ also varies in different vegetation and plant parts (Fig. 1) (Osaki et al. 2021; Andersson et al. 2012; Muller et al. 2008). To overcome this limitation, some studies have used the $\delta^{13}C$ of lignin to represent the moisture conditions at that particular time (Steinbeiss et al. 2006; Glaser 2005).

In addition, TC was used to quantify peat sequestration (i.e. carbon accumulation) in tropical peatlands, as organic matter decomposition is restricted by low micro-organism activity in anaerobic conditions (Page et al. 2011; Yulianto et al. 2004; Kuhry and Vitt 1996). In contrast, the degree of peat decomposition was quantified by TN and ^{15}N, as the decomposition of peat in aerobic conditions promotes the utilisation of ^{14}N isotopes by micro-organisms while leaving behind ^{15}N and TN-enriched peat (Fig. 1) (Krüger et al. 2015; Andersson et al. 2012). However, vegetation

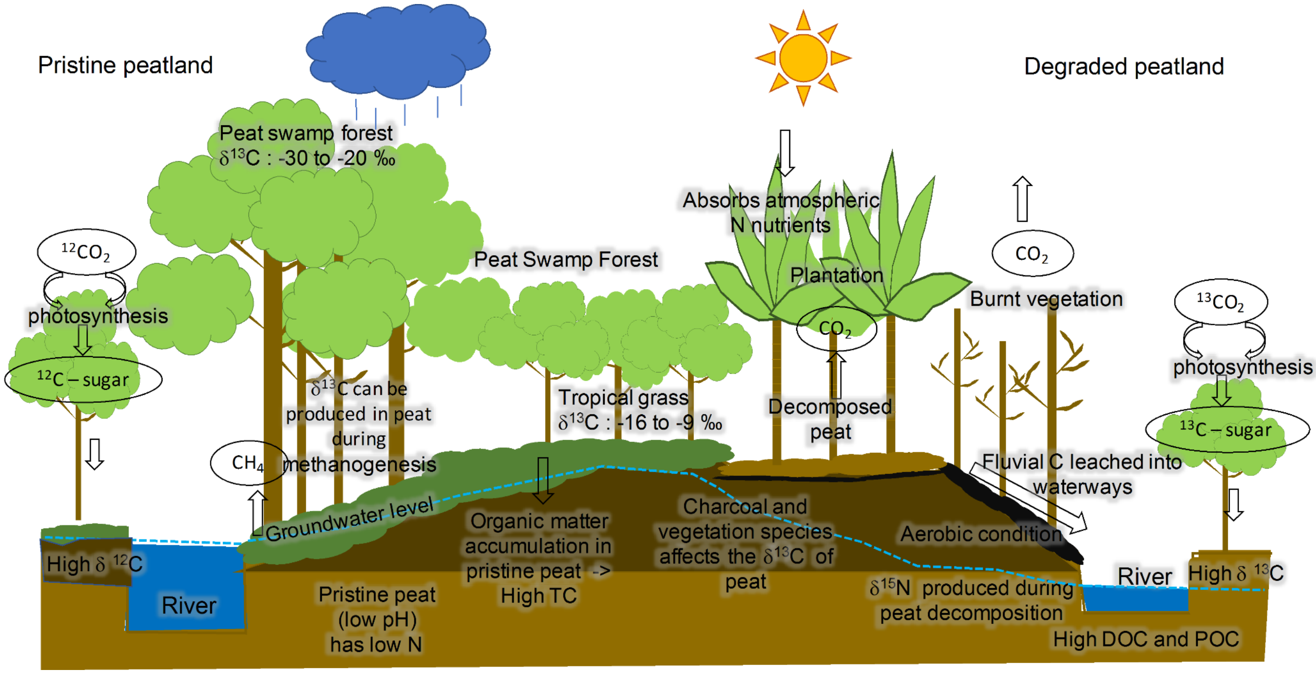

Fig. 1 Conceptual illustration of how Carbon and Nitrogen minerals (TC, $\delta^{13}C$, TN and $\delta^{15}N$) enter peat samples from both allogenic and authigenic processes, and are leached into the river as dissolved organic carbon (DOC) and particulate organic carbon (POC).

growing in aerobic peat conditions absorb N nutrients from the atmosphere instead of peat substrate due to the preferential absorption by plants for lighter ^{14}N, which results in low $\delta^{15}N$ (< 3.0‰) in plant foliage and leaf litter (Fig. 1) (Osaki et al. 2021). Further information on the peat conditions were inferred from the ratio of carbon and nitrogen concentrations, in which high values reflect peat accumulation while low values reflect peat decomposition (Andersson et al. 2012; Sjögersten et al. 2011).

Given the wide use of carbon and nitrogen proxies to quantify peat accumulation and decomposition in tropical peatland studies, we summarised the multiple interpretations of TC, TN, $\delta^{13}C$ and $\delta^{15}N$ in peat (Table 2). For example, peaks in TC could represent fire events instead of peat accumulation, as the presence of charcoal may contribute to the higher carbon content (Hapsari et al. 2017; Biagioni 2015; Yulianto et al. 2004). Furthermore, TN and ^{15}N are not only enriched by peat decomposition, but can also be enriched by rainfall, leaf litter decomposition in standing water and combustion of organic matter in fire events (Fiorentino et al. 2015; Muller et al. 2008; Turner et al. 2007; Yulianto et al. 2004). To overcome these limitations, TC, TN, $\delta^{13}C$ and $\delta^{15}N$ proxies should be verified with other proxies to ensure consistent interpretations of peat conditions.

Magnetic Susceptibility and Element Concentrations

In addition to Carbon and Nitrogen, concentrations of other elements in peat, such as Calcium (Ca), Magnesium (Mg), Aluminum (Al), Silicon (Si), Manganese (Mn), Potassium (K), Titanium (Ti) and Iron (Fe) have been used to classify peat types (ombrotrophic or minerotrophic), and identify disturbances such as leaching, groundwater level changes and geochemical deposition (Lampela et al. 2014; Muller et al. 2008; Weiss et al. 2002; Neuzil et al. 1993; Cameron et al. 1989). Furthermore, the use of magnetic susceptibility (κ) and element concentrations has the potential to differentiate between authigenic (internally derived) or diagenetic overprinting (through post-depositional processes) deposition in peat (Rothwell and Croudace 2015). The interpretations of relevant studies on the elemental concentration and magnetic susceptibility in relation to tropical peatland conditions are summarised in Table 3.

Some studies have utilised the Ca/Mg ratio to distinguish between minerotrophic (high ratio) and ombrotrophic (low ratio) peat, as Ca^{2+} is enriched in water and nutrient sources from rivers, compared to lower levels in continental/marine rainfall (Muller et al. 2008; Weiss et al. 2002). Furthermore, the transition to low values of Ti, Si and Al concentration from the top to the bottom of peat profiles represents the depth of groundwater saturated peat (catotelm) and organic-rich peat (Neuzil et al. 1993). Alternatively, some studies utilised Mn concentration trends to determine the groundwater level, as Mn has high solubility in anaerobic conditions that results in leaching of Mn (leading to low concentration in peat), and low solubility in aerobic conditions that causes precipitation of Mn (leading to high concentration in peat) (Muller et al. 2008). In addition, peaks in

Table 2 The range of interpretations of stable isotope and concentration of carbon and nitrogen to infer characteristics of peatlands environments

Geochemical proxy	Source	Interpretations		
		#1	**#2**	**#3**
Carbon isotopes	Hong et al. 2001; McClymont et al. 2010; Andersson et al. 2012; Osaki et al. 2021	Carbon isotope compositions ($\delta^{13}C$) differ between plant types: -30 to -20‰ for C3 plants and-16 to -9‰ for C4 plants.	Rainfall and groundwater level are negatively correlated to the $\delta^{13}C$ value in C3 plants/ peat due to the photosynthesis process in water-stress conditions.	Methanogenesis may enrich ^{13}C in residual material because methanogenic bacteria preferentiallymetabolize the ^{12}C-rich fraction of organic matter in anaerobic conditions.
Nitrogen isotopes	Andersson et al. 2012; Krüger et al. 2015; Osaki et al. 2021	During decomposition of peat, micro-organisms appear to utilise ^{14}N and leave behind ^{15}N - enriched peat.	Plants growing in decomposed peat (^{15}N and TN-enriched) absorb N nutrients by atmospheric N fixation due to the preference of plants for lighter ^{14}N,resulting in low $\delta^{15}N$ (< 3.0‰) in plant foliage and litter.	N fixation from soil/ peat varies for different plant species in peat swamp depending on the water level and presence of rhizobia (root nodule bacteria).
Carbon concentration; Carbon and Nitrogen ratio	Muller et al. 2008; Andersson et al. 2012; Osaki et al. 2021	Carbon concentration of peat used to calculate carbon density and storage in peatlands.	The C/N ratio of pristine peat indicates the degree of peat accumulation (high values) or decomposition (low values). However, this trend is not seen in degraded peat after land use change.	Carbon concentration can be high due to presence of charcoal and charred organic matter in the peat samples.
Nitrogen	Sjögersten et al.,2011; Andersson et al. 2012; Turner et al. 2007	N is low in organic-rich peat.	High N in peat is the result of nutrient turnover through litter inputs, peat decomposition and fire events.	An increase in N can be due to standing water (floods) while a decrease in N can be due to drainage (leaching).

K concentration were used as proxy of fire events, as K^+ is produced from biomass burning (i.e. vegetation and organic matter), which is preserved in degraded peat due to its high levels of exchangeable cations compared to organic-rich peat (Osaki et al. 2021; Weiss et al. 2002).

However, geochemical studies in tropical peatlands experience limitations due to the possibility of re-mobilisation and precipitation of these elements (Benavides et al. 2013; Weiss et al. 2002; Neuzil et al. 1993). For example, the use of Ca/Mg as a proxy of peat types can be disputed, as enriched values of Ca^{2+} in ombrotrophic peat could be due to remobilisation as a result of dissolving minerals from groundwater level increase (Muller et al. 2008). To distinguish diagenetic overprinting processes from the primary inputs in peat accumulation, low values of magnetic susceptibility (κ) and Ti are used as a conservative reference of organic-rich peat, while high values of Fe/κ are used as a proxy of post depositional processes, as Fe is redox-sensitive and mobile in anoxic conditions compared to other elements (Rothwell and Croudace 2015). As such, the peaks of κ, Fe/κ and/or κ /Ti ratios indicate redox-related diagenesis in peat, such as migration and recrystallisation of secondary chemical processes (Funk et al. 2003). Furthermore, the concentration ratios of Na/Ti, Ca/Ti, K/Ti and Fe/Ti are used to compensate for natural variations in atmospheric deposits (i.e. weathering of the Earth's crust) to extract changes in authigenic deposits of Na, Ca, K and Fe in peat (Muller et al. 2008; Weiss et al. 2002).

There are also alternative interpretations for fluctuations in element concentrations (Table 3) due to changes in local conditions such as proximity to the sea and hydrology. This includes different interpretations of Fe and Al concentration, which were found to be higher for inland peatlands compared to coastal peatlands (Neuzil et al. 1993). Furthermore, Fe was used as a proxy of atmospheric deposits in a peatland study in Indonesia instead of an indicator of anoxic conditions, as no changes were detected in Fe concentration in different groundwater level conditions (Weiss et al. 2002).

Geochronology: Radiocarbon and Radionuclides Dating (^{210}Pb and ^{137}Cs)

The common geochronological technique used in tropical peatland studies is radiocarbon (^{14}C) analysis for peat accumulated during "pre-bomb" (pre-1950AD) and "post-bomb" periods (post-1950 AD). Due to the nuclear event in 1963 which caused high levels of artificial radiocarbon (^{14}C) to be released into the atmosphere (Xu et al. 2006; Goslar et al. 2005), post-bomb samples are verified using radionuclides analysis of Lead-210 (^{210}Pb) and Cesium-137

Table 3 The range of interpretations of elemental proxies to infer characteristics of peatlands environments

Geochemical proxy (Elements)	Source	Interpretation #1	Interpretation #2
Iron (Fe)	Weiss et al. 2002; Muller et al. 2008;	Fe is immobile in degraded peat (i.e. low groundwater level and pH > 4) and can be used as a proxy for atmospheric deposits.	Fe is largely mobile in pristine peat under anoxic conditions and the peaks can indicate fluctuating groundwater level.
Calcium (Ca), Magnesium (Mg)	Weiss et al. 2002	High ratios of Ca/Mg reflect minerotrophic environments, as the inputs of Ca^{2+} are from dissolved solutes leached from the catchment and fluvial deposits.	Low ratios of Ca/Mg that are similar to atmospheric dust and rainfall reflect an ombrotrophic environment.
Silicon (Si), Aluminum (Al), Iron (Fe), Titanium (Ti),	Neuzil et al. 1993; Rothwell and Croudace 2015; Osaki et al. 2021	Decreasing Si, Al, Ti concentrations from the bottom to the middle of the peat column reflect peat formation and constant high groundwater level. Si, Al, Ti is low in peat soil compared to mineral soil, but Al (aluminosilicate) can be high at the surface peat from airborne dust.	The amounts of Al and Fe are greater in inland peat than in coastal peat deposits.
Potassium (K)	Weiss et al. 2002; Osaki et al. 2021	K^+ is produced during biomass burning and is enriched in degraded peat with elevated pH.	Pristine peat absorbs little K^+at low pH and K is leached continuously.
Phosphorus (P), Sulphur (S)	Neuzil et al. 1993; Weiss et al. 2002; Osaki et al. 2021	High concentration of S and P in surface peat at the root zone may be the result of recycling of plant matter and lifting of inorganic element constituents by leaf litter and live rootlets.	Fluctuations in S can indicate oscillating coastline as SO_4^{2-} increases in peat pore water when the bottom of peat layers contain pyrite. Pyrite minerals are found closer to coastline.
Manganese (Mn)	Muller et al. 2008	Under anaerobic conditions, Mn becomes soluble and is influenced by leaching, resulting in low concentrations.	In aerobic conditions, Mn is likely to precipitate in peat, resulting in high concentrations.
Titanium (Ti) Magnetic susceptibility (κ)	Weiss et al. 2002; Rothwell and Croudace 2015	Ti-bearing mineral such as (TiO_2) are resistant to chemical weathering and can be used as a conservative reference element of pristine peat.	High values of the ratios between Ti and Mg or Ca or K or Na or κ reflect that the elements are not naturally derived from soil but from post-depositional remobilization by groundwater, vegetation changes or fire events.

(^{137}Cs) (Cole et al. 2015; Biagioni et al. 2015; Hunt et al. 2012; Anshari et al. 2001).

Radiocarbon (^{14}C) analysis involves the measurement of ^{14}C specific activity of the organic carbon in bulk peat samples or micro/macro-fossils in peat (pollen, plant remains, wood, and charcoal) (Brown et al. 1992). The ^{14}C of the bulk peat depends on the degree of decay of organic matter in it, and the age of the peat is calculated through comparing to the specific activity and half-life of ^{14}C modern reference samples, plus the past atmospheric ^{14}C level at the time of deposition using calibration curves (e.g. IntCal12) (Reimer et al. 2013; Brown et al. 1992). However, the ^{14}C of the bulk peat samples might be variable as different plant parts and species exhibit different $^{14}CO_2$ uptake profiles due to discrimination against heavier isotope carbon (i.e. fractionation effect) (Turetsky et al. 2004; Hua et al. 2001). To overcome this, the fractionation effect can be corrected through measuring $\delta^{13}C$ of bulk peat to identify the plant composition and estimate the atmospheric ^{14}C level at the time of deposition (Turetsky et al. 2004; Brown et al. 1992). The ^{14}C measurements are accompanied by uncertainty errors from 40 to 160 years for pre-bomb peat and from 25 to 50 years for post-bomb peat (Trumbore 2000).

Radionuclide analyses of Lead-210 (^{210}Pb) and Cesium-137 (^{137}Cs) involve the counting of gamma emissions in peat sediment that were deposited from the atmosphere (atmospheric fallout) (Xu et al. 2006; Appleby 2008). Lead-210 atmospheric fallout is a natural occurrence and is assumed to be deposited at a constant rate depending on the location of the peatlands (Turetsky et al. 2004). The total ^{210}Pb activity in sediment contain of supported ^{210}Pb, which is produced from the in situ radioactive decay of natural ^{236}Ra in sediment, and unsupported ^{210}Pb from atmospheric fallout deposition (Turetsky et al. 2004). The Constant Rate Supply (CRS) model is used to calculate the ages and uncertainty errors of the unsupported ^{210}Pb activity through assuming a constant rate of supply of fallout and the documented half-life of 22.3 years (Appleby 2008). In contrast, ^{137}Cs atmospheric fallout was due to the testing of nuclear weapons from 1954 to 1963 and Chernobyl reactor fire in 1985, with a half-life of 30.2 years (Appleby 2008). As the ^{137}Cs nuclide has high mobility in acidic peat and might not be well-preserved, peak concentrations are used to verify the ages of peat sediment that accumulated after the 1963 and 1985 nuclear events (Turetsky et al. 2004).

To obtain peat accumulation ages with low uncertainty, the peat ages and the associated error bars from ^{14}C and ^{210}Pb dating are inputted into age-depth models such as r*Plum* (see full description in Aquino-Lopez et al., 2020), which is the integration of the *Bacon* (see Blaauw and Christen 2011) and CRS models. The *rPlum* model uses Bayesian statistics to reconstruct accumulation rates as a function of depth by using gamma autoregressive analyses to capture the non-linearity nature of peat deposition (*Bacon* model), and the radioactive decay (half-life) equations of ^{210}Pb activity (CRS model). The *rPlum* model incorporates different case scenarios in its piece-wise linear models, which account for different peat cores conditions, such as: (i) less compaction towards the peat surface, (ii) drastic changes in sedimentation at around 15 cm depth, and (iii) cyclic and periodic changes (Aquino-Lopez et al., 2020).

Despite the widespread use of radiocarbon and radionuclide dating, there are possibilities of obtaining inverse peat deposit ages with increasing depths, as a result of remobilisation of peat and plant matter through bioturbation or disturbances (Benavides et al. 2013; Weiss et al. 2002; Kilian et al. 2000; Van Geel and Mook 1989). Elements in peat can also be remobilised due to the “reservoir effect”, which is caused by the uptake of carbon by plants/fungi from an older peat (older ^{14}C ages) and the growth of younger rootlets in older peat (younger ^{14}C ages) (Kilian et al. 2000). However, contamination from remobilisation can be avoided through dating specific macro-fossils such as mosses, seeds or pollen found in peat (Reimer et al. 2013; Van Geel and Mook 1989). In addition, comparisons to “pollen density dating” (Middeldorp 1982), which is a proxy for peat accumulation rates, and known historical archives (Hapsari et al. 2021) can be made to provide validations of peat ages.

Tropical Peatland Degradation and Restoration Studies: Filling the Gaps Using Long-Term Data

There are gaps in knowledge regarding the dynamics/processes of degradation and restoration in tropical peatlands, which include a lack of long-term understanding to provide information on tropical peatland ecosystem feedback mechanisms, carbon losses caused by peatland degradation, and pre-degradation baselines and timeframes for recovery (Harrison et al. 2020; Dommain et al. 2016; Osaki et al. 2016; Hirano et al. 2014). The following sections will discuss how the long-term data from palaeoecological and geochemical proxies can help provide this information.

The Peat-Hydrology-Vegetation Feedback Mechanism

In tropical peatlands, the peat substrate, hydrological and biotic systems are inter-linked, meaning that any degradation to one component of the system may disrupt the conditions of the other at various spatial and temporal scales (Harrison et al. 2020; Dommain et al. 2016; Harrison 2013; Frolking et al. 2010; Ritzema, 2008; Price et al. 2016). For example, deforested tropical peatlands possess low hydraulic conductivity (permeability) (0.0004 to 0.3 m / day) and this will result in reduced groundwater level, peat decomposition from microbial activity and peat subsidence, plus leaching of humic acid, dissolved organic carbon (DOC) and sediment flux into rivers (Gallego-sala et al., 2016; Price et al. 2016; Sumawijaya 2006). Furthermore, the peat and groundwater conditions determine the vegetation structure, species composition and diversity, and zonation in tropical peatlands (Page et al. 1999; Cameron et al. 1989). Similarly, changes in vegetation over time will influence hydrological processes, such as infiltration of the incoming rainfall and evapotranspiration rates (Wösten and Ritzema 2001). Changes in vegetation species (forest vs. open-canopied vegetation) can also influence degraded (i.e. drained) peatlands as the absence/presence of tree shading influences peat decomposition rates due to temperature changes, as well as through the type of peat substrates accumulated (i.e. aromatic- and aliphatic-derived compounds) (Jauhianen et al., 2016; Lampela et al. 2014; Yule and Gomez 2009).

Long-term data from palaeoecology and geochemical proxies can help improve our understanding of these processes and feedback mechanisms, in particular identifying how changes in one system (i.e. hydrological, vegetation or peat) may impact the others over long time periods, and quantifying rates and levels of recovery following different types of identifiable disturbance event (e.g. fire). The lags in responses between the feedback processes of the peat, hydrology and vegetation system can also provide information on secondary (lagging) losses of the degradation, such as carbon losses over time from decomposition of deep peat layers (Riedinger-Whitmore 2016; Belyea 2009). In turn, this will also enhance understanding of the resilience of tropical peatland systems, and the stabilising (negative feedback) and destabilising (positive feedback) mechanisms that impact them (Page and Baird 2016; Belyea 2009). This information will be valuable for peat-hydrology models.

Understanding Carbon Losses due to Tropical Peatland Degradation

The estimation of lagged carbon losses is limited in existing tropical peatland studies. Recent estimates of carbon storage losses, carbon dioxide emissions and vegetation losses from large-scale events (i.e. fire) have been based on the immediate carbon losses calculated from observed data. For example, the annual net carbon dioxide emissions from SEA tropical peatlands arising from surface peat and biomass (vegetation) fire events were estimated at 640Mt y $^{-1}$ carbon dioxide equivalent (CO_2e), but peat decomposition through heterotrophic respiration was estimated to increase the total emission values by an additional ca. 600Mt y $^{-1}$ CO_2e (Jauhiainen et al., 2016; GFED, 2016; Hooijer et al., 2010). However, these estimates usually did not include the lagging emissions of oxidative decomposition of exposed deep organic-rich peat layers during subsidence and prolonged changes to the groundwater level and vegetation. According to Hooijer et al. (2010), the oxidative decomposition of peat in drained peatlands was estimated to produce CO_2e emissions of up to 518 Mt y $^{-1}$ in Indonesia in 2006, which was equivalent to 1.3 to 3.1% of the mean annual global CO_2 emissions from fossil fuels. In addition, carbon is lost as dissolved organic carbon (DOC) and particulate organic carbon (POC) from decomposed peats leaches into waterways (Fig. 1) (Jauhiainen et al., 2016; Moore et al., 2013; Moore et al., 2011; Alkhatib et al., 2007; Baum et al., 2007). Jauhiainen et al. (2016) estimated that the annual DOC and POC fluxes were 32 to 68% larger in peatlands affected by fire when compared to a previous non-fire year over the same time interval. However, these estimates can be variable over time as the CO_2e emissions, and DOC and POC measurements, were obtained from short-term studies and in response to the occurrence of large-scale degradation (Moore et al., 2013; Moore et al., 2011).

Palaeoecological data, in combination with geochemical proxies, can be used as additional data sources to obtain a more complete understanding of carbon losses (CO_2 emissions and fluvial carbon) from peatland degradation. From long-term TC data of deep peat, carbon storage in peat can be estimated, as can changes between years in total atmospheric and fluvial carbon emissions (Carcaillet et al., 2002). This information, together with long-term, temporally defined data on peatland conditions and fire events obtained from palynomorphs, testate amoeba and charcoal proxies will provide information on the type of degradation driving the carbon loss and the ability of the peatland ecosystem to recover post-disturbance. Although it is difficult to separate the losses of CO_2 and fluvial C from the carbon data, Mn concentration in peat can provide details on the fluvial leaching conditions of the decomposed peat, which is an indicator of the proportion of the loss of fluvial C from the total carbon losses (Muller et al., 2008).

Pre-Degradation Baselines and Timeframes for Recovery in Restoration Efforts

Most restoration effort in tropical peatlands in SEA incorporates peat rewetting strategies to raise groundwater level of drained peatlands, limit peat subsidence, decrease fire risk and support native vegetation recovery (Giesen and Nirmala 2018; Dohong et al. 2018; Dommain et al. 2016; Dohong and Lilia 2008). In Indonesia, government regulations require the maintenance of peatland groundwater level to a maximum 40 cm below the peat surface through managing land uses or constructing dams in canals (Giesen and Nirmala 2018; Wösten et al. 2008). However, this requirement is not met in all disturbed peatlands as shown by a one-year monitoring study in Central Kalimantan which reported the water table level of 40 cm below the peat surface was constraint to areas near the dams while the peatlands away from the canals had poor ability to store water due to collapsed peat macro-pore structure from fires and peat decomposition (Dommain et al. 2016; Ritzema et al. 2014; Dohong and Lilia 2008). Furthermore, the use of this standard may fail to consider the feedbacks between long-term local peat, hydrology and vegetation conditions (Giesen and Nirmala 2018; Giesen and van der Meer,2009). As such, the use of multiple proxies such as testate amoeba, and C isotopes and concentration will be able to provide the past baselines and thresholds of groundwater level conditions needed to ensure healthy peat conditions (where C is sequestered) and help establish the time needed for post-disturbance recovery.

In addition to rewetting, tropical peatland revegetation guidelines in SEA have adopted a standard prescription of planting native, flood tolerant and/or economically desirable plant species (Dommain et al. 2016; Blackham et al. 2014; Giesen and van der Meer, 2009). The success of such efforts may be impacted by the presence of local regeneration barriers in degraded peatlands, such as limited seed dispersal, low peat nutrients availability and fluctuating groundwater levels (Graham et al. 2017; Dommain et al. 2016; Page and Baird 2016). Moreover, revegetation monitoring studies are usually based on survival and/or growth rates of seedlings which can change over time and do not demonstrate if these strategies aided the overall development of peat and other system functions (Giesen and Nirmala 2018; Yule et al. 2016). Palynology can be used to identify native vegetation species that can tolerate or adapt to groundwater level changes and variable nutrient conditions in peat, and survive and recover from fire events in peat swamp forest (Gałka et al. 2017; Muller et al. 2012). Furthermore, long-term information on threshold local hydrological and peat conditions for vegetation survival, and insights of the effects of past human alterations on vegetation dynamics (i.e. indigenous burning practices) can be used to guide current revegetation efforts (Riedinger-Whitmore 2016; Page and Baird 2016; Belyea 2009).

Incorporating Long-Term Data into Peat-Hydrology Models for Restoration and Monitoring Efforts

Studies in tropical peatlands are increasingly using peat-hydrology models to project the responses and pathways of planned rewetting interventions in peatland systems and to test the efficacy of restoration activities (Page and Baird 2016; Kurnianto et al. 2015; Dommain et al. 2016; Baird et al. 2012; Wösten et al. 2006; de Vries 2003). Previous models used in tropical peatlands include the Tropical Holocene Peat Model (HPMTrop) (Kurnianto et al. 2015) and DigiBog (Baird et al. 2012) models, which simulated peat accumulation as a function of the terrain (peatland size and shape), hydrological conditions (water table), vegetation composition and inputs (litter), and hydraulic conductivity of peat (permeability) (Page and Baird 2016; Baird et al. 2012; Frolking et al. 2010). These models are complemented by hydrological-based models, such as SIMGRO and MODFLOW, to project water-table depths through simulating water flow in the saturated and unsaturated zones, and surface flow of the peatlands (Ritzema and Jansen 2008; de Vries 2003). These models are comprehensive as they considered the theoretical feedback mechanisms, long-term climate variables using regional palaeoclimatic proxies, recent peatland conditions, and anthropogenic changes to peatlands such as irrigation, drainage and groundwater use (Kurnianto et al. 2015; Ritzema and Jansen 2008; de Vries 2003).

Despite their valuable applications, the recent and long-term hydroclimate conditions used in these models are based on estimations from regional studies rather than site-based measurements (Dommain et al. 2016; Page and Baird 2016). Obtaining such measurements from multiple individual sites of varying condition and (disturbance) histories could thus help refine and improve these models. In addition, long-term records from palaeoecological or geochemical proxies in peat have not been incorporated into these models due to concerns associated with corrupted signals of peat archives from diagenetic overprinting associated with consecutive degradation processes (Dommain et al. 2016; Page and Baird 2016; Morris et al. 2015). While this concern is valid, the use of multiple proxies to verify the accuracy of the long-term data and reduce the uncertainties will provide reliable local pre and post- degradation conditions for the models (Dommain et al. 2016; Kurnianto et al.,2015).

Furthermore, long-term information on the trajectory of change, and the lags and thresholds of the responses of peat, hydrology and vegetation system to tropical peatland disturbances will help to refine the projections of the models (Page and Baird 2016). For example, Cole et al. (2015) study in coastal peatlands at Serawak, Malaysia, demonstrated resilience of peat swamp forest vegetation to past changes such as drought events and forest fires between 500 and 2000 years ago, but contemporary climatic changes of higher magnitude coupled with intensive human land use change in the last 500 years led to decline in the peat swamp forest community. These observations on the ability of the vegetation and peat system to adapt to different intensities of change is important information to be included in peat-hydrology models to assist with predicting restoration responses in the face of future climatic warming and increased climatic variability.

Information on thresholds and lags in the feedback mechanisms between these systems can be inputted into peat-hydrology models to improve projections of peat accumulation/decomposition and groundwater level. Machine learning techniques, such as multivariate adaptive regression splines and regression tree learning, can be applied to the long-term data to obtain algorithms of the feedback mechanisms, thresholds and lags of the peatland system (Raj and Gharineiat , 2021; Monteleoni et al. 2013; Mannila et al. 1998). The inclusion of algorithms that parametrise the different feedback mechanisms described here into peat-hydrology models will improve projections of peat accumulation and aid in our ability to assess whether peat in the tropics will continue to accumulate under future climate change and other forms of anthropogenic disturbance. Furthermore, the same palaeoecological and geochemical proxies can be used as a monitoring tool to help evaluate changes in peat conditions post-restoration and to examine natural regeneration of degraded peatlands, thus helping refine future restoration planning (Page and Baird 2016).

Concluding Remarks: Long-term Data for Restoration Solutions

This review has illustrated the value of both palaeoecological and geochemical proxies for understanding long-term responses to anthropogenic and climate-driven disturbances, natural recovery from these, and the application of this knowledge in the context of current efforts to actively restore degraded tropical peatlands. The use of palaeoecological and geochemical proxies in tropical peatland studies provides potential to reconstruct long-term vegetation dynamics, moisture availability, groundwater level, peat pH, peat decomposition and accumulation rates, as well as providing information on degradation history (i.e. fire events, weathering, erosion and pollution). Despite the limitations of each individual proxy, the use of a multi-proxy approach to verify and minimise any uncertainties can help to improve the reliability of long-term reconstructions. Through using long-term data to understand disturbance impacts, ecosystem dynamics and secondary losses (e.g. carbon loss) from feedback processes between the peatland systems, our ability to predict responses of peatlands to restoration efforts and degrading activities can therefore be enhanced, which should lead to improved management outcomes.

Finally, the increased use of long-term data from multiple individual tropical peatland sites of varying conditions and disturbance history in peat-hydrology restoration models to provide past information of the dynamic feedback mechanisms, natural variability, resilience and lags between responses of each system will enhance the projections of future degradation and restoration efforts. This offers potential to enhance tropical peatland rewetting and revegetation efforts through incorporating long-term information to support selection of vegetation species suitable for replanting, identify the optimum groundwater level needed to support revegetation plans in degraded peatland and predict recovery time following disturbance. In addition, the same palaeoecological and geochemical proxies can be used for peat and hydrology monitoring purposes to track the effects of the restoration efforts in the degraded peatlands, thus facilitating effective adaptive management of these crucial ecosystems.

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